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Fuel Reduction Practices and Their Effects on Soil Quality

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Cover photos: Upper left—Matt Busse, middle left—and lower left—Bob Carlson, right—Carol Shestak.

Abstract

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Soils sustain our terrestrial ecosystems, help fuel plant growth, and govern key ecosystem services such as the storage and provision of clean water, degradation of toxic compounds, and regulation of atmospheric gases. Preserving the integrity of soil thus is an earnest responsibility of land stewardship in the United States. This report provides a synthesis of soil chemical, biological, and physical responses to various prescribed fire and mechanical thinning practices and offers practical considerations for use in fuel reduction planning. A wide range of current topics, identified in a nationwide survey of natural resource managers, is discussed in detail: (1) ecological consequences of prescribed fire on soil heating, water repellency, and soil nitrogen release; (2) whole tree harvesting and nutrient removal; (3) soil compaction; (4) masticated fuel beds; (5) pile burning; (6) cumulative effects of fire and thinning; (7) coarse woody debris; and (8) soil in a changing climate. We submit that with thoughtful planning and implementation, reducing fuels while proactively managing our soils can be complementary outcomes.

Keywords: Prescribed fire, mechanical thinning, compaction, soil fertility, soil productivity.

Contents

Introduction

Soils are exceedingly complex. They contain a rich array of minerals and organic compounds, chemical reactions, physical states, and biological diversity and are responsible for countless ecological functions that belie their common appearance and colloquial namesake (dirt). Most importantly, soils sustain life by providing plants with essential nutrients, water, and physical support. They also play a major role in providing a clean water supply, degrading toxic compounds, supplying novel antibiotics for human health, and circulating greenhouse gases. Thus, properly functioning soils are a prerequisite of terrestrial ecosystems and society alike, as discovered by several ancient civilizations that failed to value their soils, to their eventual dismay and ruin (Diamond 2005).

But what of fuel management practices—are they of sufficient severity to alter soil functions? If so, should our soils be protected unilaterally, or do the benefits of fuel reduction practices outweigh possible impacts to soil? With the variety of management practices (from benign to intensive) and soils (from highly buffered to fragile), these questions are best answered on a site-by-site basis. A good starting point then is to rank the effects of fuel treatments relative to other global pressures on soil. Severe changes in soil properties result most often as a consequence of land use change (e.g., clearing of forests for agriculture production), acute disturbance (e.g., chemical contamination) or by the four main pathways identified by Lal (1997): (1) erosion from insufficient ground cover, (2) compaction from mechanized equipment, (3) nutrient depletion from unsustainable harvesting, and (4) chemical degradation from salt-affected irrigation water. Fuel reduction using best management practices is not a classic fit in these categories; however, it can affect soil erosion, compaction, or nutrient availability in certain circumstances (fig. 1) (Grigal 2000, Neary et al. 2005).

A fundamental sidebar in discussing the effects of fuel reduction practices is the knowledge that soil is not renewable in our lifetime, and it typically requires hundreds to thousands of years to develop (Jenny 1941, van Breeman and Buurman 2002). Thus, the consequences of inappropriate management, whether inadvertent or intentional, can be severe and have pushed land agencies to promote a conservative or protective approach to soil management (see USDA FS 2005). However, soil scientists also recognize that many soils are fully capable of recovering from disturbance given sufficient time (Lal 1993). Griffiths and Swanson (2001), for example, found substantial changes in soil biological properties following harvesting in an eastern Oregon conifer forest, yet full soil recovery to preharvest conditions occurred once the developing forest reached 15 to 40 years of age. This underscores

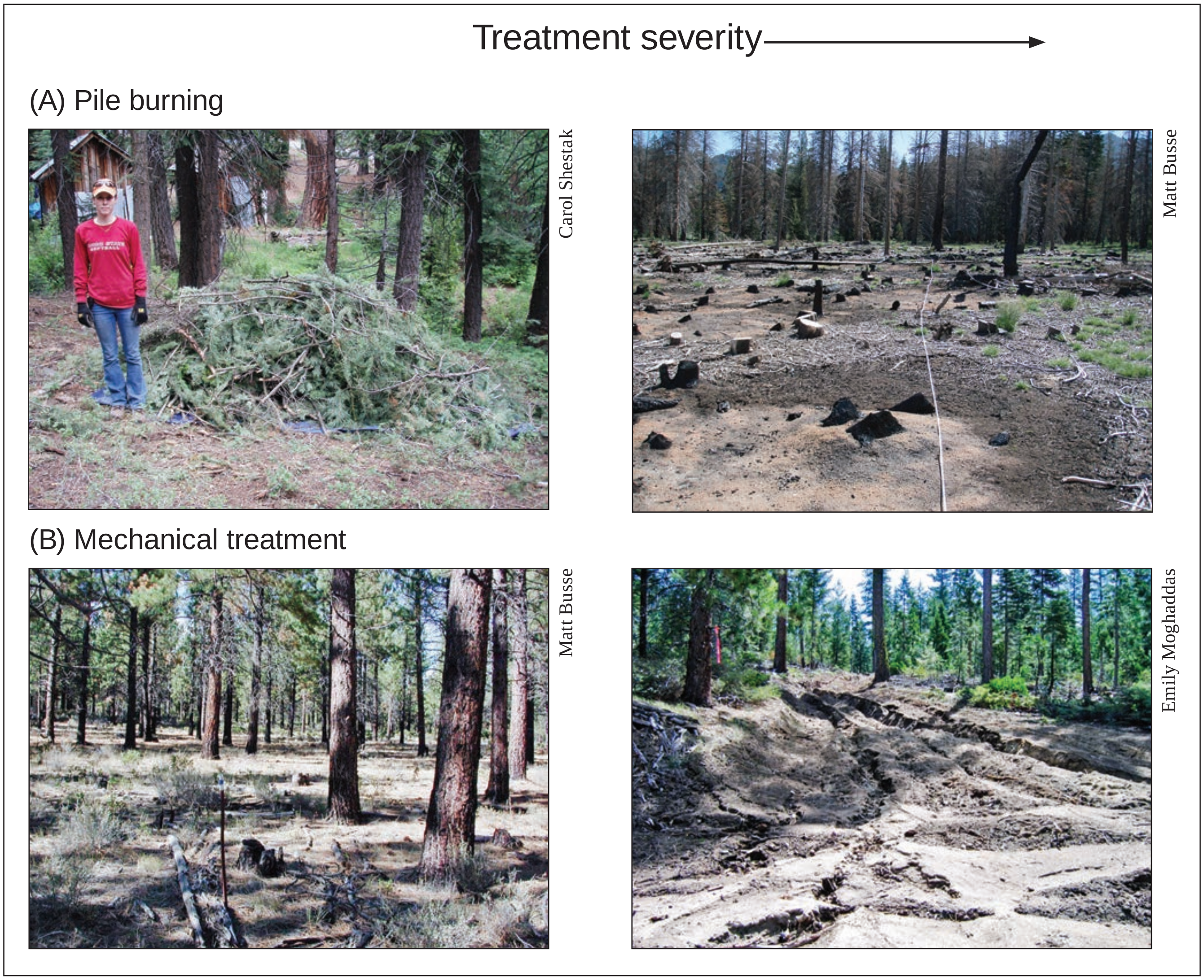

Figure 1—Comparison of pile burn (A) and machanical treatments (B) and their effect on soil. Low soil impact from small, widely spaced burn piles (upper left); high soil impact from severe burning of closely spaced piles (upper right); low soil impact from past thinning on frozen soil (lower left); high soil impact from past thinning on wet soil (lower right).

a primary challenge in managing soils: How do we blend a conservative approach that protects the natural resource while also acknowledging the resilient ability of many soils to recover from disturbance?

Two additional issues come into play when assessing the effects of fuel reduction practices on soils. First, reducing fuels and restoring landscapes is an iterative process that requires multiple treatments throughout the life of a stand. The effect of repeated treatments on soil is not well studied, which moderates our confidence to properly manage soil. Or, as stated by Davis et al. (2010), "Small changes in soil properties as a result of soil disturbance that might seem scientifically insignificant at one point may become significant when the soil disturbance is repeated in multiple rotations as an acceptable management practice." Second, the objectives for managing soils need to be clarified within a societal framework. Is soil strictly a medium for forest or agricultural productivity, or is its greatest value related to water quality, biodiversity, carbon (C) sequestration, or a combination of these services? Burger et al. (2010) point out that these services are not necessarily complementary and that the best soil management practices for maintaining biodiversity in some forests may not be the best for maximizing tree growth or for reducing greenhouse gas emissions. In this regard, we canvassed resource managers and field experts in a nationwide survey for their definition of "soil productivity." Fewer than half of the respondents (55 out of 124) asserted that soil productivity relates to the ability to support plant growth, while the others believed that the term applies to a broader context that includes water quality, erosion, and biodiversity.

Assessing the health of soils must be based on objective scientific analyses of changes in soil properties relative to pretreatment conditions. This task is made difficult by the wide variation in soil properties across the United States and, at times, by the unpredictable nature of responses of soil to fuel treatments. With this in mind, our purpose in this synthesis is to identify (1) what constitutes a healthy soil (one that is resilient and easily restored following single or multiple treatments) and (2) the environmental conditions, soils, and treatments that may lead to adverse damage.

Wildfire Trends and Fuel Reduction Practices

A century-long U.S. policy of fire suppression and exclusion has disrupted the natural fire regime of many ecosystems, resulting in excessive accumulation of dead vegetation, increased stand densities, and a shift to plant species that are not adapted to fire. Thus the paradox of successful fire exclusion: the more efficient at suppressing wildfires we become, the larger the wildfire problem becomes (Brown and Arno 1991) (fig. 2). Early logging practices have added to the problem by

Curt Hudson, The Associated Press

Figure 2—Wildfire burning through Wharton State Forest in southern New Jersey.

removing large, fire-resistant trees, resulting in extensive forests of smaller trees (Agee and Skinner 2005). In the contiguous United States, 36 to 81 million ha (90 to 200 million ac) are estimated to have unnaturally dense fuel accumulations (USDA GAO 2007). Additionally, warmer temperatures, lower precipitation, and reduced snowpack since 1986 have increased the duration and intensity of the wildfire season in the Western United States (Westerling et al. 2006).

Area burned by wildfire peaked in the United States in 2006 (fig. 3), with the number of hectares burned generally greatest in Western and Southern States. Wildfire area from 2004 through 2007 reached 2.8 to 3.6 million ha (7 to 9 million ac) and suggests a baseline for fire activity that includes increased “megafires” and extreme wildland-urban interface (WUI) incidents. With climate change and urban expansion, this baseline may increase, and extreme WUI wildfires may become a more common occurrence.

The extent of wildfire area differs greatly between the different regions of the United States and also between years in the same region. For example, the Rocky Mountain region approached 1.2 million burned hectares (3 million ac) in 2007, but dropped to >162 000 ha (400,000 ac) in 2008 (fig. 4). In the drought year of 2008, wildfires consumed >600 000 ha in the Texas/Oklahoma region as compared to about 40 000 ha in 2007. These year-to-year shifts across regions are indicative of rapidly changing fuel conditions that accompany periods of drought, unusual precipitation patterns, or recent fuel reduction practices and wildfires.

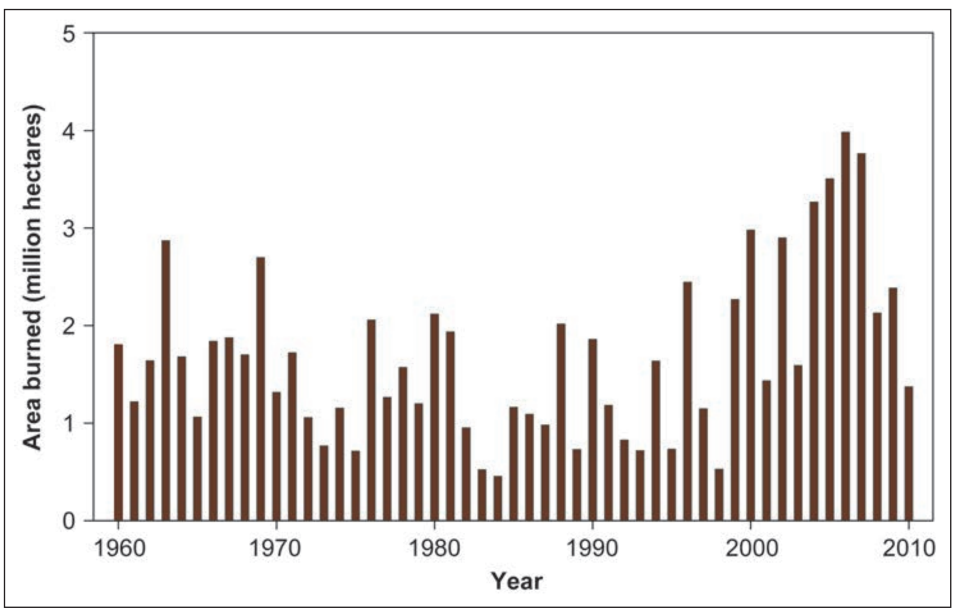

Figure 3—Total area burned by wildfire in the United States from 1960 through 2010. Data are from the USDA NIFC (2010).

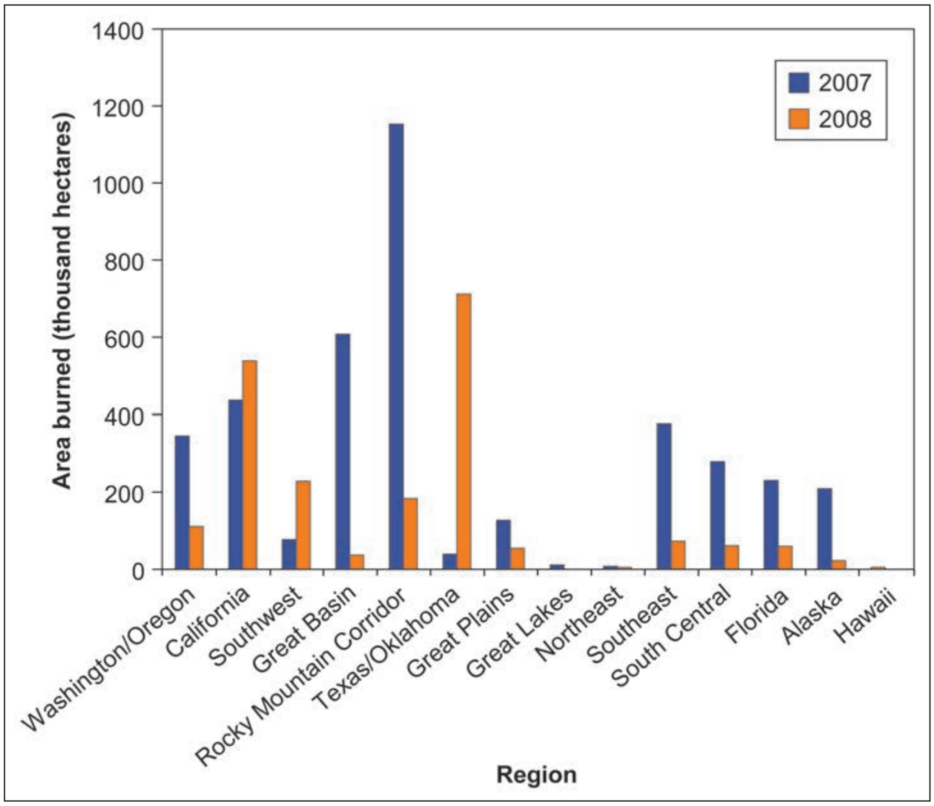

Figure 4—Wildfire acreage compared across different regions in the United States for 2007 and 2008. Data are from the USDA NIFC (2010).

Most of the acreage burned in the United States occurs in just a few wildfires that burn under extreme conditions (The Brookings Institution 2005). These large "megafires," along with extreme fires at the WUI, have resulted in a recent dramatic rise in fire suppression costs (fig. 5). Megafires often burn during severe fire conditions (periods of prolonged drought with large accumulations of dead and live biomass) and exhibit extreme fire behavior characteristics. Annual fire management activities (mainly fire suppression) accounted for only 13 percent of the U.S. Department of Agriculture Forest Service (USFS) total budget in 1991, but increased to 48 percent by 2009 (Ingalsbee 2010) as the annual cost of federal fire suppression surpassed $1.5 billion beginning in 2002 (fig. 5). In 2006, 20 wildfires on USFS land alone cost $500 million, or nearly one-third of the year's expenditures on fire suppression (Ingalsbee 2010). Although megafires represent <1 percent of all wildfires, they account for about 85 percent of total suppression-related expenditures on federal lands (The Brookings Institute 2005).

Nationwide efforts to reduce the cost of fighting wildfires and to restore the function and health of our wildlands have relied heavily on the use of prescribed burning (understory burning, slash burning, pile burning), various mechanical

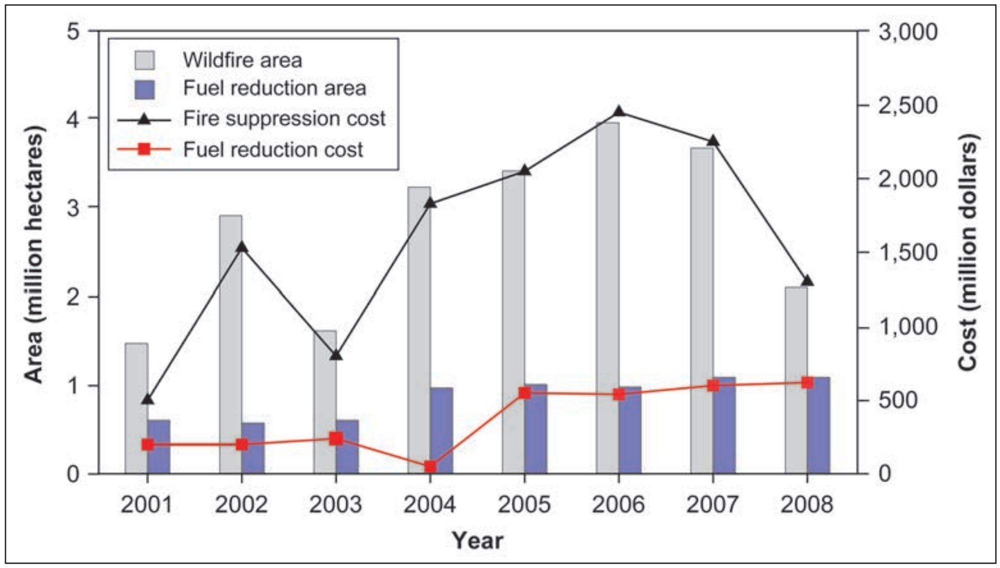

Figure 5—Wildfire and fuel reduction area in the United States and their estimated costs from 2001 through 2008. Data are from the USDA NIFC (2010). Costs include all federal agencies.

Figure 6—Examples of fuel reduction methods: (top left) prescribed fire, (top right) piling and burning, (bottom left) thinning, and (bottom right) masticating.

practices (thinning, masticating/chipping, mowing), and their combinations (fig. 6). Collectively, these treatments reduce wildfire hazard (fuel load and fuel continuity), limit fire intensity and rate of fire spread, restore forest health, protect life and property, provide defensible space that aids fire crews in protecting communities, and protect environmentally sensitive areas from catastrophic wildfire. Fuel reduction programs and the selection of preferred treatment practices differ among federal agencies as a consequence of their unique land stewardship missions. For example, both the USFS and the Bureau of Land Management use combinations of prescribed burning and thinning to support their multiple-use policy. In contrast, the mission of the Fish and Wildlife Service focuses on the conservation of wildlife habitat, and thus they primarily conduct prescribed burns to improve habitat. For like reasons and because fire is a natural process, prescribed burning is the preferred fuel treatment of the National Park Service. An alternative approach to these fuel reduction approaches is to allow fire to resume its natural role. Allowing wildfires to burn as a resource benefit, a policy formerly known as Wildland Fire Use (WFU) or the “Let Burn” policy, is the structured management of naturally ignited wildfires with the goals of restoring natural fire regimes and reducing heavy fuel accumulation (USDA and USDI 2005). In 2005, there were 96 000 ha (237,000 ac) of WFU across all agencies, a drop from 198 000 ha (489,000 ac) in 2005 (USDA NIFC 2010). The effects of this wildfire practice on soil resources have not yet been adequately studied and will not be a topic of this synthesis report.

A comparison of federal lands treated by prescribed burning versus mechanical treatments is shown in figure 7. Fire is the most common fuel reduction practice throughout the United States, with the highest acreages in 2008 found across the country’s southern perimeter. Use of mechanical treatments was more common in the Western United States with >280 000 ha (>700,000 ac) treated in 2008. We presume year-to-year treatment of acreage is highly variable owing to changing weather conditions for fire prescriptions, budget constraints, and wildfire activity.

Fuel reduction practices can affect a multitude of soil properties, which poses an interesting question for land managers: Is there a subset of soil properties that can be monitored that is affordable, easy to sample and analyze, and ecologically meaningful? This issue has been debated among soil scientists for decades without clear resolve (see the appendix for a more detailed discussion of soil quality principles). Further, soil properties most indicative of detrimental changes differ between fuel reduction practices, making comparisons among treatment types problematic. Indicators based on soil physical properties (porosity, water infiltration, soil strength, compaction) are most commonly used to identify soil changes following

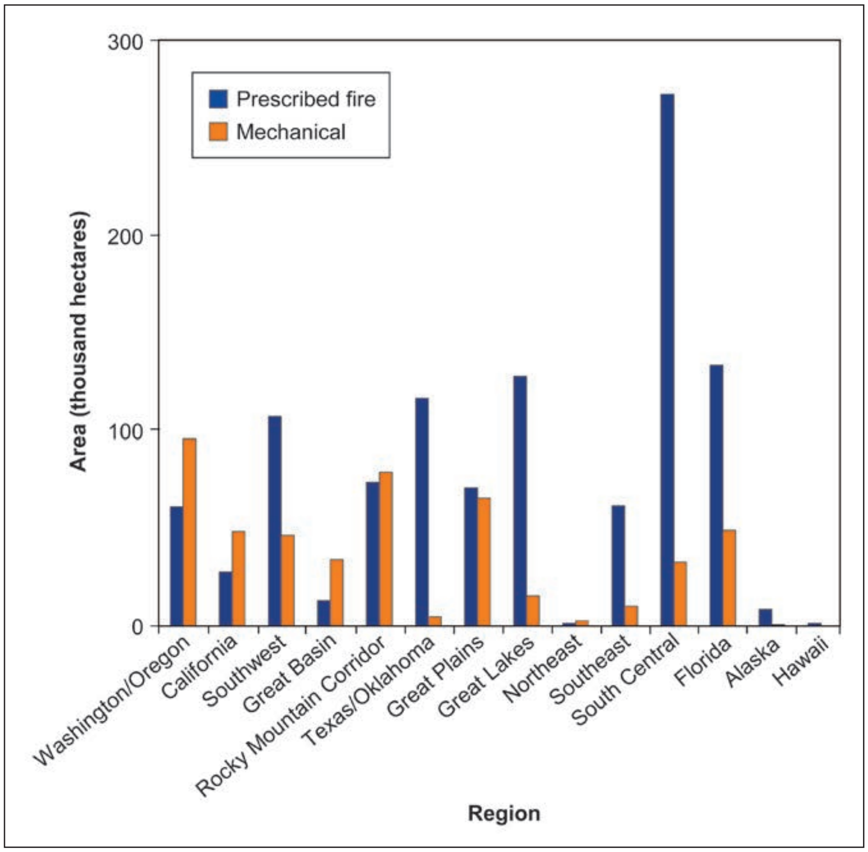

Figure 7—Regional acreage treated by prescribed burning and mechanical thinning in 2008, excluding state and private lands. Data are from the USDA NIFC (2010).

mechanical treatment. Prescribed burning, on the other hand, results in soil chemical and biological effects that include oxidation of surface and soil organic material, changes in nutrient availability and pool size, changes in pH, and lethal heating to biota and fine roots. A simple, one-size-fits-all soil measurement that accurately documents change does not exist.

Wildlands are not fireproof. Fuels will continue to accumulate and wildlands will burn. Climate change likely will alter the majority of ecosystems in the United States, dramatically changing their structure, composition, and distribution of species, and thus enhance the role wildfire will play in these systems (Chambers and Pellant 2008). Consequently, the need for fuel reduction programs will continue to grow, as will the importance of managing these programs with soil quality and conservation in mind. In this report, we offer an indepth examination of 12 soil-fuel reduction issues that were identified as key informational needs in a nationwide survey of resource managers. In addition, a detailed introduction to forest soil quality principles is presented in the appendix.

National Survey of Land Managers—Soil Issues and Concerns

We surveyed natural resource managers and practitioners to assess current perceptions of fuel treatments and their potential for causing detrimental soil disturbance. Survey questions appraised (1) whether soil productivity is an important consideration of managers when conducting fuel reduction treatments, (2) which soil properties are of greatest concern, and (3) what unanswered questions or informational needs managers have regarding soils. Our intentions were to capture current attitudes toward soil management and to identify perceived knowledge gaps, and then to use the information gained in selecting topics for discussion in the succeeding section on ecological effects and management considerations.

A survey of government personnel representing the U.S. Department of Agriculture Forest Service (USFS); U.S. Department of the Interior Bureau of Land Management (BLM), National Park Service (NPS), Fish and Wildlife Service (FWS), and Bureau of Indian Affairs (BIA) was conducted in 2008. Survey recipients were limited to federal employees (owing to legal constraints) involved either directly or indirectly in fuel reduction projects as identified using government databases, workshop and seminar lists, and environmental document reports. We used SurveyMonkey,[1] a Web-based online survey program (http://www.surveymonkey.com), to distribute the survey to federal employees across all regions of the United States.

Of 1,146 individuals surveyed, 209 responded (18 percent). Forest Service employees accounted for 78 percent of the responses, followed by BLM (11 percent), NPS (8 percent), FWS (3 percent), and BIA (1 percent). Fire-related disciplines accounted for the largest portion of respondents, followed by "soil" and "silviculture," with respondents in all disciplines averaging greater than 15 years of related work experience (table 1). Regional response was evenly distributed across the Western United States, while participation from the Southern, Eastern, and Alaska regions was noticeably lower.

Survey questions asked are as follows:

Question 1. How important is soil productivity when conducting fuel reduction treatments?

Most respondents defined soil productivity as the long-term ability of the soil to grow native vegetation (produce biomass), to support plant growth and community

[1] The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

Table 1—Number of survey respondents by discipline and region

Discipline	Total number of respondents	Years experience (mean)	Region	Number of respondents by region
Fuel management[a]	53	17	Pacific Southwest	35
Fire management[b]	46	18	Pacific Northwest	34
Soil	36	19	Intermountain	29
Silviculture	30	25	Rocky Mountain	26
Plant, wildlife ecology	13	25	Northern	24
Natural resources	10	24	Southwest	19
Fire ecology	9	16	Southern	11
Hydrology	6	19	Eastern	12
Other	6	11	Alaska	3

[a] Fuel reduction focus.
[b] Fire prevention, detection, and suppression operations.

composition, and to meet the requirements of sustainable ecosystems. About 8 out of every 10 respondents agreed that they take soil productivity into consideration-prior to and following fuel reduction treatments (fig. 8). This high percentage was consistent among regions and disciplines, indicating a fairly universal and positive attitude toward soil productivity.

Few respondents ranked soil as a primary concern, however. We provided six criteria for selecting a fuel reduction treatment: cost, effectiveness, ease of use, soil quality, environmental factors (such as wildlife, air quality or invasive species), and legal factors (such as the National Environmental Policy Act [NEPA] or litigation), and asked the respondents to rank each criterion from most important to least important when selecting fuel treatments. Treatments included prescribed/broadcast burning (understory or scattered slash), mechanical thinning, and all fuel reduction practices (thinning, fire, pile burning, masticating, and their combinations).

Respondents ranked treatment effectiveness (35 percent response) and treatment cost (31 percent response) as the most important criterion when selecting broadcast burning as a treatment (fig. 9). Few ranked soil quality (3 percent response) as the most important criterion. Treatment effectiveness was also the main criterion (55 percent response) given for conducting mechanical thinning treatments. For all fuel treatments combined, effectiveness (43 percent response) again was the most important consideration. Nearly all respondents ranked soil quality of low to moderate importance when selecting fuel treatments (fig. 10).

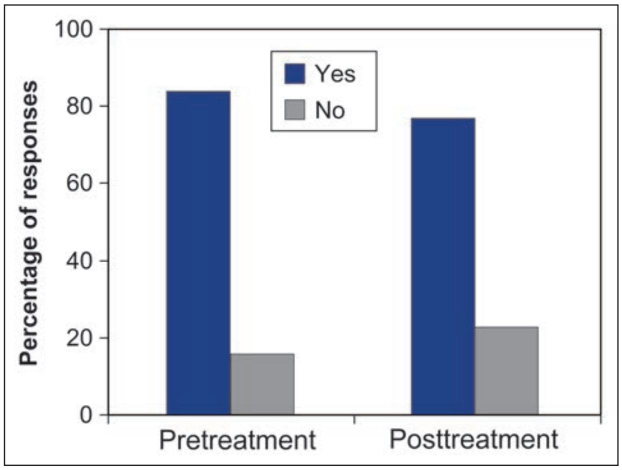

Figure 8—Survey response to the question, "Is soil productivity a consideration when conducting fuel reduction treatments?" (n = 153).

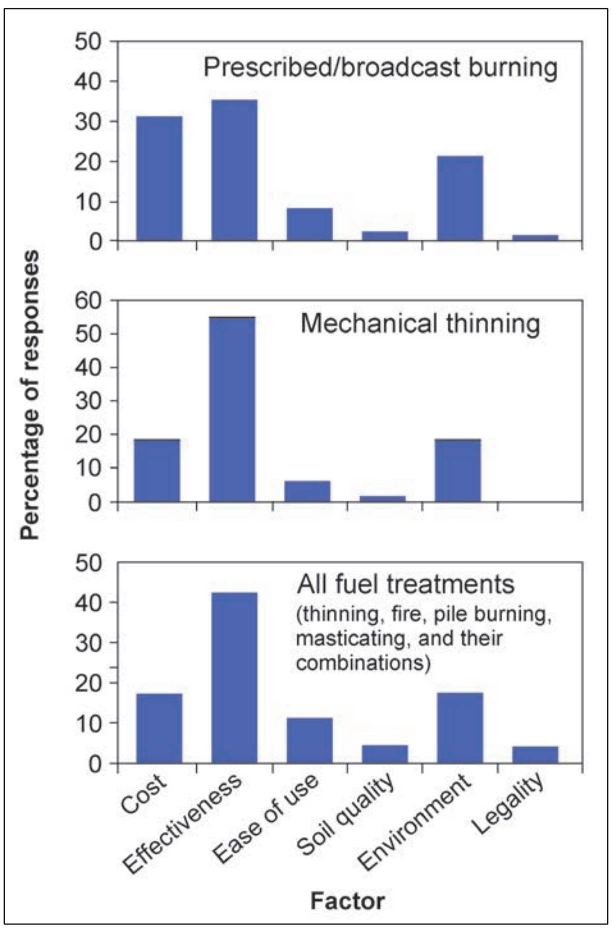

Figure 9—Survey response to the question, "What is the most important factor when selecting a fuel reduction practice?" (n = 131).

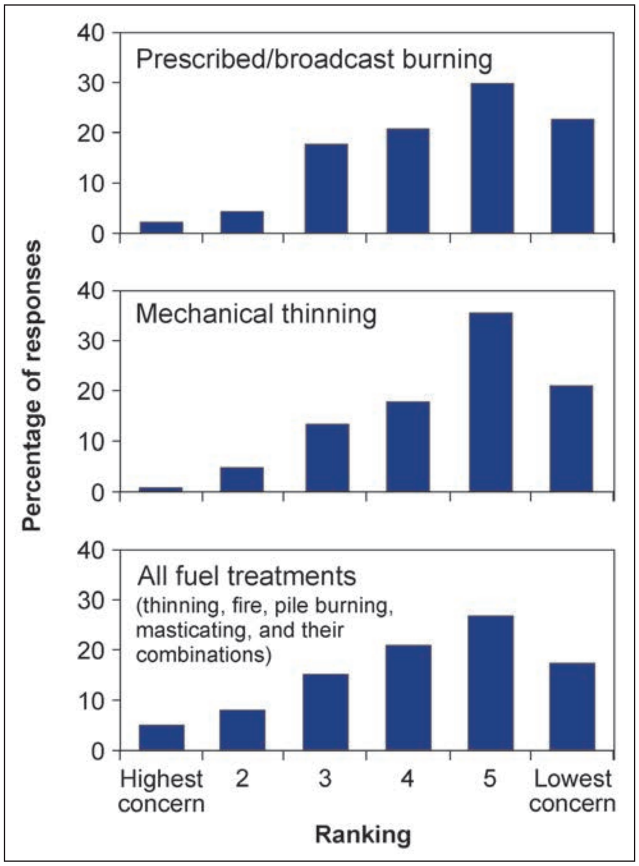

Figure 10—Survey response to the question, "How important are soil issues when selecting a fuel reduction treatment?" (n = 131).

Question 2. Which soil properties are of greatest concern when applying treatments?

We asked "What is your level of concern for various soil factors when conducting fuel reduction treatments (rating from 1 = high concern to 5 = low concern), and provided a comment box to allow each respondent to clarify their selections as needed. We assigned ratings 1 and 2 = "high concern," rating 3 = "medium concern," and ratings 4 and 5 = "low concern." Soil factors included compaction, erosion, forest floor depth, soil organic material (SOM), soil heating, nutrient status, soil pH, microbial and faunal health (soil biota), and mycorrhizae. Treatments were grouped into three categories: prescribed burning (broadcast and pile), thinning (mechanical, chain saw, mastication), and their combinations.

Treatments

Prescribed Burning

Forest floor depth was the greatest soil concern among respondents when applying fire, with 64 percent rating it high, followed by concerns for SOM, soil heating, erosion, and nutrient status (fig. 11). Concern for litter depth appeared to be closely related to nutrient protection, bare soil for seedling germination, and erosion control. Concern was relatively low for soil compaction, soil pH, and soil biota. Some respondents noted that prescribed fire is rarely hotter than a wildfire and that effects to soil properties are usually low. Others commented that prescribed fire generally will not have soil heating issues, but there should be awareness of postfire wind and water erosion. Both broadcast burning and jackpot burning (igniting concentrated fuels) were generally perceived as having positive ecological effects.

Ninety percent of fuel managers answered “yes” to conducting pile burns. Many commented that they preferred to burn piles during the late fall or winter when there is at least 2 to 3 inches of snow on the ground to help prevent fire escape and reduce mop-up time. Several respondents noted that air quality concerns and poor smoke dispersion makes winter pile burning contentious at times, however. Summer was the least preferred season for pile burning because of the high risk of fire escape, residual heat and soil damage, and greater mop-up efforts required.

Respondents noted that hand piles are typically smaller than 3 m (10 ft) in diameter and burn for about 8 hours or less, whereas mechanized piles are generally greater than 3 m in diameter and burn for 8 to 24+ hours. The largest piles reach

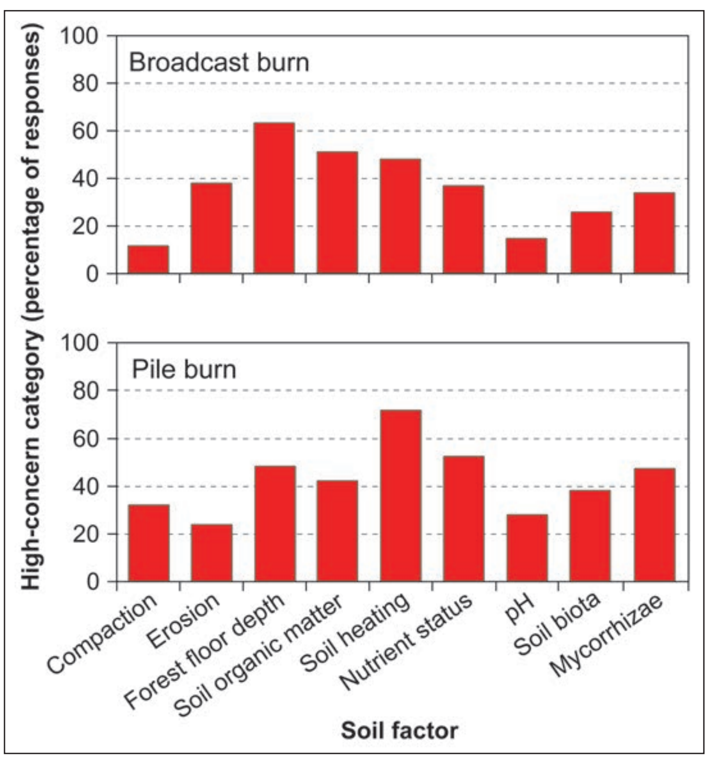

Figure 11—Percentage of survey respondents expressing “high concern” for the effects of fire on selected soil properties.

about 21 m (70 ft) in diameter and over 4.5 m (15 ft) tall, and are typically composed of whole-tree harvested material and burned at landing sites.

Soil heating was the major concern when selecting pile burning, with 70 percent of the respondents rating it a high concern (fig. 11). Concerns for nutrient status, forest floor depth, and mycorrhizae followed with responses of 52, 48, and 47 percent, respectively. Several respondents commented that pile-to-pile variability in burn intensity and duration played a significant role in the extent of soil heating. Although soil heating was a major concern, some respondents considered piles as a limited problem owing to the small area affected. However, others noted that they commonly have a large number of piles per acre. Postburn mitigation practices included (1) shoveling soil from outside the piles to recolonize the site and (2) using backpack blowers to spread the ash so that the site was colonized faster by native vegetation.

Thinning

Soil compaction was the greatest concern when using mechanical thinning, with 71 percent of respondents rating it a high concern (fig. 12). Erosion received a high rating from 50 percent of the respondents. In contrast, concern for thinning effects on soil chemical and biological properties was generally low. Several respondents

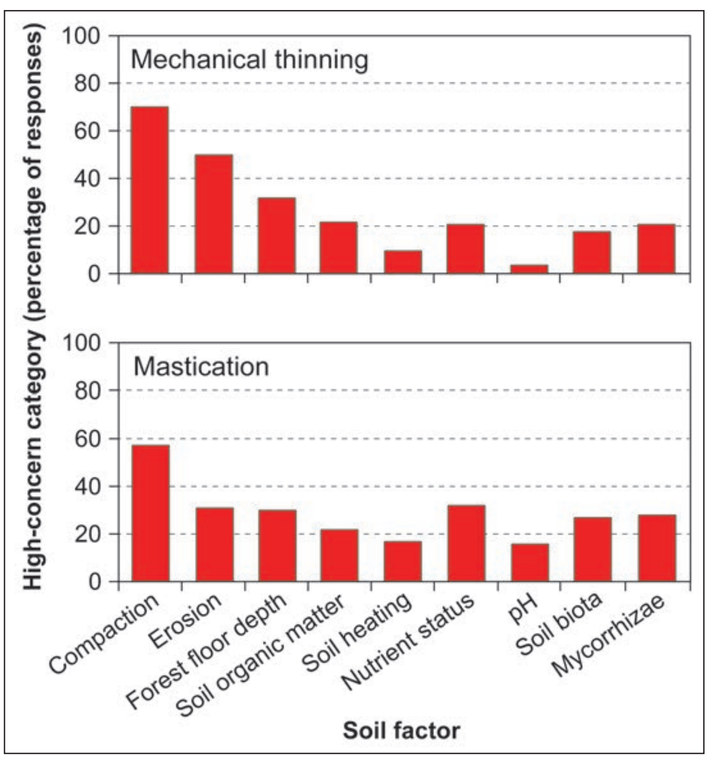

Figure 12—Percentage of survey respondents expressing "high concern" for the effects of thinning on selected soil properties.

commented that the size and type of equipment used when thinning are key considerations. Mitigating measures that were mentioned included (1) placing slash in the path of the equipment using cut-to-length forwarder/harvesters to lessen detrimental compaction, and (2) harvesting over snow to reduce soil impacts.

There was little concern for detrimental soil effects owing to hand thinning (data not shown). The “low concern” response was expressed by about 70 to 80 percent of respondents for all soil properties. Factors responsible for the lack of high concern included (1) no heavy equipment involved, (2) foot traffic is often on downed slash, and (3) most hand thinning is conducted on steep slopes.

Soil compaction was also a priority concern of 58 percent of the respondents for mastication treatments (fig. 12). Again, several respondents mentioned that the level of concern depended on the intensity of the treatment and size and type of equipment used. Most concerns focused on equipment and the amount of ground area covered when masticating vegetation. Mitigating measures that were mentioned included (1) using designated skid trails to reduce the amount of surface area affected, (2) masticating when soils were dry as soil moisture content greatly affects the impacts of compaction, (3) selecting the correct type of equipment for the job (4), using very low tire pressures, and (5) limiting the number of passes. There was some concern that deep residue layers, when present, would affect the soil carbon to nitrogen (C:N) ratio and related microbial processes such as nutrient mineralization.

Thinning and Burning

Soil properties that received the highest concern when combinations of thinning and burning were considered included compaction, erosion, forest floor depth, SOM, and soil heating (fig. 13). A number of respondents noted that fuel reduction treatments on federal lands must adhere to regulations specified by NEPA. They noted that resource specialists must evaluate the effects a proposed action will have on soils, and prescriptive treatments are recommended that meet local or regional soil standards and guidelines. Thus, treatment prescriptions typically focus on limiting soil compaction, soil erosion, litter depth, and achieving a coarse woody debris (CWD) target.

Several respondents mentioned that improved methods for posttreatment validation monitoring are needed. In this regard, most respondents noted that posttreatment soil monitoring was conducted. Visual observation of ground cover following prescribed fire and soil compaction following mechanical thinning were the most popular monitoring measures. However, monitoring time periods were relatively short, most lasting less than 1 year.

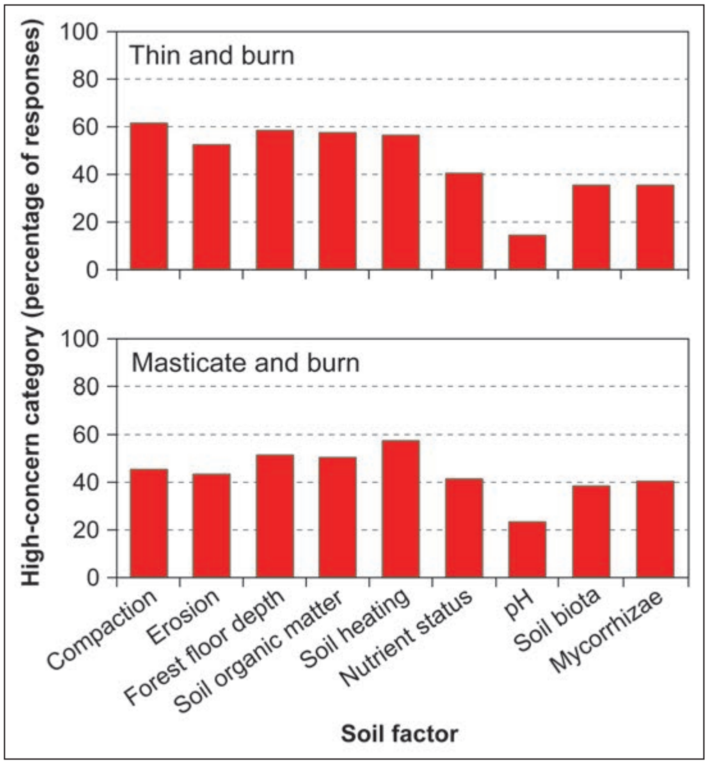

Figure 13—Percentage of survey respondents expressing "high concern" for the effects of thinning and burning on selected soil properties.

Question 3. What unanswered questions and information needs do you have?

To identify additional concerns that were not addressed in the survey questions, we asked, "Do you have any soil concerns specific to your region that have not been discussed?" Below is a summary list of responses by topic. In several cases, the information needs were repeated by respondents across regions.

Fire:

- What is the impact of pile burning on long-term soil productivity?
- Does fire sterilize soils? How much soil heating is acceptable?
- What soil temperatures during burning promote the germination of native plants and discourage the germination of invasive plants?
- What temperatures result in increased water repellency?
- What is the effect of fire on soil fertility and its relationship to invasive plants?
- How do riparian soils respond to fire?
- What is the effect of burning on soil pathogens?

Thinning:

- Does whole tree harvesting affect soil productivity? How much harvest slash should be retained to balance concerns for soil productivity versus those for wildfire hazard?
- How do soils differ in their response to compaction, rutting, or displacement?
- How severe is the loss of soil nutrients and carbon when thinning to presettlement conditions?
- What are the short- and long-term effects of mastication on soil temperature, moisture, and productivity? Can the depth of masticated residues affect soil productivity?
- Is harvesting in riparian areas detrimental to soil productivity?
- How damaging is harvesting on slopes >30 percent?
- What are the best mitigative practices to limit soil damage during harvesting?

General:

- What is the erosion potential of various soils following the removal of forest floor or understory vegetation cover?
- Do fuel reduction treatments detrimentally affect water infiltration, storage, and transmission in soil?
- How much CWD is needed to maintain soil productivity?
- Are soil quality standards for fuel reduction treatments overly restrictive? Improved definitions and thresholds for detrimental soil disturbance are needed, as are improved mitigative measures to protect soil.
- What are the long-term effects of fire exclusion on soil productivity, organic matter content, and erosion?
- What are the best practices in landslide-prone areas?
- How can we best protect organic soils from excessive rutting when harvesting, from rapid oxidation when dry or exposed, or from consumption when burned?

In conducting this survey, we hoped to capture current managerial perceptions of fuel treatments and their potential for causing detrimental soil disturbance. We used this information and the questions put forward as the foundation of this report, and attempt to answer many (but not all) of these issues in the next section.

Ecological Effects and Management Considerations

Here we address the major soil and environmental concerns voiced in the survey results. Treatment impacts on soils are explored and input to help managers weigh the risks and benefits of fuel treatment options is provided. The topics range widely and are separated into two themes for simplicity—prescribed fire and mechanical fuel reduction—acknowledging that some overlap exists between themes. It is not our intent to provide a comprehensive assessment of all soil issues (see Neary et al. (2005) for a general review of fire and soil); instead we offer interpretive synthesis of research findings and also discuss important knowledge gaps associated with the following topics:

Prescribed fire:

- Soil heating
- Soil water repellency
- Soil nitrogen (N)
- Repeat burning
- Pile burning
- Coarse woody debris

Mechanical fuel reduction:

- Whole tree harvesting and nutrient removal
- Soil compaction
- Masticated fuels—new practices, new concerns?
- Masticated fuels—to burn or not to burn
- Thinning and burning–early results from the Fire and Fire Surrogate (FFS) study
- Soils in a changing climate

Prescribed Fire

Prescribed fire is an effective tool for reducing fire hazard and restoring forest health by replenishing available soil nutrients, stimulating plant growth, improving wildlife habitat, and maintaining biological diversity (Reinhardt et al. 2008). Examples include broadcast or underburning (surface fires of low to moderate fireline intensity that leave the overstory essentially intact), postharvest slash burning, and machine and hand pile burning (fig. 14). Dr. Harold Biswell helped reintroduce fire in the 1940s using controlled burning in longleaf pine (*Pinus palustrus* Mill.)

Figure 14—Prescribed fire examples; burning of understory fuels (top left), jackpot burning of slash (top right), pile burning of thinning slash (bottom left), burning of masticated fuels (bottom right).

forests of the Southeastern United States (Biswell 1989). After moving to the west coast, he questioned the policies behind fire suppression and began promoting the use of prescribed fire in the oak and pine forests of northern California (van Wagtendonk 1995). However, his work remained underappreciated, and complete fire exclusion was practiced by land managers until the early 1960s (van Wagtendonk 1995). Prescribed fire became more acceptable with the Wilderness Act of 1964, which identified the need for fire as part of a natural wilderness ecosystem. The necessity for prescribed fire became more apparent in the 1990s with the encroachment of communities into forests resulting in catastrophic Wildland-urban interface fires. Air quality concerns (smoke and haze) and risk of fire escape somewhat constrain the use of prescribed burns at present, especially in highly populated areas. Still, the use of prescribed burning greatly exceeds that of mechanical thinning as a fuel reduction treatment in the Southern United States, whereas total acreage for mechanical thinning surpasses that of prescribed burning on federal lands in the Western United States (see fig. 7).

Soil Heating

Key points:	
Ecological effects	**Management considerations**
• Mineral soil is a poor conductor of heat. Soil damage (if any) is usually limited to the top few centimeters for all but high-severity burns.	• Encourage low- to moderate-severity burning and incomplete forest floor consumption to avoid mineral soil damage.
• Soil moisture is a critical factor controlling heat transfer.	• Burn when soils are moist (>20 percent by volume) when possible to limit heat penetration.
• Comparatively low soil heating results in fine root mortality.	• Generally this is not a problem for low- to moderate-severity burning as soil temperatures are not excessive • Burn when forest floor layers are moist (>65 percent) when possible or avoid burning if heavy root prolifera-tion is detected in the duff layer. • Alternatively, mixed-severity burning with attending root mortality may be appropriate to reestablish forest heterogeneity.
• Smoldering of thick duff layers greatly extends heat duration and may kill roots and affect tree vigor. • Soil micro-organisms are killed across a wide range of temperatures. However, even severe burning does not sterilize soil	• Raking of litter and duff from the base of old-growth trees **may** protect against fire-related damage. • Expect short-term changes in micro-bial function and diversity after severe burning. Long-term effects, although possible, are difficult to prove and are not a driver of manage-ment decisions.
• Soil carbon and organic matter losses from mineral soil are nominal unless burning is sever. • Ephemeral increases in soil pH, calcium, magnesium, and potassium result when fuel consumption is high.	• Monitoring of soil chemical proper-ties may be warranted if extensive areas of severe burning are antici-pated or for nutrient-poor sites.

Why soil heating is of concern—
All methods of prescribed burning generate soil heat which—under certain circumstances—can alter soil properties and functions. Changes in soil fertility, organic matter content, water infiltration, soil mineralogy, and nutrient availability are among the many potential responses to burning that may either benefit or degrade soil quality. And although only 8 to 10 percent of the heat produced during a fire is radiated downward to the soil (Hungerford 1989), soil temperatures can still surpass 250 °C during broadcast burning (Haase and Sackett 1998) and approach 500 °C during pile burning (Roberts 1965). Indirectly, fire affects short-term microbial processes such as nutrient immobilization, mineralization, and nitrification by raising soil temperature and reducing soil moisture. Hillslope erosion and leaching losses of fire-released soil nutrients may further result during postfire rain events. Wind erosion and its ability to severely degrade soil quality is another indirect effect, although it is typically associated with wildfire and not fuel reduction burning (Bormann et al. 2008). Thus, there are many incentives for understanding and developing an awareness of soil heating effects (box 1).

Threshold temperatures offer insight into the potential problems associated with soil heating. These temperatures have been established in controlled experiments for many soil properties and functions (table 2), and are best thought of as "ballpark" values as they have not been exhaustively tested across a variety of soils. Nevertheless, the values show that several key biological properties such as micro-organism, root, and seed survival are vulnerable at reasonably low soil temperatures, whereas changes in soil physical and chemical properties require higher temperatures that are more typical of severe burning. In fact, the high temperature threshold for the volatilization of many nutrients (775 to 1960 °C) is rarely, if ever, reached in mineral soil for even the most severe of burns.

Note that some of the threshold values encompass a wide temperature range, making definitive statements about fire effects and soils difficult. For example, the lethal temperature for bacteria ranges tremendously owing to physiological differences among bacterial species (e.g., differences in cell wall structures and spore-forming abilities confer a variety of levels of heat tolerance). A wide temperature range is also reported for soil carbon (C) and N volatilization, although the majority of loss is thought to occur near the upper limit of their temperature ranges (Badia and Marti 2003, Guerrero et al. 2005). Additionally, the threshold values can vary considerably depending on soil water content at the time of burning as discussed below.

Box 1

Does Severe Burning Sterilize Soil?

Dec. 2003

Brett Goforth

Dec. 2005

Brett Goforth

Brett Goforth

Soils often take on a moonscape-like, deathly appearance after severe burning. A reddish-orange coloring, symptomatic of extreme heating and oxidized organic matter, is not uncommon following burning of large slash piles (landing piles) or where large downed wood has been consumed slowly during wildfire. The damage is obvious to trained and untrained eyes alike. But are these soils sterile, without life? And if so, will life return? The answer to these questions lies in how we define the term sterile. To those meaning devoid of plant life, the answers are yes. Severe burning easily produces temperatures in surficial soil that kills the entire seedbank (>120 °C). Plant life will take several years to recolonize these soils, eventually aided by wind deposition of seeds from adjacent vegetation. Still, sterilization is relatively brief given this definition. A different answer results if we use the definition of "sterile" provided by the dictionary ("free from living organisms"). Countless studies have shown the dogged ability of some soil microbial species to survive severe burning (see Hebel et al. 2009), and suggest that the survivors may even flourish in the postfire environment. In fact, to kill all soil micro-organisms takes exacting conditions that are met typically by pressure cookers and laboratory autoclaves (121 °C with steam or 650 to 1,000 °C with dry heat), but that are not attained in the soil profile even during the most severe of fires. Further, the recolonization process by surviving micro-organisms and by those organisms disseminated on the soil surface by wind or water begins almost immediately following fire.

Does severe burning sterilize soil? No. Life persists. But this is a technical answer and it does not speak to the multitude of soil changes that may be caused by severe burning. Loss of organic material, increased soil erosion, changes in surface physical and chemical properties, and alteration of long-term soil productivity are at risk. In the case of soil micro-organisms, severe burning does not eliminate their presence, but it does dictate which species remain and which species are consumed. Not sterile for sure, but of such concern that efforts to limit the damaging effects of severe burning are a long-standing objective of forest stewardship.

Table 2–Examples of threshold temperatures for soil physical, chemical, and biological properties[a]

Soil property	Threshold (°C)	Source
	Low	
Microbial death:		
Bacteria	50 to 400	Hungerford et al. 1991
Nitrifying bacteria	75 to 140	DeBano et al. 1977
Vesicular arbuscular mycorrhizae	94	Klopatek et al. 1988
Seed mortality	50 to150	Beadle 1940
Fine-root mortality	40 to 70	Zeleznik and Dickman 2004
Soil water loss	60 to 100	Hungerford 1991
	Moderate	
Particle aggregation (e.g., clay conversion to sand)	200 to 500	Terefe et al. 2008
Carbon and organic matter oxidation	200 to 500	Raison et al. 1985
Nitrogen volatization	300 to 500	Hungerford et al. 1991
Soil structure and aggregate stability loss	300	
Amino acid loss	350	Hungerford et al. 1991
Water repellency	>270 to 300	DeBano and Krammes 1966
Ectomycorrhizal fungi death	100 to 155	Dunn et al. 1985
	High	
Nutrient volatization:		
Potassium	775	Raison et al. 1985
Phosphorus	775	Raison et al. 1985
Calcium	1240 to 1485	Raison et al. 1985
Manganese	1960	Raison et al. 1985
Magnesium	1107	DeBano 1991
Sulfur	375 to 900	Tiedemann 1987
Clay structure transformed or destroyed	>550	Certini 2005, Douglas 1986, Ketterings et al. 2000

[a] These values are not absolute and can vary depending on to the heterogeneity of soils in natural systems.

Applying the principles of thermal conductivity and heat capacity—
Soil heat transfer during burning is a complex process involving a long list of soil physical properties (e.g., moisture, texture, porosity, density, particle contact), fuel characteristics (e.g., mass, moisture, surface area, structural arrangement), and fire attributes (e.g., rate of spread, fire weather). Scientific advances in predicting soil heating have been considerable, particularly through the development and validation of heat transfer models (e.g., Aston and Gill 1976, Campbell et al. 1995, Enninful and Torvi 2008, Massman and Frank 2004, Preisler et al. 2000, Steward et al. 1990). Most theoretically based models (e.g., FOFEM, http://frames.nbii.gov/metadata/tools/FOFEM_5.7.html) rely heavily on mathematical derivations of soil thermal conductivity (ability to transfer heat) and heat capacity (amount of energy required to increase soil temperature). How these physical principles of heat transfer are influenced in practice by soil depth, moisture, and texture are discussed below.

***Soil depth*—**Soil is not a good conductor of heat. Consequently, both peak temperature and heat duration decline precipitously with soil depth during burning. Figure 15A illustrates this, as a stair-step decline in the heat pulse is shown with increasing depth from the soil surface. Indeed, most studies show that damaging soil heat, even during severe burning, is usually limited to a fairly thin layer of surface soil (Busse et al. 2005, Massman and Frank 2004, Monsanto and Agee 2008). The reason for this lies in the extensive array of pores found throughout the soil matrix, which trap heated air and reduce heat penetration, analogous to home insulating materials. Heat transfer is thus relatively low in soil as the thermal conductivity of porous materials (like soil) is far lower than it is for solid materials. Smoldering fires in thick duff layers or large-diameter wood are important exceptions to this rule as trapped heat can lead to considerable soil heat penetration (Haase and Sackett 1998, Hebel et al. 2009, Hood 2010).

***Soil water*—**Soil water is considered the most important factor controlling heat transfer in soil (Frandsen and Ryan 1986, Hartford and Frandsen 1992). This is demonstrated in figure 15, where heat penetration is dramatically lower in moist soil than dry soil for a comparable fuel load and fuel consumption. Similar observations led Busse et al. (2010) to recommend burning when soil volumetric moisture content was 20 percent or greater as a means to limit soil heating beneath large fuel loads such as masticated fuels or slash piles. The basic principle is that soil water sharply increases the heat capacity of soil, resulting in a high-energy requirement to evaporate water prior to any substantial increase in soil temperature. Thus, although heat travels faster in moist soil than dry soil based on the principles of thermal conductivity (Jury et al. 1991), its movement is restricted by the energy-quenching effect of water.

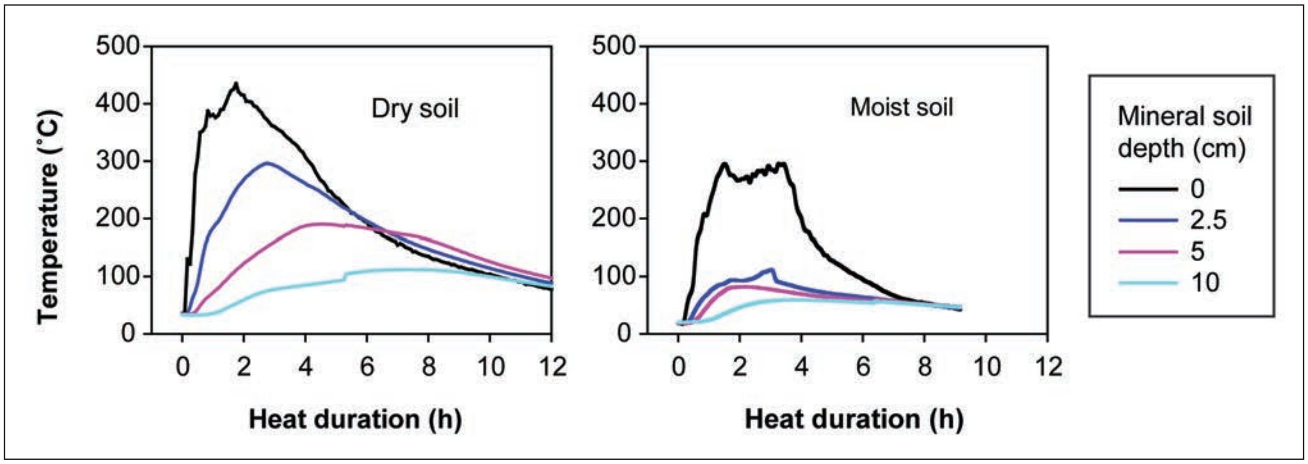

Figure 15—Heat pulse in dry versus moist soil during burning of masticated fuels. Soil moisture contents are 5 percent for dry soil and 27 percent for moist soil. Only soil moisture was varied, as experimental conditions included the same fuel loading (169 Mg ha^{-1}), fuel consumption, and soil type. Adapted from Busse et al. 2005.

***Soil texture*—**Coarse-textured soils (sands) have greater thermal conductivity than finer textured soils because of their low porosity and high surface contact between particles (Aston and Gill 1976). Consequently, temperatures and heat penetration are theoretically greater in sandy soils than in loam or clay soils of similar rock content during burning. However, there is little experimental evidence to support this theory. In fact, Busse et al. (2010) found that heat transfer was similar across a range of soil textures during burning. Soil moisture far outweighed the role of soil texture as a determinate of heat transfer in their study.

Organic matter content and physical compaction are confounding factors when assessing the role of texture in soil heat transfer. For example, the heat pulse into soils rich in organic matter from surface fires can be substantially reduced because of the high water-holding capacity and low thermal conductivity of organic matter. Organic soils (e.g., peat soils in the Southeast or Alaska) will burn intensively, however, if ignited when dry. Additionally, compaction can dramatically increase maximum soil temperatures and heat duration during burning (Busse et al. 2010). Compaction increases the thermal contact between soil particles and reduces the air-filled pore space, resulting in a gain in soil thermal conductivity.

How hot does it really get?—

This is not an easy question to answer because variation is the rule of thumb. For example, Haase and Sackett (1998) showed that soil can surpass lethal temperatures for more than 20 hours when thick forest floor layers are completely consumed by prescribed burning. In comparison, Trammell et al. (2004) found that temperatures never reached 50 °C in the surface mineral soil even when temperatures in the litter layer exceeded 500 °C in hardwood forests of central Kentucky. Shea (1993) found a similar lack of heat transfer during fuel reduction burning in young ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) stands as shown in figure 16.

Many researchers have measured soil heat profiles during prescribed burning; their findings are summarized in table 3. Although the findings are far from complete, especially regarding the variety of fuel loadings and fuel moisture contents typically encountered when burning, several generalizations can be made:

- Forest underburning produces minimal soil heating except in areas where duff layers are completely consumed. Therefore, detrimental heat damage should not be expected in most cases.
- Small slash piles result in moderate soil heating in the surface 5 to 10 cm (2 to 4 in). However, the range in reported temperatures does not suggest any major changes in soil properties with the exception of potential root, seed bank, and microbial mortality. See section on pile burning for more details.
- Large slash piles and, in particular, those containing a high proportion of large-diameter wood result in high soil temperatures and long heat durations. Detrimental heating effects on soil properties should be expected in the top 10 cm (4 in) or more.
- Grassland fires produce nominal soil heating. The dominance of fine fuels in these systems ensures that burn duration time is generally low and soil temperatures are minimal.

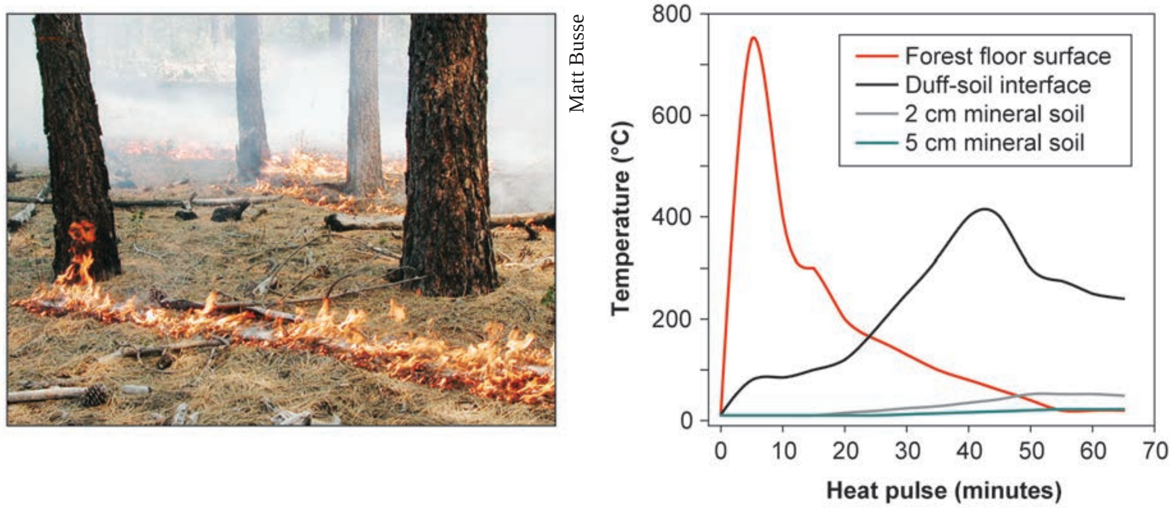

Figure 16—Nominal heat penetration in soil during spring prescribed underburning in ponderosa pine forests. The burns were low to moderate intensity (0.6- to 1.0-m flame lengths) with fuel loads ranging from 88 to 113 Mg ha^{-1}. Fuel consumption ranged from 40 to 60 percent on a mass basis. The coarse-textured pumice soils were moist (30 to 40 percent gravimetric moisture) at the time of burning. Data adapted from Shea (1993).

Table 3—Maximum soil temperatures recorded during prescribed burning in different ecosystem and fire types

Ecosystem and fire type	Fuel consumption	Soil depth	Maximum soil temperature	References
		Centimeters	*°C*	
Forest underburn	Low to moderate (25 to 75% percent duff loss)	2 5	25 to 40 20 to 30	Bradstock and Auld 1995, Penman and Towerton 2008, Shea 1993, Trammell et al. 2004, Vines 1968
	Complete	2 4 to 5	280 113 to 200	Haase and Sackett 1998, Neary et al. 2005
Forest slash pile:				
Hand pile	Complete	2 to 3 10	80 to 150 35 to 80	Beadle 1940, Floyd 1966, Meyer 2009
Tractor pile	Complete	2 5 10	235 to 400 150 to 350 225	Massman and Frank 2004, Shea 1993
Wood pile	Complete	2.5 5 10	500 314 to 340 190 to 230	Roberts 1965
Grassland	Complete	2 4	36 27	Scotter 1970
Chaparral	Complete	2.5 5	88 to 199 35 to 75	DeBano et al. 1979

We offer the following short synopses on root mortality, ectomycorrhizal fungi and soil organic matter (SOM) loss, and changes in soil pH and cations as examples of soil properties affected by low-, moderate-, and high-threshold temperatures, respectively. Fire-induced changes in soil N are discussed in more detail later in this chapter.

Root mortality—Prolonged soil heating kills fine roots and can injure large structural roots (Guo et al. 2004, Swezy and Agee 1991, Zelznik and Dickman 2004). By convention, most studies consider 60 °C as the lethal soil temperature for root mortality (Preisler et al 2000, Varner et al. 2009) even though root destruction has been measured at temperatures as low as 52.5 °C when the duration of heating is sufficiently long (Zelznik et al. 2004).

Based on the range of soil temperatures observed in literature (table 2), temperature thresholds for root mortality are generally not exceeded in surface mineral soil layers during low- to moderate-severity fuel-reduction burning. In comparison, excessive root damage is an ongoing concern in old-growth forests that have developed thick duff mounds following decades of fire exclusion (fig. 17). Swezy and Agee (1991) were among the first to show that low-severity prescribed burning

Carl Skinner

Figure 17—Deep duff mounds can result in lethal heating when burned.

resulted in elevated mortality of old-growth ponderosa pine, and they hypothesized that root damage from the heat generated by smoldering duff was a contributing factor. In support, both Preisler et al. (2000) and Varner et al. (2009) showed that soil temperatures can exceed the root lethal threshold to considerable depths in the mineral soil beneath thick duff layers. Varner et al. (2009) further reported a link between the duration of lethal soil heating and carbohydrate drain in coarse roots, suggesting that prolonged smoldering of the duff exacerbates postfire tree stress. These findings support a model of postfire stress where injured trees use available carbohydrate storage to replenish killed or injured fine roots, depleting carbohydrate reserves and thereby compromising the tree. Of course, many other factors in addition to soil heating, particularly bark beetle attack, are also known to contribute to postfire tree stress and mortality (Fettig et al. 2006, Kolb et al. 2007, Perrakis and Agee 2006).

Raking duff mounds away from the base of large trees is a recommended management practice for lowering fire intensity, lethal soil heating, and, consequently, root and tree mortality (Hood 2010, Kolb et al. 2007). However, supporting evidence for the effectiveness of raking has been inconclusive to date (Hood 2010, Knapp et al. 2011, Laudenslayer et al. 2008, Swezy and Agee 1991). These findings led Kolb et al. (2007) to suggest that “the effects of raking treatment may be site specific” and acknowledge that tree mortality is a function of numerous site factors (crown scorch, tree species and age, postfire insects, drought), not just soil heating effects.

In discussing options for limiting soil heating when burning duff mounds, Hood (2010) emphasized that duff moisture content is the critical determinate of duff ignition and smoldering duration. She suggested that a moisture content greater than 65 to 85 percent is sufficient to restrict duff smoldering and thus limit excessive soil heating and root damage. Her recommendations for managers included:

- For large-scale operations where thick duff layers (>12 cm) are prevalent, burn during the dormant season when duff moisture content is high, or within 2 days after a significant rain event in the Southeast.
- For protecting individual trees in small stands, burn when the lower duff layer is moist and either (1) lightly rake or blow some, but not all, of the duff away from the tree base, (2) burn within snow wells at the base of trees at least 1 year prior to broadcast burning, or (3) mop-up with water during burning.
- For protecting individual trees with visible fire scars, rake all litter and duff from the base and apply snow, if available, to the scar before burning.

Perrakis et al. (2011) further suggested that managers use a gradual approach for restoring forests with excessive duff accumulation by first attempting to improve the vigor of large trees prior to an initial entry of extremely low-intensity burning. They found that mortality was greatest for trees showing poor crown structure and minimal radial growth in recent years, and suggested that managers consider thinning nearby trees to increase preburn tree vigor.

These recommendations offer a gentle, conservative approach for reintroducing fire into old-growth systems when the survival of high-value trees is paramount. In other cases, mixed-severity burning that kills roots and low- and mid-canopy trees may be appropriate to restore natural variability in forest conditions. For example, Collins et al. (2011) found that moderate-severity burning in old-growth, mixed-conifer stands in Yosemite National Park conserved large trees and produced variable stand structures that resembled historical conditions. In comparison, they found that low-severity burning was mostly ineffective at restoring stand densities or structural heterogeneity, key ecosystem components for maintaining resilience against modern stressors such as disease, wildfire, and climate change (Stephens et al. 2010). Using prescribed fire to reestablish forest heterogeneity should also lead to patchy (mosaic) forest floor layers, ranging from bare soil to unburned areas, which may be most appropriate for meeting desired fuel conditions while safeguarding soil quality (Knapp and Keeley 2006).

Ectomycorrhizal fungi—Ectomycorrhizal (ECM) fungi are found naturally in soils where they colonize roots and form a mutualistic relationship with conifers and some shrub species. Plants support ECM by providing carbon (C) (20 percent of a plant's photosynthate may be allocated to support ECM fungi). In return, ECM fungi provide plants with nutrients, water, and protection from plant pathogens

(Smith and Read 1997). Fungal hyphae effectively increase the surface area of roots, enabling plants to better thrive in water and nutrient-limited conditions. In addition, ECM fungi release exudates that help bind soil particles into aggregates, thereby improving soil structure, aeration, and water infiltration (Miller and Jastrow 2000).

Because ECM fungi require living roots to survive, any fuel reduction treatment that alters forest vegetation may have profound effects on ECM. Soil compaction from heavy machinery can harm the mycelium (a matlike mass of fungal hyphae), reduce mushroom productivity (Wiensczyk et al. 2002), and reduce ECM root tip abundance and diversity (Amaranthus et al. 1996). Prescribed broadcast burning can reduce fungi populations indirectly by increasing soil pH, inducing a nutrient flush, and reducing litter and duff levels, or directly when the heat pulse is excessive (Kipfer et al. 2010, Stendell et al. 1999). Repeated burning has also been shown to significantly reduce ECM biomass (Hart et al. 2005a) and species diversity (Tuininga and Dighton 2004).

Season of burn is an additional concern for ECM survival and function in many Western forests (Cairney and Bastias 2007). Fall burning when soils are dry can result in higher burn severity and ECM damage compared to moist spring burning. For example, Trappe et al. (2009) found that only fall burning resulted in changes in ECM fungal productivity and fruiting patterns. Smith et al. (2004) reported that fall prescribed burning in dry ponderosa pine stands significantly reduced duff depth, live root biomass, and ECM species richness compared with low-severity spring underburning. Thus spring burning when soil and duff moisture is comparatively high appears to be a good alternative for maintaining ECM populations. Additional forest management strategies to maintain ECM fungi diversity and health are listed in table 4.

Soil organic matter—Surface organic matter is oxidized by fire as temperatures exceed 200 °C (Johnson et al. 2004), releasing carbon dioxide (CO_2) and associated gases, charcoal, and ash. Although this a seemingly straightforward combustion process, Tinoco et al. (2006) stated that the effects of fire on surface and mineral soil organic matter “is one of the most complex subjects in systematic research on environmental impacts.” In practice, mineral soil temperatures rarely exceed 200 °C during low- to moderate-severity burning, let alone reach higher temperatures (400 to 500 °C) where the majority of soil organic matter is lost (Guerrero et al. 2005). Indeed, Rau et al. (2009) actually found an increase in soil C immediately following low-severity burning that was likely due to the incorporation of ash, charcoal and partially burned organic matter into the mineral soil (box 2).

Table 4—Forest management strategies to maintain ectomycorrhizal (ECM) fungi diversity and health[a]

Strategy	Management practice
Provide a source of ECM fungal inoculum	• Retain live trees • Retain areas of old-growth forest across the landscape
Provide differing habitats or microsites to promote fungal community diversity	• Avoid uniform prescribed burns of high severity • Minimize disturbance to forest floor during treatment • Retain standing and downed coarse woody debris
Maintain a diverse and robust ECM fungal population	• Avoid restoration projects that involve grass seeding • Encourage regeneration of a mixture of tree species
Conserving ECM fruiting body production	• Minimize soil compaction from heavy machinery and trampling • Minimize disturbance to forest floor • Retain standing and downed coarse woody material

[a] Source: adapted from Wiensczyk et al. 2002.

Box 2

Soil Organic Matter (SOM) as a Key Indicator of Soil Health

- Provides a reservoir of nutrients that is gradually released to the soil
- The equivalent of 1 percent SOM can release 9 to 14 kg (20 to 30 lb) of available nitrogen per year (Vigil et al. 2002)
- Regulates biotic activity, providing a carbon and energy source for soil microbes
- Absorbs and holds up to 90 percent of its weight in water, with most of the water being available to plants
- Contributes to the formation of stable soil aggregates
- Improves soil permeability and aeration
- Contributes up to 20 to 80 percent of the cation exchange capacity of a soil

Most studies show relatively few short-term changes in total mineral soil C owing to burning. For example, Johnson and Curtis (2001) found no overall effect of prescribed fire or wildfire on soil C in a meta-analysis of 48 observations from 13 publications, with relatively few outliers showing significant soil C loss (e.g., Grier 1975). Surprisingly, they found that prescribed fire alone resulted in a small decline in soil C (<6 percent), whereas wildfire generally resulted in higher soil C content. They attributed the increase in soil C after wildfire to the incorporation of unburned residues and deposition of charcoal (box 3). In addition, they noted

Box 3

Black Carbon: Do Charcoal Additions Improve Soil Quality?

Carol Shestak

Charcoal accumulation in the surface 5 cm of soil after pile burning.

Carol Shestak

Comparison of postburn (left) and preburn soil (right) from beneath a pile burn.

The dark, rich terra preta soils in the Amazon Basin offer an unmistakable example of how continual additions of fire-derived charcoal can greatly improve soil productivity. Along with providing a reservoir of essential plant nutrients, the stable charcoal in these soils contains a high concentration of carbon (C) that improves soil water-holding capacity, nutrient retention, detoxification of plant- and microbial-inhibiting compounds, and soil-warming capability. These soils developed in localized areas centuries ago where charcoal from cooking fires was repeatedly mixed with soil. Today, there is growing interest in the role of charcoal not only in terra preta soils, but in soils exposed to forest and grassland burning (wildfire or prescribed fire) and in soils receiving soil biochar amendments from bioenergy projects. Recent review articles point to the importance of soil charcoal in improving soil quality and stabilizing the global C cycle (DeLuca and Aplet 2008, Forbes et al. 2006, Preston and Schmidt 2006); however, they stop short of confirming these claims because of an acknowledged scarcity of supporting scientific data. So we ask the question, where along the perceived gradient from "no impact" to "critical soil constituent" is charcoal found? Here is an abbreviated summary of the current state-of-the-knowledge of soil charcoal:

- Charcoal is a common biproduct of fire that is generated when the combustion process becomes oxygen-limited.
- The aromatic ring structure of charcoal confers high internal porosity and surface area, resulting in a high capacity for water, nutrient, and chemical sorption (Preston and Schmidt 2006).
- Charcoal is relatively inert. Estimates of its longevity range from 650 to >10,000 years depending on exposure to oxygen and possible consumption in succeeding fires (Ohlson et al. 2009).
- The production of charcoal differs considerably depending on fire regime, fuel type, loading, moisture, temperature, and fire intensity (Ohlson et al. 2009). However, useful information on prescription conditions for fuel reduction burning that maximize charcoal production is lacking.

Box 3 (continued)

- Approximately 0.7 to 3 percent of burning organic matter is converted to charcoal, although the degree of spatial variability in soil charcoal content is high within a fire perimeter (Czimczik et al. 2003, Ohlson et al. 2009). This suggests that charcoal may act primarily at localized "hotspots" to improve soil quality.
- Most estimates of soil charcoal content come from wildfire sites. We found no quantitative information on soil charcoal produced by fuel reduction treatments. Only DeLuca and Aplet (2008) provided a rough estimate on soil charcoal from a hypothetical 200-year prescribed fire cycle (a modest addition of 4 Mg ha^{-1}).

Charcoal offers numerous benefits to soil chemical and physical properties. However, uncertainties about prescription conditions for optimal production and the high level of spatial variability following burning underscore the fact that insufficient knowledge exists to assess the practical importance of soil charcoal derived from fuel reduction practices. Or as pointed out in an indecisive statement by Zachrisson et al. (1996), "the levels of charcoal in…. forest soils are sufficient for them to have a possible ecological effect."

an average 10 percent increase in soil C for those studies that collected data for a minimum of 10 years after fire. But unlike short-term changes in soil C, definitive evidence of long-term soil C changes after fire is clouded as it depends on many site-specific variables (fire frequency, fire severity, vegetation structure and composition, postfire presence of N-fixing vegetation, charcoal production, damage from previous entries).

Soil pH and cation exchange capacity—Soil pH is a critical attribute that affects nutrient availability and the toxicity of elements like aluminum and iron. Generally, soils with neutral or near-neutral pH (about 5.5 to 7.7) are considered the most chemically balanced to support the extensive variety of soil processes critical to all ecosystems (Fisher and Binkley 2000). During combustion, base cations are mineralized from surface fuels leaving behind a nutrient-rich ash layer. The first rains leach the cation-laden ash into the soil, providing a flush of nutrients for new growth. These cations increase soil pH by displacing the H and Al ions adsorbed on the negative charges of the soil colloids. Increased soil pH after fire is a well-documented phenomenon whose ecological effects depend, in part, on preburn conditions. For example, a postfire increase in pH may be very beneficial for a low pH soil because of is positive effect on nutrient availability, whereas fire-induced changes may be biologically nominal in neutral pH soils (Franklin et al. 2003).

Two factors help determine the extent of soil pH change: (1) amount of fuel consumed, and (2) soil buffering capacity, or a soil's ability to resist change as a function of its total cation content, cation exchange capacity, and reserve acidity. In effect, greater fuel consumption leads to greater release of cations in ash and thus greater pH change, whereas highly buffered soils are capable of absorbing postfire increases in available cations without concomitant changes in pH. Typically, fertile clay soils have higher buffering capacity than sandy soils and are more resistant to pH change. The examples presented in table 5 show that low-severity burning (e.g., broadcast burning) results in only minor gains in soil pH, whereas high-severity pile burns result in considerably larger increases. In a meta-analysis of fire and fire

Table 5—Examples of pH changes in the top 0 to 5 cm of soil following broadcast underburning (BB) or pile burning (PB)

Ecosystem and study location	Type of burn	Burn severity	Soil pH: Pre-burn	Soil pH: Post-burn	Potential ecological effect
Ponderosa pine, Arizona[a]	BB	Low	5.27	5.49	Minor
Ponderosa pine, Montana[b]	BB	Low	5.00	5.30	Minor
Ponderosa pine/white fir, Oregon[c]	BB	Low	5.78	5.93	Minor
Oak, Kentucky/Tennessee[d]	BB	Low	4.55	4.81	Minor
Oak/pine woodlands, Alabama[e]	BB	Low	4.80	4.70	Minor
Oak hardwoods, Massachusetts[f]	BB	Low	4.20	4.43	Minor
Loblolly/longleaf pine, South Carolina[g]	BB	Low	4.60	4.30	Minor
Mixed conifer, California[h]	BB	High	5.19	5.83	Moderate
British Columbia Canada[i]	PB	High	3.48	5.22	Large
Ponderosa pine, Arizona[j]	PB	High	5.9	7.0	Moderate

[a] Grady and Hart (2006).
[b] Gundale et al. (2005).
[c] Trappe et al. (2009).
[d] Franklin et al. (2003).
[e] Nobles et al. (2009).
[f] Neill et al. (2007).
[g] Binkley et al. (1992).
[h] Moghaddas and Stephens (2007).
[i] Arocena and Opio (2003).
[j] Korb et al. (2004).

surrogate treatments, Boerner et al. (2009) noted that significant increases in soil pH occurred where fire severity was greatest or where pretreatment base status was the lowest. They also affirmed the commonly held knowledge that changes in soil pH are ephemeral (Neary et al. 1999, Sherman and Brye 2009), as they remained high during the first year following fire but not in subsequent years.

Like soil pH, postfire increases in plant-available cations are often short-lived. For example, Franklin et al. (2003) noted that significant postfire increases in soil nutrient concentrations had returned to preburn levels within 1 to 3 years. Soil nutrient reserves may even be depleted if fire substantially reduces the cation exchange capacity (CEC), which is a measure of a soil's ability to adsorb and release plant-essential cations. Cation exchange capacity is closely tied to soil texture and SOM content: both clay particles and SOM have high surface areas and negatively charged sites that confer high CEC. Decreases in CEC are often noted after severe burning when high temperatures consume SOM and eliminate cation exchange sites (St. John and Rundel 1976, Seymour and Tecle 2005). Nobles et al. (2009) observed lower retention of soil potassium (K) and sodium (Na) after fire due to leaching losses, whereas less mobile cations like calcium (Ca) remained unchanged in the surface horizon. Boerner et al. (2009) noted that effects of low- to moderate-severity fire on cation availability were modest and were limited mainly to increased Ca concentrations during the first year following fire at several sites across the United States. Thus, similar to the effects of fire on soil pH and organic matter, changes in plant-available cations will likely be moderate, at most, except with high-severity burning.

Soil Water Repellency

Key points:

Ecological effects	Management considerations
• Water repellency results from the presence of naturally formed hydrophobic coatings on soil particles. Hydrophobic compounds are volatilized during fire and move downward through soil along a temperature gradient before recondensing.	• Some degree of soil hydrophobicity soils is likely unavoidable following prescribed fire. Burning thick duff, particularly when soils are dry, will increase repellant conditions. Managing for a diversity of fuel consumption (allowing some unburned, some complete consumption) may create an acceptable mosaic of repellant conditions.
• Repellency intensifies at soil temperatures from 175 to 200 °C. These temperatures are readily reached by burning heavy fuels when soil is dry, but infrequently attained when soil is moist. • Repellency may be destroyed if soil temperatures exceed 270 °C, such as under burn piles. Definitive evidence is inconclusive, however.	• Expand the use of mosaic burning patterns and uneven forest floor consumption to limit the spatial extent of hydrophobicity.
• Hydrophobic soils tend to remain water repellant until moisture content increases above 10 to 13 percent. Postfire erosion may be high during rain events when hydrophobic soils are dry.	• Vegetation or forest floor cover is crucial to limit erosional losses in the first year after fire. Limit extent of complete forest floor consumption.

Resistance to wetting, or soil water repellency, results from hydrophobic coatings on soil particles, which reduce the affinity between soil and water and can lead to localized surface erosion (Doerr and Thomas 2000, Hallet 2007). These water-repellent compounds are produced from decomposing plants, root exudates, some fungal species, surface waxes from plant leaves, and decomposing duff and litter (Fogel and Hunt 1979, Hallett 2007). Evergreen plants (e.g., eucalyptus, pines, and Mediterranean shrublands) are most commonly associated with water repellency, particularly trees and shrubs with high foliar resin, wax, or aromatic oil contents (Doerr et al. 2000). The magnitude and persistence of repellency is further influenced by soil textural differences, as sandy soils usually exhibit higher repellency than clayey soils because their small total surface area is more easily coated by hydrophobic substances than are finer sized particles with high total surface area (Doerr et al. 2006, Woche et al. 2005).

Prescribed burning of vegetation and litter can substantially alter soil water repellency given the right heating conditions. Hydrophobic compounds, when volatilized by fire, move downward through soil along a temperature gradient and condense on cooler soil particles (DeBano 1981), usually at a soil depth between 0 to 8 cm (Huffman et al. 2001). Water repellency generally intensifies at soil temperatures between 175 to 200 °C owing to a strengthening of bonding between water-repellent substances and soil particles (Doerr et al. 2000, Savage 1974). Above 270 to 300 °C, hydrophobic substances are thought to be irreversibly destroyed (DeBano et al. 1976). However, recent observations (Hubbert, unpublished) show repellency occurring below pile burns at soil temperatures well above 300 °C.

Most prescribed fires produce a mosaic of non-, low, moderate, and high repellency across the landscape (fig. 18). The mosaic pattern is a function of nonuniformity of fire temperature and duration, forest floor consumption, vegetation type, and soil moisture and texture (Hubbert et al. 2006). Therefore, one should not expect the spatial distribution of soil water repellency to be uniform across a broad landscape, or even at smaller scales in steep watersheds (Robichaud and Miller 1999).

Water repellent soils typically alternate between repellent and nonrepellent states in response to postfire rainfall patterns (Dekker et al. 1998, Doerr and Thomas 2000, Shakesby et al. 2000). In most cases, soil water repellency increases in dry soil and either decreases or vanishes following periods of precipitation (fig. 19) (Crockford et al. 1991, Ritsema and Dekker 1994). In fact, several studies show that repellency is greatly reduced when soil moisture content exceeds 10 to 13 percent by volume (Dekker et al. 2001, Hubbert and Oriol 2005, MacDonald and Huffman 2004).

Because soil repellency is greatly increased when soils are dry, we would expect pronounced overland flow and subsequent erosion when storm events follow prolonged dry periods (Doerr et al. 2000). However, Imeson et al. (1992) noted that although water-repellent soils can contribute to high overland flow rates locally, any effects at the hillslope or catchment scale are dwarfed by high spatial variability in infiltration rates. In fact, most studies have only inferred a causal link between water repellency and erosion, and have failed to isolate the erosional impacts of water repellency from the confounding effects of losses in vegetation cover, litter cover, or soil aggregate stability (Doerr et al. 2000, Hubbert et al. 2006, Scott 1993).

The degree of postfire reduction in water infiltration is dependent on the spatial distribution of hydrophobicity on the landscape (Shakesby et al. 2000). Local areas of repellency will reduce infiltration, but may pond water and enhance preferential flow down cracks and root channels. In this case, preferential flow can move both

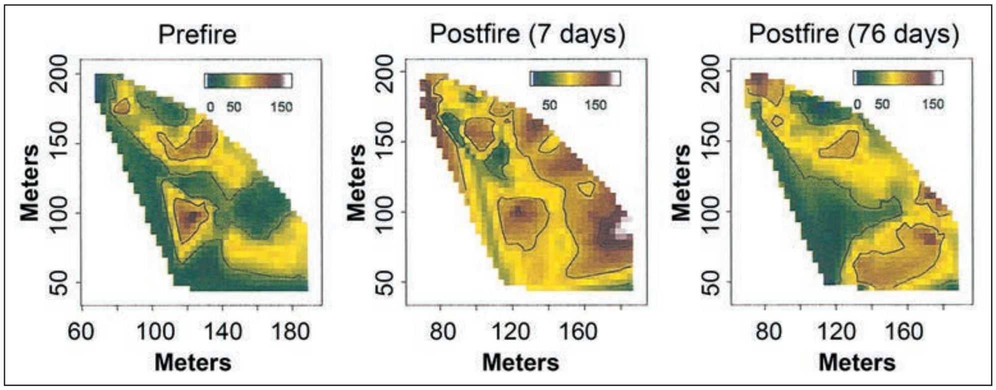

Figure 18—Effects of fire on spatial and temporal patterns of water repellency at the soil surface. Repellency increases from green to red in the legend. Green and light green represent no repellency and very low repellency. Yellow represents low repellency. Brown represents moderate to high repellency. Red represents very high repellency. Adapted from Hubbert et al. (2008).

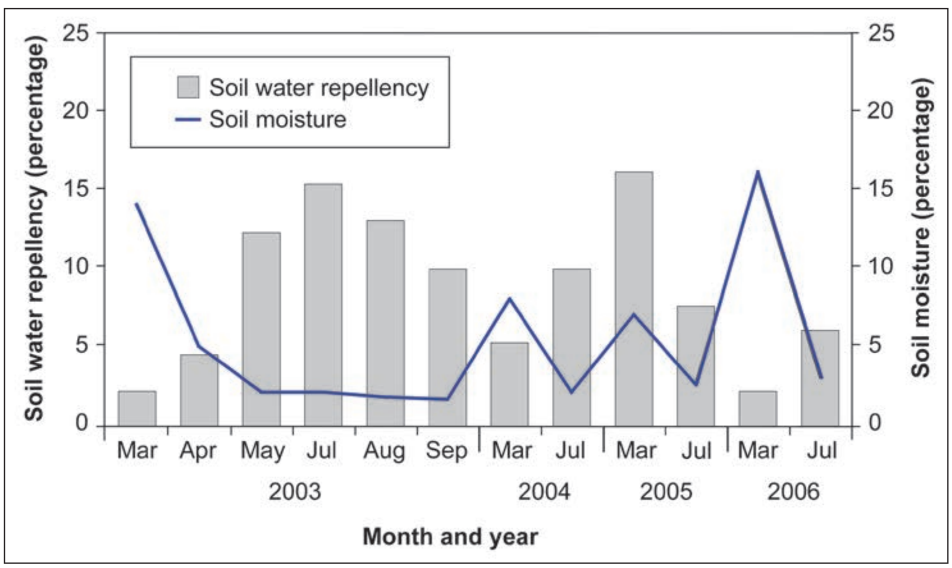

Figure 19—Soil water repellency is generally greatest when soils are dry. Data were collected following the September 2002 Williams Fire in the San Gabriel Mountains, California. Adapted from Hubbert and Oriol 2005 and Hubbert et al. 2012.

vertically and laterally through the subsurface, removing water that would otherwise contribute to overland flow. On the other hand, enhanced transfer of water to the subsoil may result in mass failures in clayey soils prone to landslides (Booker et al. 1993). Using simulated rainfall, patterns of localization of overland flow were observed by Cerda et al. (1998) in southwest Spain and by Imeson et al. (1992) in North America. Both studies found that the extent of overland flow was limited, however, because of the interspersed presence of hydrophilic soils.

Soil Nitrogen

Key points:

Ecological effects	Management considerations
• Loss of nitrogen (N) during fire varies widely, from nominal to a large proportion of the ecosystem total.	• Greater fire severity leads to greater duff consumption, and, consequently, N loss. • Estimate N loss for prefire planning or postfire accounting as 10 kg ha^{-1} for every 1 Mg ha^{-1} of duff consumed. • Develop simple accounting procedures to document cumulative N losses from repeated fuel reduction treatments.
• Nearly all loss of N is from litter and duff.	• Impacts to mineral soil N are not a strong consideration for burn programs.
• N loss is strongly influenced by the mass and moisture content of the forest floor.	• Burning of moist duff layers (> 65 percent moisture) limits duff consumption and N loss.
• Whether N losses are damaging to soil or site productivity depends on how fertile a site is. Nutrient-limited sites are at risk of detrimental N loss from severe burning. Conversely, large N losses may be inconsequential on sites with large N reserves.	• Avoid severe burning and complete duff consumption at nutrient-poor sites. • Consider mechanical treatments for extremely poor sites.
• A flush of plant-available N results after fire because of downward movement and condensation of fire-volatilized N.	• Low- to moderate-severity burning probably does not release enough inorganic N to noticeably benefit plant growth. • Postfire plant responses are more likely a function of reduced plant competition and favorable seedbed conditions.

Prescribed burning can result in two major changes in soil N: (1) loss of total N to the atmosphere during the consumption of litter and duff layers, and (2) a pulse of plant-available N owing to downward movement and condensation of fire-volatilized inorganic N. This is a double-edged sword—simultaneous loss of total N and gain of inorganic N—with the two processes linked such that burns resulting in a high loss of total N also yield the greatest increase in plant-available N (Wan et al. 2001). From there, a cascade of related soil N transformations may follow suit when burning is sufficiently hot (e.g., changes in microbial N pools, mineralization rates, nitrification, N fixation, enzyme activity). Unfortunately, predicting the exact quantity of total N lost, inorganic N gained, or changes in other N properties is difficult given the variable findings from literature, themselves reflecting the range of site characteristics and burn prescriptions used throughout the United States.

Acknowledging the importance of N to forest function and the wide range of reported soil N responses to prescribed fire leads us to ask the following questions:

- How extensive are changes in total soil N and inorganic N?
- What site conditions and burn prescriptions lead to the greatest changes in soil N?
- Are N losses damaging to ecosystem health? If so, will N eventually be replenished?

These questions are addressed here from the standpoint of a single burn, recognizing that repeated burning is a more complicated and often preferred component of fuel reduction programs. The following information then is intended as a foundation for understanding the effects of repeated burning on soil N, which is discussed later in the chapter.

How extensive are changes in total soil N and inorganic N?—

Total soil N—In a classic review of fire and soil N transformations, Raison (1979) stated that "in any one fire, nutrient loss is usually but not universally small relative to the total soil and biomass reserve." If true, then the assumption for modern-day prescribed burning is there should be little concern about soil N loss, regardless of fire severity. But what does "usually but not universally" mean? And how small is "small"? As a counterpoint, Johnson and colleagues (Caldwell et al. 2002; Johnson et al. 1998, 2009; Murphy et al. 2006) argued that awareness of soil N loss is one of the most important aspects of fire-soil dynamics. Table 6 articulates these contrasting views by showing published examples of N loss that range from minor to severe during a single prescribed fire. Clearly, any blanket statement about the effects of prescribed burning on soil N loss is too simplistic without attention given to preburn soil conditions and fire severity. However, it is important to recognize that N losses can be severe in certain circumstances and that preventative efforts to limit N loss may be needed or encouraged.

Nearly all N lost from the studies reported in table 6 originated from litter and duff, not mineral soil. For example, Moghaddas and Stephens (2007) found no significant changes in mineral soil N content even though up to 723 kg ha^{-1} N was volatilized from the forest floor (fig. 20). Similarly, in a meta-analysis of fire and

Table 6—Nitrogen (N) loss during prescribed burning from selected published studies across the United States

Extent of N loss	N loss	Region	Fire type	Study
	kg ha^{-1}			
Low to moderate	40 to 88	SE	UB	Bell and Binkley 1989
(0 to 200 kg ha^{-1})	39 to 194	SE	UB	Hough 1981
	11 to 40	SE	UB	Richter et al. 1982
	55	W	CTLB	Gundale et al. 2005
	55	SE	UB	Hubbard et al. 2004
	54	SE	UB	Kodama and Van Lear 1980
	150	SW	UB	Covington and Sackett 1984
	56 to 61	W	UB	Caldwell et al. 2002
	94	W	SB	Jurgensen et al. 1981
	1 to 134	SE	UB	Vose and Swank 1993
	119	W	UB	Klemmedson et al. 1962
	114	W	WTB	Murphy et al. 2006
Moderate to severe	252	W	CTLB	Murphy et al. 2006
(200 to 750 kg ha^{-1})	200 to 600	W	SB	Little and Ohmann 1988
	225 to 571	W	SB	Little and Klock 1985
	347 to 435	W	CTLB, WTB	Shea 1993
	362	W	UB	Caldwell et al. 2002
	353	W	UB	Klemmedson et al. 1962
	551 to 723	W	UB, CTLB + masticate	Moghaddas and Stephens 2007

Regions include SE = Southeast, SW = Southwest, W = West. Fire types are UB = underburn, WTB = whole tree thin and burn, CTLB = cut-to-length thin and burn, and SB = slash burn.

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Figure 20—High-severity burning with near-complete consumption of surface residues resulted in large losses of forest floor nitrogen (N), but little change in mineral soil N (Moghaddas and Stephens 2007).

soil N studies, Wan et al. (2001) found that prescribed burning had essentially no effect on mineral soil N content. These findings clarify that soil N loss is primarily a surface organic matter phenomenon and suggest that efforts to limit N loss—when deemed necessary—focus on conservation of the duff layer.

Inorganic N—Soil solution ammonium (NH_4) and nitrate (NO_3) are the primary inorganic N compounds assimilated by plants. Without fire, their concentrations are usually low and vary with seasonal trends in plant uptake, leaching loss, and N mineralization, nitrification, and immobilization rates (Fisher and Binkley 2000). Fire acts to jumpstart the soil inorganic N pool as a portion of the volatilized N gas travels downward and condenses as NH_4 in the forest floor or mineral soil. Ammonium concentrations are usually elevated immediately after burning and are then converted to NO_3 by nitrifying bacteria within several months. Wan et al. (2001) found in their meta-analyses that prescribed fire resulted in more than a doubling of the peak concentrations of NH_4 and NO_3 on average. They also found that the increase in both NH_4 and NO_3 was greatest in the surface 2.5 cm of mineral soil and that the increase was ephemeral, lasting for 1 to 2 years before returning to preburn levels.

Most fire studies report inorganic N gains on a part per million or concentration basis, making ecological interpretations difficult. Increases in N concentration may be statistically significant, but whether they affect plant N uptake and growth is often ambiguous. Fortunately, a few studies have also estimated the inorganic N flush on an area basis, which provides a more complete picture of the biological significance of this process (Covington and Sackett 1992, Hart et al. 2006, Jurgensen et al. 1981). For example, Covington and Sackett (1992) found that high severity prescribed burning in old-growth ponderosa pine consumed nearly 100 Mg ha^{-1} of forest floor material and released a modest 34 kg ha^{-1} of plant-available N. Their low- to moderate-severity burns in younger stands released only 4 to 15 kg ha^{-1} of inorganic N, which is in general agreement with the findings of Jurgensen et al. (1981) and Hart et al. (2006). These few studies suggest that prescribed burning will not release enough inorganic N to noticeably benefit plant growth unless burning is fairly severe or is conducted annually or fairly frequently. Instead, postfire responses by vegetation are more likely due to improvements in seedbed properties (lower forest floor depth, greater light availability, less plant competition for water and nutrients, seed scarification) than to a flush of inorganic N.

What site conditions and burn prescriptions lead to the greatest changes in soil N?—
Site conditions that lead to high forest floor content such as (1) high litterfall rates in dense forest stands, (2) slow decomposition rates in cold or dry climates, and (3) a lack of recent forest floor disturbance from burning or harvesting contribute to the greatest potential for N loss. Actual N loss then is a function of the forest floor N content and the burn prescription that either encourages or limits duff consumption. Not surprisingly, duff moisture content is the primary prescriptive variable explaining duff consumption (Brown et al. 1985, Hille and Stephens 2005). Burning of moist duff layers results in low consumption (fig. 21) and presumably low N loss.

It is possible to use duff consumption as a surrogate for estimating N loss. For example, studies from the Southeast United States have shown that forest floor consumption on a weight basis is an excellent predictor of total N loss in pine forests (Hough 1981, Schoch and Binkley 1986). We extended this concept to forests across the United States by plotting N loss versus forest floor consumption for the studies listed in table 6. A strong, positive relationship was found among all sites, showing that 10 kg ha^{-1} N loss resulted for every 1 Mg ha^{-1} of forest floor consumed (fig. 22). Interestingly, low N losses were most common for studies from the Southeast where warmer annual temperatures and consistent seasonal precipitation patterns suggest higher duff decay rates, lower duff accumulation, and wetter duff layers compared to western forests. Fire planners and soil scientists can use the predictive relationship from figure 22 to help set burn prescriptions when limiting N losses from fire is recommended, or as a means to estimate N losses when burning. Consider an example from young ponderosa pine forests in central Oregon where total ecosystem N is approximately 2500 kg ha^{-1} (Little and Shainsky 1995). If loss of no more than 15 percent of the site N, or 375 kg ha^{-1}, is recommended, then burn prescriptions and fuel moisture contents can be selected using a fire model such as Consume 3.0 (http://www.fs.fed.us/pnw/fera/research/smoke/consume/consume30_users_guide.pdf), which targets forest floor consumption of 37.5 Mg ha^{-1} or less. Alternatively, a more severe burn in this forest type that hypothetically consumes 90 percent of the forest floor (60 Mg ha^{-1} of duff) will volatilize 600 kg ha^{-1} N, or 24 percent of the total ecosystem N.

Are N losses damaging to ecosystem health? If so, will N eventually be replenished?—
We are not aware of any empirical evidence that a single prescribed burn has ever altered plant composition or growth in healthy ecosystems through its effect on

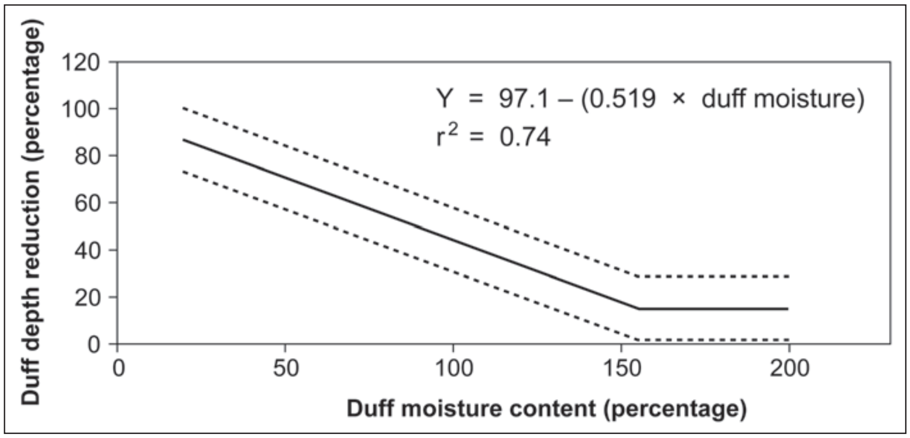

Figure 21—High duff moisture content at the time of prescribed burning limits duff consumption. Data are from several conifer stands in the northern Rocky Mountains (Brown et al. 1985). Dashed lines are one standard error from the regression line. Duff moisture content refers to the average moisture content in the lower one-half of the duff.

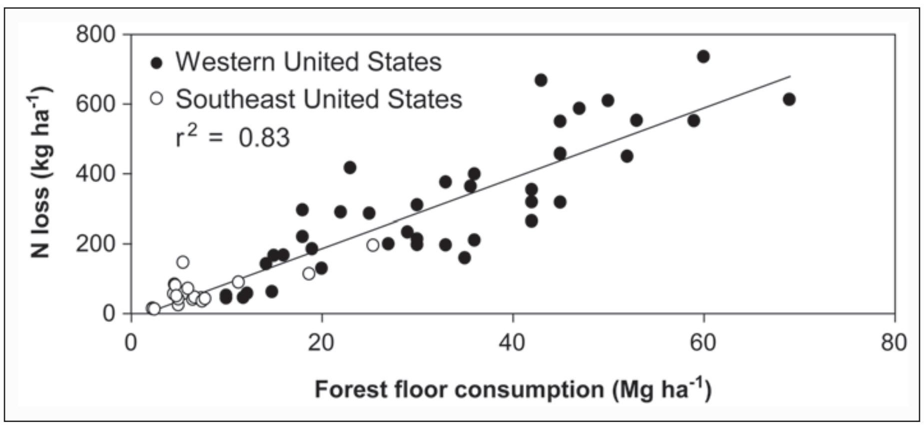

Figure 22—Predicted nitrogen (N) loss as a function of forest floor consumption. Data are from studies conducted in the Western and Southeastern United States, as listed in table 6.

total soil N. Certainly the range of N loss for low- to moderate-severity burns (0 to 200 kg ha^{-1}) represents a minor proportion of the total N in most ecosystems and should have little impact on essential functions. Whether larger N losses associated with severe burning are ultimately damaging to ecosystem health depends largely on the size of the total N reservoir at a given site. For example, the loss of 723 kg ha^{-1} N measured by Moghaddas and Stephens (2007) in a productive mixed-conifer forest is quite dramatic, yet it represents only about 5 percent of their site's total N reservoir[2] and presumably will not affect site functions. A similar loss from a less productive forest may have different results, however. Shea (1993) found that about

[2] Powers, R.F. 2010. Personal communication. Research forester, retired. Pacific Southwest Research Station, 3644 Avtech Parkway, Redding, CA 96002.

50 percent consumption of the forest floor in dry, nutrient-poor ponderosa pine forests volatilized 435 kg ha^{-1} N, or about 18 percent of the total N reservoir. Although this N loss has not resulted in a change in plant diversity or productivity to date (Busse et al. 2009a), it represents a sizable deficit that will not be easily replenished and will likely become exacerbated by future treatment.

A quick procedure to estimate the percentage of the total ecosystem N lost by burning is to (1) know your site—estimate the total N reservoir in soil, forest floor, and vegetation using published results or professional experience; (2) estimate the total N loss from fire by assuming 10 kg ha^{-1} N loss for every 1 Mg ha^{-1} of forest floor consumed; (3) divide the N loss in step 2 by the total N reservoir estimated in step 1. Although crude, this method offers a rapid assessment of potential changes in soil quality. This information can also be used as a starting point for developing a running total of a site's N budget following additive treatments.

Any reductions in total N resulting from fire may be effectively offset if postfire increases in N mineralization rates (microbial release of NH_4) result. This can occur if fire reduces the forest floor depth and results in warmer soil temperatures, greater availability of NH_4 substrate, and, consequently, greater microbial activity and N turnover (Schoch and Binkley 1986). However, there is no consistent evidence in literature to support this sequence of events. Studies of postfire N mineralization have found no effects (Hart et al. 2006, Hubbard et al. 2004, Kaye and Hart 1998, Moghaddas and Stephens 2007), positive effects (Schoch and Binkley 1986), detrimental effects (Bell and Binkley 1989), and variable effects (Gundale et al. 2005, White 1986) of fire on soil N mineralization. In fact, DeLuca and Zouhar (2000) found contrasting short-term increases and longer term declines in mineralizable N when comparing a chronosequence of burns in western Montana forests. Therefore, any expectations that increases in N mineralization will partially offset the loss of N may be unrealistic.

Finally, we ask the academic question of how long it would take to replenish N losses resulting from fire by a combination of atmospheric N deposition and N fixation by free-living soil bacteria or N-fixing plants (the assumption here is that N needs to be fully replenished, which may not be valid in many cases). Table 7 highlights the tremendous range in time that would be required to balance ecosystem N following fire. For example, millennia are needed to replace N loss from a low- to moderate-severity prescribed fire if the only N source is from free-living, N-fixing soil micro-organisms. In contrast, N losses of 100 kg ha^{-1} can be offset in as few as 3 years in moist forests near urban centers by a combination of N deposition and symbiotic N fixation. The variation in atmospheric N deposition with proximity to

Table 7—Years required to offset nitrogen (N) losses from prescribed fire by atmospheric deposition, free-living N fixation, or symbiotic N fixation[a]

N process	Condition	Estimated N fixation rate	Years required to offset N loss of: 100 kg ha^{-1}	400 kg ha^{-1}
		(kg/ha/yr)		
Atmospheric deposition	Remote forest	1	100	400
	Near urban center	15	7	27
Free-living N fixation	Cool, moist forest	0.2	500	2,000
Symbiotic N fixation	Dry forest, *Purshia* understory/ 30 percent cover	1	100	400
	Dry forest, *Ceanothus* understory/ 30 percent cover	10	10	40
	Moist forest *Ceanothus* understory/ 30 percent cover	20	5	20

[a] N fixation rates are estimates from Weathers and Lynch (2011) for atmospheric deposition, Hendrickson (1990) for free-living N fixation, and Busse (2000) for symbiotic N fixation.

urban centers is attributable to the strong influence of localized air pollution and N emissions (Weathers and Lynch (2011). In fact, deposition rates near large cities are sufficiently high to surpass critical loads for N saturation, leading to possible disruption of key ecosystem functions (Fenn et al. 2010). Here, the use of repeated prescribed fire to volatilize N is a potential management option for releasing excess N and avoiding N saturation conditions (Fenn et al. 2010).

If we assume that N gains from atmospheric deposition are balanced in natural (remote) ecosystems by N losses from leaching, then postfire recruitment of N-fixing plants becomes an important mechanism for replacing N loss. As an example, Johnson et al. (2004) predicted a complete recovery of postfire N by the presence of the N-fixing species snowbrush (*Ceanothus velutinus* [Douglas x Hook.]). However, the success of this process will differ considerably depending on which N-fixing species are present and the amount of ground cover they occupy. Slow-growing shrubs and small trees (e.g., bitterbrush, mountain mahogany [*Cercocarpus* Kunth]) likely fix enough N to meet their own limited demands but contribute little to the soil N budget (Busse 2000). Faster growing shrubs, legumes, and trees can provide much higher inputs of fixed N if their presence is widespread (fig. 23) (Domenach et al. 1989, McNabb and Cromack 1983). In some cases, burn prescriptions can be tailored to enhance recruitment of N-fixing plants, particularly for fire-adapted species like snowbrush that are capable of vegetative sprouting or for species that have a well-established seed bank. Because these are site-specific processes, we recommend talking with local plant ecologists about the potential for recruiting N-fixing plants following fire.

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Figure 23—Postfire recruitment of nitrogen (N)-fixing shrubs like snowbrush can facilitate the recovery of site N losses. In this example, snowbrush cover (about 30 percent) will fully replenish N losses from a low- to moderate-severity fire in about 10 years.

Repeated Burning

Key points:

Ecological effects	Management considerations
• Soil effects are controlled by burn severity and fuel consumption for both individual and repeated burns.	• Prescribe burn conditions to meet fuel **and** soil objectives.
• Fire exclusion causes nutrients to accumulate in forest floor fuels and dead organic matter. Large amounts of nutrients can be lost when heavy fuel loads are burned.	• Use low- to moderate-severity burning and leave a sufficient portion of the duff layer intact to temper nutrient loss. Estimate the quantity of nutrients you want to retain, or how much you are willing to lose, and select burn conditions and forest floor consumption levels to match your objectives.
• The more frequently a stand is burned, the greater the nutrient loss from the forest floor compared to long-unburned controls.	• Increase the interval between burns or design repeat burns to be heterogeneous, leaving areas of forest floor unburned, when this is a concern.
• Repeat burning can greatly reduce forest floor fuels and nutrients, but there are no consistent effects on mineral soil characteristics.	• Impacts to mineral soil nutrients may not be a strong factor when deciding to repeat burn, except at sites with specific nutrient deficiencies or concerns.
• Repeat burning may stimulate nitrogen (N)-fixing understory plants in some vegetation types, helping replenish lost N.	• Take note of understory development following prescribed fire. Where fuel objectives are not compromised, allow postfire N-fixing plants to establish.
• Nutrient-limited sites are at risk of productivity impacts from repeat burning.	• Consider mechanical methods to reduce fuels.

Many fire-adapted ecosystems prevailed for thousands of years with frequent fire. Reintroduction of fire, on some periodic basis, is often recommended to restore or maintain these systems and curtail hazardous fuel accumulations. Perhaps the most extensive research on the forest impacts of long-term, frequent prescribed fire programs has come from the U.S. southeastern Coastal Plain. The pine woodlands in this area, often dominated by longleaf or loblolly pine (*Pinus taeda* L.), are among the most fire-adapted forests in the United States. Fires historically burned at 1- to 4-year intervals (Brockway and Lewis 1997, Wade et al. 2000), and forest management practices often include frequent prescribed fire as a silvicultural tool to reduce fuels and help maintain these southern pine systems (Stanturf et al.

2002). Related research has also been conducted in the Northeast (Neill et al. 2007, Tuininga and Dighton 2004), Southwest (Covington and Sackett 1986, Wright and Hart 1997), and Northwest (Busse and Riegel 2009, Busse et al. 2009a).

Repeat fire reduces forest floor biomass and nitrogen with limited impacts on soil total nitrogen—
Numerous soil studies have examined impacts following 20 to 65 years of annual burning, as well as comparisons with fires every 2, 3, 4, or more years in southern pine forests (Bell and Binkley 1989, Binkley et al. 1992, Brockway and Lewis 1997, Jorgensen and Hodges 1970, McKee 1982, Waldrop 1987). The consistent results from these studies show reductions in forest floor biomass and the concomitant loss of nutrients in the combusted material. Some portion of the nutrients may be lost to the atmosphere and moved offsite, whereas others may be moved downward into the mineral soil or redistributed locally as fine ash. Studies typically show N losses from the forest floor. The more frequent the burns, the greater the cumulative reduction in forest floor N content. For example, following 30 years of prescribed fire treatments in the Coastal Plain of South Carolina, forest floor N mass was reduced by 29, 60, 60, and 85 percent following fires every 4, 3, 2, and 1 years, respectively, relative to the 480 kg N ha^{-1} in the control stand (Binkley et al. 1992). McKee (1982) also reported increased N losses from the forest floor with increasing fire frequency at Coastal Plain sites. At a Florida site, annual burning reduced forest floor N by 95 percent, whereas prescribed fires every 4 years led to a 72 percent reduction, relative to the 131 kg N ha^{-1} in the control. At a South Carolina site, annual burning reduced forest floor N by 68 percent, and periodic fire every 7 years resulted in a 32 percent loss of N relative to the 408 kg N ha^{-1} in the control (McKee 1982).

Despite dramatic reductions in forest floor mass and nutrients following decades of repeat burn treatments, there is no clear and consistent effect of long-term repeat burning on total N contained in mineral soil. Impacts from prescribed fires are typically limited to the uppermost soil layers, and the following examples show data from the top 10 to 15 cm of mineral soil. For example, soil N decreased by 110 kg N ha^{-1}, or 7 percent, following 20 years of biennial burning in southwestern pine stands (Wright and Hart 1997). During 30 years of prescribed fire, soil N both increased and decreased at a South Carolina Coastal Plain site, depending on burn frequency and season (McKee 1982). Comparing measurements at 10 and 30 years of treatment, the winter burns increased soil N by 3 and 10 percent, respectively, after periodic (about 7-year intervals) and annual burns, whereas summer burns decreased soil N by 9 and 22 percent, respectively, for the same burn frequencies. During the same 20-year period, soil N in the control stand increased by 4 percent. The author suggested that seasonal influences may be related to the growth of N-fixing plants following burns (McKee 1982). After 30 years of repeat burning in a loblolly/longleaf pine forest, Binkley et al. (1992) measured minor gains of 1 and 6 kg N ha^{-1} in the surface 10 cm of mineral soil following burn intervals of 1 and 3 years, respectively, and losses of 25 and 1 kg N ha^{-1} after 2- and 4-year intervals, respectively, but none of these were significantly different than the 1280 kg N ha^{-1} contained in the unburned control. Prescribed fires typically burn at low to moderate intensities, and the resulting degree and duration of heating generally retain most soil N (Certini 2005, Johnson et al. 2009).

Covington and Sackett (1986) proposed that repeated use of prescribed fire can enhance forest productivity by releasing periodic pulses of available, inorganic N in the form of NO_3 and NH_4. However, following 20 years of biennial burning of ponderosa pine stands in the Southwest, inorganic N levels did not differ between burn plots and controls (Wright and Hart 1997). Using restoration treatments designed to emulate historical vegetation and fuel conditions in the inland West, Hart et al. (2005b) showed that the N released from frequent prescribed fires was less than a third of the plant demand for N at the site. They argued that, rather than directly cycling N to maintain productivity, repeat fires are more important in maintaining open stand conditions that allow N-fixing herbs and microbial N mineralization to supply N for plant uptake. Following 9 years of annual burning in a longleaf pine stand, Greene (1935) found that legume density was more than twice that measured in unburned areas. He suggested that accumulating plant debris in the unburned stand had a smothering effect on leguminous plants, and would lead to reduced soil N compared to the frequently burned area. In contrast, Busse and Riegel (2009) reported that frequent fires in Oregon ponderosa pine forests likely hindered reproduction of N-fixing bitterbrush shrubs in favor of grasses. Fire suppression there has increased the abundance of bitterbrush, which acts as a ladder fuel in dry forests. These conflicting reports indicate that each forest system responds uniquely to frequent fire. Taken together, we surmise that while prescribed fires often release available N, this may not occur in all forest types. The role of N-fixing plants in increasing soil N following prescribed fire is also site specific. Where N-fixers are stimulated by prescribed fire, they may be an important, though not sole, source for replenishing some of the N lost through combustion.

Prescribed fire frequency in contemporary, managed landscapes—
Once an initial prescribed burn has been implemented (with or without thinning), how do we decide if, or when, to burn again? Considering that many forests historically burned with very short return intervals (one to two decades or less), repeat burning is often proposed or justified in the context of historical fire regimes. Forest managers recognize the historical role of frequent fires as a fundamental

disturbance agent that influences plant succession, nutrient cycling, and other ecosystem processes. Presumably, frequent fire kept fuel loads in check, and nutrients lost during fires did not reduce forest productivity and soil quality because, over time, they were replenished by inputs from plants, the atmosphere, or mineral weathering. Management activities including fire suppression, timber harvesting, grazing, recreational use, and urbanization have tremendously altered forest stand conditions, disturbance regimes, and associated ecological processes (fig. 24). Knowledge of historical fire regimes can provide an important frame of reference when designing forest management strategies (Mutch and Cook 1996), but impacts to nutrient pools, availability, and cycling should not be overlooked when considering restoration of fire regimes on today's landscapes.

Fire exclusion has altered nutrient cycling and storage by allowing organically bound nutrients (unavailable for plant uptake) to accumulate in organic matter on the forest floor. Substantial amounts of N can be lost when heavy fuel accumulations burn during repeat prescribed fires. For example, heavily stocked ponderosa pine stands in central Oregon were burned in 1991 and 2002, and the resulting N loss from the two fires was 550 kg ha^{-1}, or 22 percent of the total ecosystem N of the site (Busse and Riegel 2005). This forest is relatively dry and infertile compared to more productive stands in moister climates. The frequent use of prescribed fire in this or similar forests may affect long-term site productivity by volatilizing greater and greater proportions of the total site N. Forests with greater overall N stores and nutrient cycling rates should be more resilient to N losses from frequent fire because of their increased ability to buffer against nutrient changes.

Figure 24—Mixed-conifer stands characterized by frequent low- to moderate-severity historical fire regimes. Open stand in the Sierra San Pedro Martir, Mexico, has experienced little to no fire suppression (left). Dense, ingrown stand in the Sierra Nevada, USA, has experienced aggressive fire suppression for more than a century (right), but has remnant large trees with fire scar evidence of frequent historical fires (right, inset).

Pile Burning

Key points:

Ecological effects	Management considerations
• The amount of soil heating during pile burning varies greatly depending on fuel composition. In particular, piles dominated by large-diameter wood (>25 cm) can produce extreme soil heating.	• When possible, avoid burning piles that contain a high percentage of large wood. However, extreme soil heating may be of little concern if these piles are widely spaced and occupy a small percentage of the land surface. • Piles containing a mix of fuel sizes (e.g., precommercial thinning slash—poles, limbs, tops) will generally not produce excessive soil temperatures.
• The diameter and arrangement of hand piles (numerous small piles vs. fewer large piles) appears to have little effect on soil quality.	• Other considerations (cost, fire risk, human safety) are probably more important than potential soil effects when planning pile size specifications.
• The amount of land surface covered by piles is a critical soil consideration.	• Determine the amount of ground coverage occupied by piles of all types. Visual estimates, small plots, or transects can provide rapid estimates and help when assessing cumulative effects of repeated treatments. • Expect some changes in postburn soil properties such as (1) lower total carbon (C) and nitrogen (N), root biomass, microbial community size and function, and water infiltration; (2) increased calcium (Ca), magnesium (Mg), potassium (K), pH, inorganic N, and tree growth from the “ash-bed effect.” • Where ground coverage is high (>15 to 30 percent) and large-diameter fuel is prevalent, consider leaving some piles unburned for several years or scattering some of the wood to decay on site.

Soil heating during pile burning can be extreme. Massman and Frank (2004) measured soil temperatures of 400 °C beneath a large slash pile, with temperatures remaining elevated for several days. Significant changes in soil physical, chemical, and biological properties are likely in this circumstance (see table 2). But not all pile burns result in extreme soil temperatures or soil damage (Busse et al. 2013, Meyer 2009). The severity of an individual burn depends on a complexity of fuel, soil, and climatic conditions at the time of ignition, making it difficult to generalize about the effects of pile burning on soils (fig. 25). Here we offer a brief background on observed soil responses to pile burning, along with (1) results from a case study from the Lake Tahoe Basin that compare the relative importance of pile size, fuel composition, and distance from streams on soil and site quality, and (2) an outline of factors to consider when specifying preferred pile conditions.

Background—
Studies have reported a variety of changes in surface soils from pile burning. Common responses include volatile loss of organic C and N, transient increases in plant-available N and phosphorus, reduced microbial activity, and changes in soil mineralogy (Covington et al. 1991, Jiménez Equilín et al. 2007, Loupe et al. 2007, Massman and Frank 2004, Miller et al. 2005). Ulery and Graham (1993) noted redder hues in surface-burned soils, significantly reduced organic C contents, and the formation of sand-sized aggregates that altered the soil texture. In addition, they observed the collapse of interlayer spacing of secondary clay minerals and the dehydroxylation of iron-bearing phyllosilicates (Ulery et al. 1996). Goforth et al. (2005) observed a significant increase in calcium carbonate and soil pH in surface soils where logs had thoroughly combusted. At temperatures that normally occur under pile burns, Hubbert et al. (2006) noted an increase in soil bulk density and a decline in soil porosity, factors that could lead to reduced water infiltration. Decreased infiltration in the surface soil may also occur owing to soil water repellency formed from condensation of organic compounds in the soil profile (DeBano 1981).

Pile burning is also responsible for the so-called "ash-bed effect" in which the release of nutrients (particularly N, Ca, Mg, K) from organic materials can temporarily augment soil fertility (Knoepp et al. 2005). As an example, York et al. (2009) found 10-year height and diameter growth of conifer seedlings as high as 49 percent greater within pile burn perimeters compared to adjacent, unburned ground. Because the study area was generally free of competing vegetation, they surmised that the tree response was due to improved soil N availability following burning.

Carol Shestak

Ken Hubbert

USDA Forest Service

Ken Hubbert

Figure 25—Piling, curing, and burning of slash piles.

Pile burning in the Lake Tahoe Basin: a case study—

Lake Tahoe straddles the borders of California and Nevada in the Sierra Nevada Mountains, and is treasured for its deep blue waters and picturesque scenery. Extensive timber harvesting in the 1800s, grazing, fire exclusion and suppression, drought-induced insect outbreaks, and urban development have resulted in substantial changes in the composition and structure of the surrounding forests in the Lake Tahoe Basin (LTB). Today's forests have higher fuel loads, stand densities, and presence of shade-tolerant tree species than were found prior to settlement (Taylor 2004). Consequently, LTB planners and managers are balancing their efforts to reduce fuel hazards and restore forest condition while minimizing the input of sediments and nutrients into Lake Tahoe.

Pile burning of forest residues is now commonly practiced throughout the LTB as part of restoration efforts to reduce wildfire hazard and improve forest condition.

To better understand the effects of pile burning on soil properties and processes, we installed a series of research plots to address the following questions:

- How hot does soil get beneath pile burns?
- To what extent does pile size or fuel composition affect soil heating?
- Does burning within riparian zones result in unwanted release of nutrients to streams?
- What are the short- and moderate-term effects of burning on selected soil physical, chemical, and biological properties?
- What is the range in ground coverage occupied by piles within the LTB?

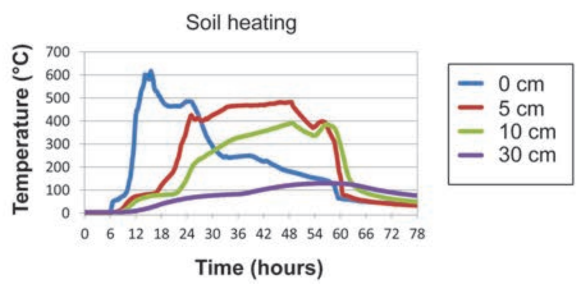
Soil heating was recorded at several depths in the soil profile beneath piles ranging from 1.8- to 6.1-m (6- to 20-ft) diameter with dominant fuel composition ranging from small woody slash to large-diameter bole wood. Burning was conducted in late fall, with fuel consumption essentially complete. Nutrient release in surface and subsurface water and changes in soil chemical, physical, and biological properties were monitored for two growing seasons after burning. Findings were as follows:

- Fuel composition was the primary driver of soil heating. Piles containing large wood from insect-killed trees reached extreme temperatures for extended durations, whereas piles from small-diameter thinning operations had shorter heat durations and moderate temperatures (fig. 26).
- Pile size had little effect on maximum soil temperatures or heat duration except when large-diameter wood was present in high amounts.
- Soil temperature declined rapidly with soil depth (fig. 26). The highest temperatures and durations were measured in the surface 5 cm, suggesting that the heating effects during pile burning are limited to surface soil. Also, soil heating was greatest beneath the pile center and declined considerably toward the pile edge.
- Nutrient release in surface and subsurface water was relatively low in the initial year after burning.
- Short-term changes in physical, chemical, and biological properties were found in the surface 5- to 10-cm of mineral soil. Mixed responses in the initial year included loss of fungal biomass, increased soil C content (from ash), and inconsistent changes in soil water repellency, total soil N, inorganic N, microbial biomass, and nutrient content.
- Ground coverage occupied by piles averaged 8 percent, with a range from 1 to 35 percent (n = 71 sites). Only a few sites, those with high tree mortality and, consequently, high fuel loads, exceeded 15 percent ground cover.

Fuel type 1:

- Dominated by large bolewood, including numerous pieces >40-cm diameter
- Severe soil heating to a depth of 10 to 30 cm

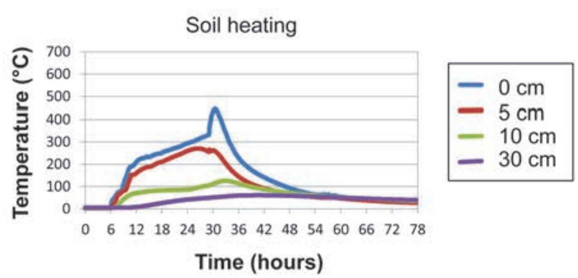
Bob Carlson



Fuel type 2:

- Mix of all fuel sizes (most common fuel type among inventory plots)
- Moderate soil heating to a depth of 5 cm

Carol Shestak



Fuel type 3:

- Mostly small-diameter wood and slash
- Low soil heating

Carol Shestak

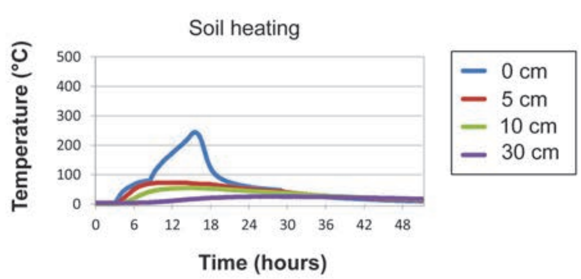


Figure 26—Soil heat pulse during pile burning in the Lake Tahoe Basin varies considerably depending on fuel type. (adapted from Busse et al. 2013).

Factors to consider when selecting pile specifications–
Factors to consider when selecting pile specifications include pile conditions, pile size, pile spacing, and repeated pile burning needs.

1. Pile conditions:

- **Fuel composition.** Fuel composition differs from site to site as a function of pretreatment fuel loading, stand conditions, and associated management objectives. For example, piling of precommercial thinning slash in conifer stands typically results in an assortment of small- and medium-sized fuels (<23 cm diameter), whereas piles built in areas with substantial tree mortality may contain high amounts of large woody material (fig. 27). Burning of large bolewood can have a tremendous effect on heating dynamics (Monsanto and Agee 2008) and postburn soil properties (Hebel et al. 2009). Results from the Tahoe case study further suggest that wood size is the primary factor controlling soil heating during pile burning. A simple precaution in cases where the consumption of large wood is anticipated would be to ensure that the percentage of ground coverage occupied by piles is well under 15 percent.

Matt Busse

Figure 27—A worst-case scenario. Preparing for pile burning of an entire stand of beetle-killed lodgepole pine (*Pinus contorta* Douglas ex Loudon). Ground cover in this example is near 30 percent.

- **Fuel moisture**. Reasonably dry and well-cured fuels lower the risk of a smoldering fire and, thus, lower the potential for high soil heating.
- **Pile quality**. Excessive mixing of mineral soil within piles can restrict the upward flow of heat and presumably result in excessive soil heating. However, to our knowledge, no published study has quantified a threshold for soil content within piles that translates to excessive trapping of heat. Modern forestry practices (e.g., use of brush rakes, operator awareness in limiting soil additions during tractor piling) are well suited to promote the construction of machine piles with a minimum of soil mixing.

2. Pile size:

Large piles with high fuel loads (e.g., machine piles) obviously generate more heat than smaller piles (e.g., hand piles) assuming near-complete fuel consumption. But this factor alone does not dictate that the highest soil temperatures or the greatest damage will be found beneath the largest piles. Most heat energy rises during burning (fig. 28) (Hungerford et al. 1991), particularly if fuels are well cured and dry, suggesting that the degree of soil heating is not proportional to the size of a pile. In support of this claim, Seymour and Tecle (2005) showed little effect of pile size on postfire soil physical properties, and Busse et al. (2013) found no significant relationship between pile size and maximum soil temperature or heat duration for piles ranging from 1.8- to 6.1-m (6- to 20-ft) diameter in the LTB case study.

USDA Forest Service, Deschutes National Forest.

Figure 28—Flaming and upward movement of heat.

Pile size can influence the amount of ground disturbance within a treatment unit. Numerous small piles result in greater ground coverage compared to fewer, large piles, assuming a common fuel loading per hectare. Thus the potential exists for more damage on an area basis if smaller piles are constructed. However, the differences in ground coverage resulting from pile size are not great, particularly for low and moderate fuel loads (fig. 29). This suggests that decisions regarding optimal size and number of piles per treatment unit can be made in most cases based on factors such as cost effectiveness, fire risk, and operator safety, rather than potential soil effects.

3. Pile spacing:

Extreme soil heating may be of little concern if it occurs beneath widely spaced piles that occupy little of the total land surface. Conversely, site and soil quality damage will occur when soil heating is extreme and the treatment unit has a high density of piles or plans for repeated treatments. Figure 30 identifies the number of piles per hectare required to exceed 15 percent ground cover for piles ranging in diameter from 1.8 to 6.1 m (6 to 20 ft). For example, it takes nearly 620 piles per hectare (250 piles per acre) to exceed 15 percent ground cover when the average pile diameter is 1.8 m (6 ft). Determining ground coverage and ensuring that it does not exceed 15 percent, when practicable, is an easy step that can be estimated from a simple field measurement of the number of piles per hectare within a treatment unit.

4. Repeated pile burning:

The cumulative effects of pile burning are not well understood; we could find no scientific literature on the topic. Whether the effects on soil are additive and therefore become increasingly detrimental to soil functioning with successive

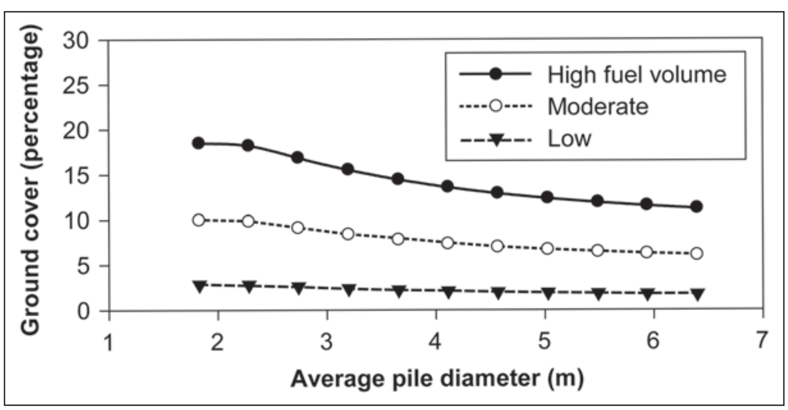


Figure 29—Amount of ground coverage occupied by piles within a treatment area as a function of pile diameter and total fuel volume (high = 1300 m^3 ha^{-1}, moderate = 700 m^3 ha^{-1}, low = 200 m^3 ha^{-1}).

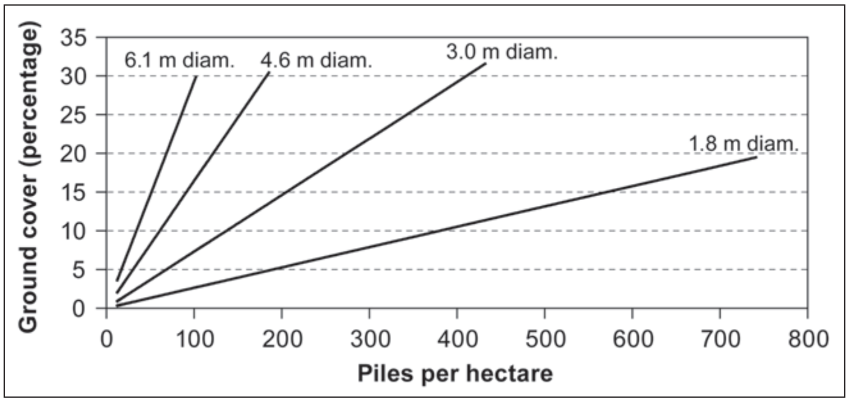


Figure 30—Estimating ground coverage occupied by piles for a range of pile sizes.

treatment, or, alternatively, whether soils effectively recover between burn cycles is untested. Certainly any cumulative effects will vary from site to site depending on burn severity, soil resilience, extent of ground coverage, and the length of time between burns. Frequent burning of wood piles on low fertility sites, for example, may be detrimental, whereas pile burning of mostly small slash material every 10 to 15 years on more productive sites is likely of little concern. But beyond these simple examples, it is impossible to make any definitive statements without relying on unproven assumptions. Again, a prudent approach to deal with this unknown is to document pile burn conditions for each entry. Qualitative assessments of fuel type, ground coverage, and burn severity can be made rapidly and entered in a database format to document site history for use in planning additional burns. Simple quantitative measures of soil quality, such as organic matter content or water infiltration rate, may provide additional planning insight for those sites with unique soil concerns.

Coarse Woody Debris

Key points:

Ecological effects	**Management considerations**
• Coarse woody debris (CWD) contributes to wildlife habitat, erosion protection, and fire hazard in many ecosystems. However, little definitive evidence exists proving or disproving the essential role of CWD in maintaining soil quality.	• Determining appropriate CWD retention levels remains a guessing game. Consider developing site-specific guidelines that set the upper limit based on fire hazard concerns and the lower limit by wildlife habitat needs or by erosion control on steep ground. Any possible contributions to soil quality can then be viewed as fortuitous. • Higher CWD loadings may be appropriate where larger log sizes dominate because of their benefit to wildlife habitat. • Spatial variability in CWD is often pronounced at broad landscape scales. Meeting rigid targets may be neither feasible nor desirable on a treatment-area basis.

Coarse woody debris (CWD; woody material >7.5 cm diameter) is well recognized as a fundamental structural and functional attribute in many forests (Harmon et al. 1986). Fire, insects, ice storms, and windstorms can introduce large quantities of CWD to the forest floor as trees fall and break. Woody debris, in turn, provides habitat and protection for wildlife, acts as an erosion control barrier, and serves as an energy source for numerous fungal and insect species (Harmon et al. 1986). Decaying wood also provides available summer moisture as well as habitat for fine roots, micro-organisms, and mycorrhizal fungi. During summer dry periods, fungal hyphae penetrate CWD and extract water and nutrients for critical sustenance. A food chain develops as invertebrates feed on the microbes, and in turn, mammals, reptiles, and birds feed on invertebrates. Nitrogen is released back into the soil through microbially mediated nonsymbiotic N fixation that is common in CWD. Additionally, nutrients and water are provided to conifers through associations with ectomycorrhizal fungi that inhabit decaying CWD.

Guidelines for CWD retention have been developed in many regions to safeguard the long-term presence of downed wood. For example, Graham et al. (1994) used ectomycorrhizae as a bio-indicator and recommended the following amounts of CWD to be left onsite in Montana: 11 to 20 Mg ha^{-1} (5 to 9 tons ac^{-1}) (Douglas-fir/ninebark), 27 to 54 Mg ha^{-1} (12 to 24 tons ac^{-1}) (Douglas-fir/pine grass; subalpine fir/bear grass; subalpine fir/twinflower), 16 to 31 Mg ha^{-1} (7 to 14 tons ac^{-1}) (grand fir/bear grass), and 18 to 40 Mg ha^{-1} (8 to 18 tons ac^{-1}) (subalpine fir/blue huckleberry). In Arizona, they recommended 11 to 29 Mg ha^{-1} (5 to 13 tons ac^{-1}) for ponderosa pine/fescue and Gambel oak habitat types. These values were later adopted by Brown et al. (2003) in their recommendation for CWD levels in inland West forests even though they remain to be rigorously tested or validated.

Recruitment of CWD to meet forest guidelines, including the provision of multiple decay classes (fig. 31), requires thoughtful planning. This is particularly true for fuel reduction treatments, as they can potentially modify CWD mass and distribution across the landscape. For example, prescribed fire can consume CWD, particularly logs in the later stages of decay (sound CWD is typically left unconsumed by underburning), whereas thinning operations may fragment decaying logs and disrupt their functional integrity. Alternatively, both practices may boost levels of small-diameter CWD if killed trees are retained onsite.

How much CWD to leave after prescribed burning or thinning does not come with a refined answer. Differing climates, forest types, and fire-return intervals all suggest the necessity for site-specific CWD guidelines. Adding to this uncertain equation is the fact that the spatial distribution of CWD is typically quite variable at landscape scales. For example, Rubino and McCarthy (2003) found that slope position strongly influenced the distribution and abundance of CWD. Lower slope positions had greater accumulations of CWD as a result of transport from upslope positions. Similar observations of spatial variability led Ganey and Vojta (2010) to suggest that maintaining average CWD loadings on every piece of ground is neither feasible nor desirable. They encouraged the use of malleable, landscape-scale targets that account for the many ecological roles of CWD while also addressing the reality of spatial heterogeneity and fire hazard concerns.

In this context, it is intriguing to ask whether CWD serves as an important contributor to soil productivity. Removing woody biomass from the forest floor rather than letting it decompose onsite may affect soil chemistry, which in turn influences soil fertility and plant growth. Further, CWD is an active source of nonsymbiotic N fixation, accounting for up to 50 percent of the total N fixed in some forests (Jurgensen et al. 1997). However, the relative amount of N fixed in CWD is exceedingly small compared to the total pool of ecosystem N. And although decomposing wood helps replenish soil nutrients, the net benefit is thought to be small because of the exceedingly low nutrient content of CWD. As an example, Prescott and Laiho (2002) noted that CWD did not contribute significantly to C, N, or phosphorus (P)

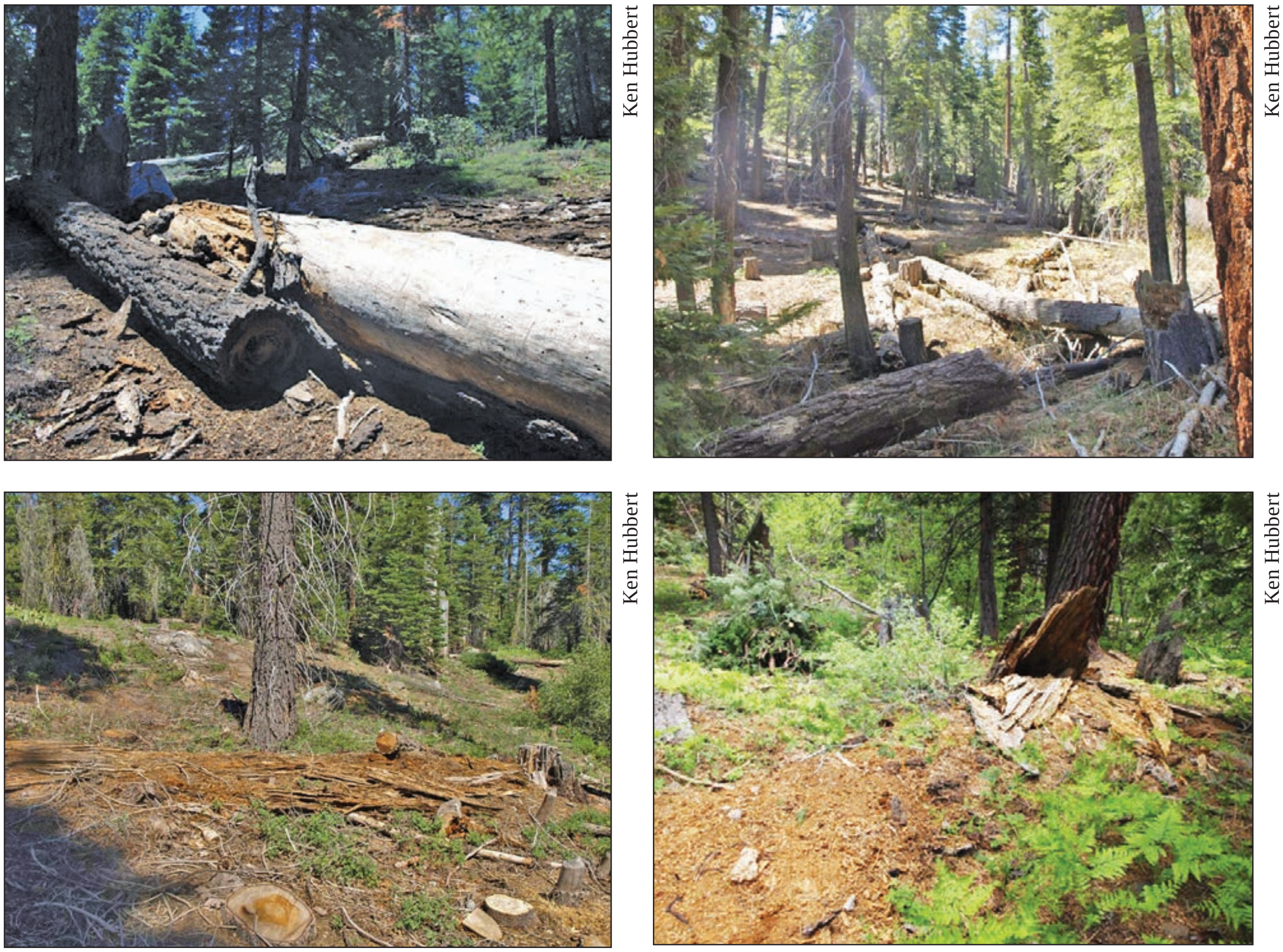


Figure 31—Photo series showing the five coarse woody debris decay classes: top left—decay classes 1 and 3; top right—decay class 2; bottom left—decay class 4; and bottom right—decay class 5.

cycling in the coniferous forests of the Rocky Mountains. At fir sites, they reported that CWD released 5 percent of the N, as compared to pine sites where the contribution of CWD to N and P release was ≤2 percent. In coniferous forests of central Oregon, Busse (1994) also found that the role of CWD in nutrient cycling was small. Adding to these findings are observations that CWD often occupies only a small fraction of the ground area in fire-adapted forests (Ganey and Vojta 2010, Stephens et al. 2007). Collectively, these observations suggest that CWD may not appreciably improve soil productivity or soil quality in many systems, and that managers may first consider other ecosystem components such as wildlife habitat or erosion control rather than soil nutrient cycling when identifying desired conditions for CWD.

Mechanical Fuel Reduction Treatments

Treating hazardous fuels in densely stocked stands often requires removing or processing many small-diameter stems per hectare. Conventional harvest techniques are being modified to reduce crown and ladder fuels, but few equipment options have been specifically designed to treat these fuels (Rummer et al. 2005). Equipment innovation will surely prompt questions about soil impacts. For example, novel use of in-woods chippers for fuel reduction can chip trees up to 60 cm (2 ft) diameter directly into a box mounted on a forwarder. By eliminating the need to skid the trees to a landing, soil compaction or displacement may be reduced, but the vehicle tire pressure and soil conditions will also play a role in minimizing compaction. Increasingly, whole-tree harvest methods are employed to treat fuels because smaller trees, including nonmerchantable materials, are targeted for removal and few slash treatments are required. Mechanical fuel treatments do not always include the harvest or removal of forest products, however. Onsite treatments can reduce ladder or crown fuels by shredding or crushing them at the stump and rearranging them on the forest floor, or cutting and piling them into burn piles.

Forest soil scientists have measured impacts of heavy forestry equipment and skid trails for more than 50 years, using physical parameters including bulk density, permeability, and macroporosity (Dyrness 1965, Steinbrenner and Gessel 1955). Although skidding equipment and the forest products they remove have changed substantially during this time, skidding impacts are largely unchanged—skid trails increase bulk density and reduce permeability and porosity compared to surrounding areas. To limit the deleterious conditions caused by skidding, much of the research supports the notion of creating a designated skid trail system and reusing existing skid trails (Ampoorter et al. 2007, Han et al. 2009, McIver et al. 2003, Olsen and Seifert 1984). However, the previous transportation network may be poorly suited for the current harvest. For example, landings may be located in sensitive areas such as riparian zones or near seeps or springs. If a landing will not be used, it is highly unlikely that the skid trails servicing that landing could be used either. Multiple treatments are often necessary to meet fuel management objectives (Reinhardt et al. 2008). Mechanical harvests can leave a surface fuel bed of slash, or regeneration of brush and trees over time can create an undesirable fuel load that may warrant additional treatments. Without careful planning, repeated stand entries can increase the skid trail density to cover up to 80 percent of the stand (Froehlich 1981).

Whole Tree Harvesting and Nutrient Removal

<table>
<tr><th colspan="2">Key points</th></tr>
<tr><th>Ecological effects</th><th>Management considerations</th></tr>
<tr><td>• Whole tree harvesting transports more nutrients offsite than removing only tree stems.</td><td>• In most cases, the site reserves will remain adequate to supply nutrients to remaining vegetation.
• Gain perspective by comparing estimated levels of nutrient removal with the amount held in the soil or whole ecosystem. For a ballpark estimate of the nutrients removed in crown material:
(1) Estimate crown fuel biomass, based on measurements or literature.
(2) Estimate crown nutrient content, based on measurements or species-specific or site-specific literature.
(3) Determine the percentage of crown to be removed.
Nutrients removed = (1) × (2) × (3).</td></tr>
<tr><td>• Nutrient-poor sites are most sensitive to whole tree harvesting:
▪ Glacial outwash sands
▪ Shallow soils
▪ Many coarse-textured soils</td><td>• Options to minimize nutrient loss:
▪ Harvest deciduous stands when dormant, so foliar nutrient losses are minimized.
▪ Backhaul slash and redistribute within the harvest area, being mindful of fuel loads.
▪ Extend the reentry period to allow more time for nutrient inputs.
▪ Fertilize.</td></tr>
<tr><td>• More nutrients are removed by whole tree harvesting productive sites.</td><td>• Productive sites are more resilient to nutrient removal. They tend to have greater rates of nutrient input and cycling, so have more to give.</td></tr>
</table>

Why whole tree harvesting is of concern—
Whole tree harvesting removes the entire aboveground portion of a tree (fig. 32). This raises concerns about nutrient loss and long-term site productivity because branches and foliage are removed along with the tree stems (Janowiak and Webster 2010). Although the foliage makes up a relatively small proportion of tree biomass, typically ranging from about 5 to 15 percent, depending on species and diameter (Ter-Mikaelian and Korzukhin 1997), it can contain more than half the nutrients stored in a tree (Little and Shainsky 1995, Phillips and Van Lear 1984). Two to three times the nutrients are removed from a site when the crowns are harvested along

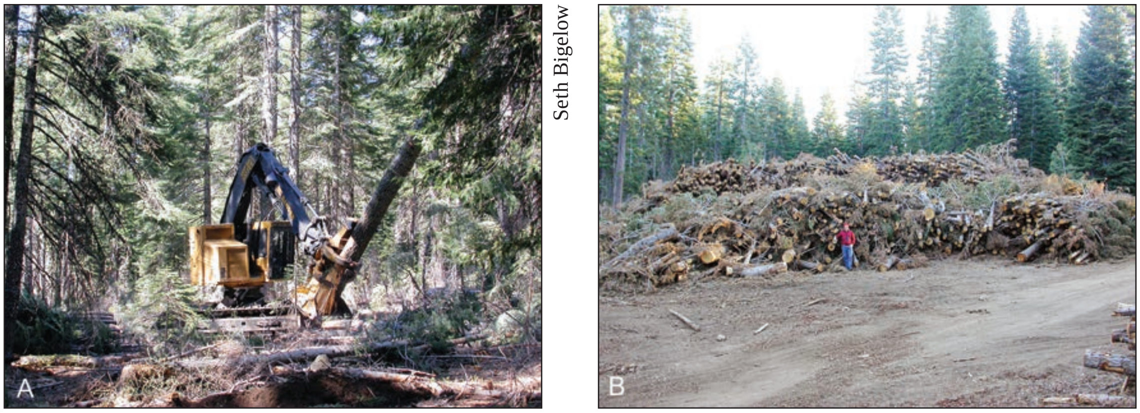


Figure 32—Feller bunchers are commonly used to harvest whole trees during fuel treatments. (A) A swing boom feller buncher can reach out to harvest fuels, eliminating the need to drive to each tree. (B) Whole tree harvest material is often processed at a landing.

with the boles (Alban et al. 1978, Patric and Smith 1975). With time, any harvesting of nutrients can deplete the soil if they are not balanced by inputs from precipitation, dry deposition, N fixation, and mineral weathering.

Many whole tree harvest studies document clearcutting effects—
Concerns over whole tree harvest on site nutrients have existed for decades (e.g., Kimmins 1977, Mälkönen 1976), and until recently, most research in U.S. forests has focused on clearcut harvests. Recent studies found that whole tree clearcuts led to only slight or insignificant reductions in tree growth in the regenerated stands relative to bole-only clearcuts. Following clearcuts in the U.S Pacific Northwest, the whole tree treatment resulted in a minor decrease in basal diameter growth, about 1 cm, compared to the bole-only treatment at 5 years of age (Ares et al. 2007). Concentrations of foliar N did not differ between the treatments, and the authors suggest that the small growth differences may have been caused by changes in microclimate between the treatments. A similar diameter decrease of 1.5 cm was measured in a 23-year-old plantation in North Wales following whole tree clearcutting (Walmsley et al. 2009). At this site, there were no significant differences in stand height, basal area, or density between whole tree and bole-only treatments at this age. In the Southeastern United States, Johnson et al. (2002) found no effect of clearcut residue treatments on tree biomass after 15 years at a site in Tennessee. In the same study, biomass was reduced 17 percent following whole tree clearcutting after 18 years at a site in South Carolina. The relative increased growth following bole-only harvesting was attributed to both greater nutrient uptake by trees and soil enrichment from the logging residues left onsite.

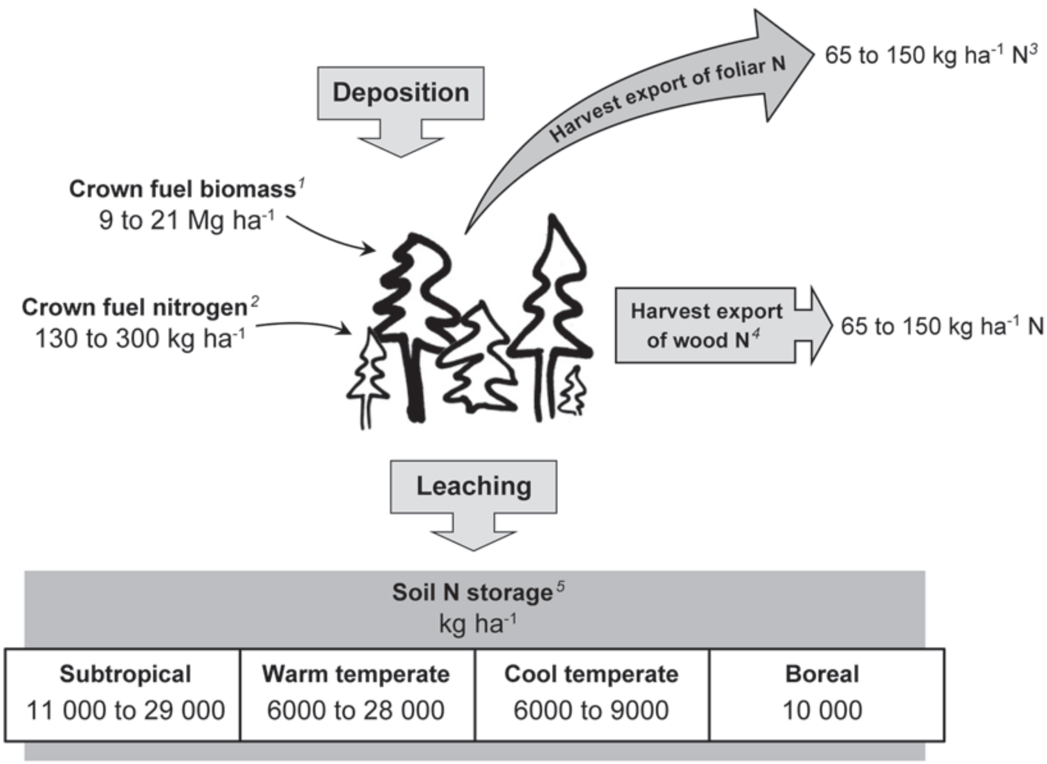
In addition to comparing vegetative growth following whole tree harvesting, studies often analyze soil nutrient capital to assess productivity impacts. A meta-analysis of data from 13 publications showed that whole tree clearcuts reduced soil C and N by an average of 6 percent relative to pretreatment data or controls (Johnson and Curtis 2001). A number of the included studies removed the forest floor in addition to logging residues. In soils under 10-year-old plantations, C concentrations declined where both forest floor and harvest slash was removed, but total C pools remained unchanged relative to the bole-only clearcut treatment (Powers et al. 2005). Powers et al. (2005) emphasized that the drop in C concentration was due to the loss of the forest floor rather than removal of harvest slash. Studies of stands greater than about 15 years old suggest that whole tree clearcut impacts to soil C and N stocks diminish with time (Jandl et al. 2007, Johnson et al. 2002, Jones et al. 2008, Walmsley et al. 2009).

Fuel treatments are often designed as stand thinnings, and remove far less biomass than a clearcut prescription. For example, fuel reduction thinnings in dense Sierra Nevada stands removed an average of 12 percent of the standing live volume (Collins et al. 2007), which in this case was equivalent to 21 percent of the basal area (Stephens and Moghaddas 2005). Fuel reduction treatments in dense stands typically reduce basal area by 20 to 45 percent (Boerner et al. 2008a) while retaining a majority of the standing volume onsite. Fuels are often thinned from below, so that understory, suppressed and intermediate trees are thinned before codominant or dominant ones. These lower crown positions have proportionately less canopy biomass (Reinhardt et al. 2006), and therefore fewer canopy nutrients, than the dominant overstory. Relying on whole tree clearcut studies to infer nutrient loss impacts following fuel reductions would grossly overestimate effects to soil nutrient pools and stand productivity.

Most research on whole tree thinning comes from Nordic countries—
Few whole tree thinning studies were found for U.S. forests, let alone fuel reduction thinning. However, in a fuel reduction study in dry forests of central Oregon, Busse et al. (2009a) compared the effects of whole tree harvest, bole-only removal, and thinning without biomass removal on vegetation responses. With periodic measurements spanning 17 years following the treatments, they found no differences among treatments in tree growth, shrub cover, or herbaceous biomass. When compared to the other residue treatments, whole tree harvest did not reduce the site potential or soil nutrient status of the relatively infertile sites they studied (Busse and Riegel 2005).

Whole tree thinning studies are most prevalent in forests of Nordic countries (i.e., Denmark, Finland, Norway, and Sweden), and have shown varying and inconsistent impacts on site productivity. Nearly all studies examined treatments in Scots pine (*Pinus sylvestris* L.) and Norway spruce (*Picea abies* [L.] Karst.) stands, two dominant forestry species in this region. In an analysis of 15 thinned stands of young Norway spruce and Scots pine, Jacobson et al. (1996) found, on average, no growth differences between whole tree and conventional (bole-only) harvests 5 years after treatment. However, they did observe both significant increases and decreases in growth in individual whole tree harvested stands. Ten years after these thinnings, they found that the whole tree harvest reduced stand growth an average of 5 to 6 percent compared to bole-only harvesting (Jacobson et al. 2000). Again, they also observed significant growth increases and decreases in individual stands. Other studies have shown only short-term growth reductions or no productivity impacts following whole tree thinnings. A Danish study of early whole tree thinnings of Norway spruce reported growth reductions for four growing seasons after thinning, but no significant growth differences over the next 6 years (Nord-Larson 2002). In addition, several studies in Sweden found no significant growth reductions 5 to 10 years after whole tree thinning treatments (Egnell and Leijon 1997, Mård 1998). In studies where reduced tree growth was observed, the effect was generally attributed to nutrient removals during the whole tree harvest. Some authors (e.g., Jacobson et al. 2000) allow that reduced growth may also be the indirect effect of altered microclimate and mineralization rates resulting from thinning, or to greater competition with ground vegetation. Where no growth reduction was measured, study authors suggested that atmospheric deposition of nutrients may have compensated for losses during the harvest.

Nutrient balance sheets: how much is removed from the existing reservoir?— Balance sheets are useful to compare nutrient inputs, outputs, and onsite reserves (Smith et al. 1986). We can estimate the amount of nutrients removed during whole tree fuel reduction by first taking stock of what is held in the crowns and boles and then asking how much material is removed. For example, canopy fuels in dense stands with high fire hazard conditions in the Western United States ranged from 9 to 21 Mg ha^{-1} (Reinhardt et al. 2006). Other extensive studies examining conifer crown fuel loads in the Western United States also fell within this range (Cruz et al. 2003, Fulé et al. 2001). With a few assumptions about the N content of the leaves and amount of crown removed, we determined that 65 to 150 kg N ha^{-1} might be removed as crown material during fuel treatment in these stands (fig. 33). Assuming an equal portion of N is stored in the wood, 65 to 150 kg N ha^{-1} may also be removed as bole material. For many U.S. forests, 130 to 300 kg N ha^{-1} represents only



[1] Mass of foliage and very fine branches (from Reinhardt et al. 2006).

[2] Foliar nitrogen (N) mass was determined assuming a foliar N content of 1.4 percent, which indicates an adequate N supply for many western conifers (Carter 1992, Garrison et al. 2000).

[3] Fuel treatment assumed to remove half the crown volume (e.g., Fulé et al. 2001).

[4] Nitrogen mass in boles assumed to be equal to N mass in crowns.

[5] Soil N storage in varying Holdridge life zones, based on a global analysis of thousands of soil profiles (Post et al. 1985).

Figure 33—Conceptual approach to estimate N losses resulting from whole tree fuel reduction thinning. To evaluate the impact of a single harvest removal, it is important to consider annual N deposition into the system, leaching out of the system, and the standing pool stored in the soil.

a few percent of the N stored in the soil, and even less of ecosystem N onsite. Of course, the actual amount of N removed will differ by stand and thinning treatment. Remember that N may be chronically added to forest stands by deposition, or lost by leaching, the levels of which vary by geographic region. By estimating local deposition or leaching rates, one can consider harvest N removals in terms of how quickly they will be replenished (table 8). Fertile sites with deep, rich soils are more resilient to whole tree harvests than poor sites, such as shallow soils over bedrock or coarse textured soils (Raulund-Rasmussen et al. 2008). More nutrients are generally exported during whole tree harvests from fertile sites compared to poor ones,

Table 8—Example soil nitrogen (N) capital and balance accounts for a site thinned by whole tree harvest

Credit/debit	N kg ha^{-1}	Explanation
Soil pool	9800	The soil acts as a large reservoir to store N. Most soil N occurs in organic form, which is not readily available for plant uptake and cannot be easily leached. Plant-available inorganic N is slowly released through decomposition and mineralization pathways.
Harvest export[a]	-200	Whole tree thinning removes some of the N capital from the site. The time required to replenish lost N depends on rates of inputs and outputs. Harvesting alters soil microclimate, which can increase or decrease the amount of N available for plant uptake by altering rates of decomposition and N mineralization.
Annual deposition	7	Nitrogen is continually added to terrestrial systems as both dry and wet deposition.
Annual leaching	-2	Nitrogen leaching losses typically occur as plant-available nitrate.

[a] The harvest removal represents less than 3 percent of the soil N pool. Assuming that deposition and leaching rates remain constant, N is added in this example at a rate of 5 kg ha^{-1} yr^{-1}. The N removed from the whole tree thinning treatment would be replenished in about 40 years, or sooner if the abundance of N-fixing vegetation increases after treatment.

but higher levels of nutrient inputs and cycling rates often allow for rapid replacement of the lost nutrients. Busse and Riegel (2005) estimated that whole tree fuel reduction in central Oregon removed 4 percent of ecosystem N, whereas bole-only harvest removed 1 percent. A whole tree clearcut in coastal Washington removed only about 5 percent of ecosystem N, compared to 3 percent for the bole-only harvest (Ares et al. 2007). In both these studies, differences in subsequent vegetative growth between bole-only or whole tree harvests were minimal.

Before developing a nutrient balance sheet, it can be useful to consider the greatest risk factors for nutrient loss from whole tree harvest: soil depth, site class, texture, and nutrient buffer. These are highlighted in the decision tool below (fig. 34).

Existing guidelines for whole tree harvesting and biomass or slash removal—Because of the increased focus on fuel treatments and removals of nonmerchantable biomass as an energy resource, a number of states have developed management guidelines or recommendations that address the impacts of forest removals on site nutrients, soil productivity, and long-term sustainability. Although biomass harvests may not be designed to reduce wildfire hazard, they often remove whole trees and slash from forest stands, and can lead to similar concerns regarding soil nutrient

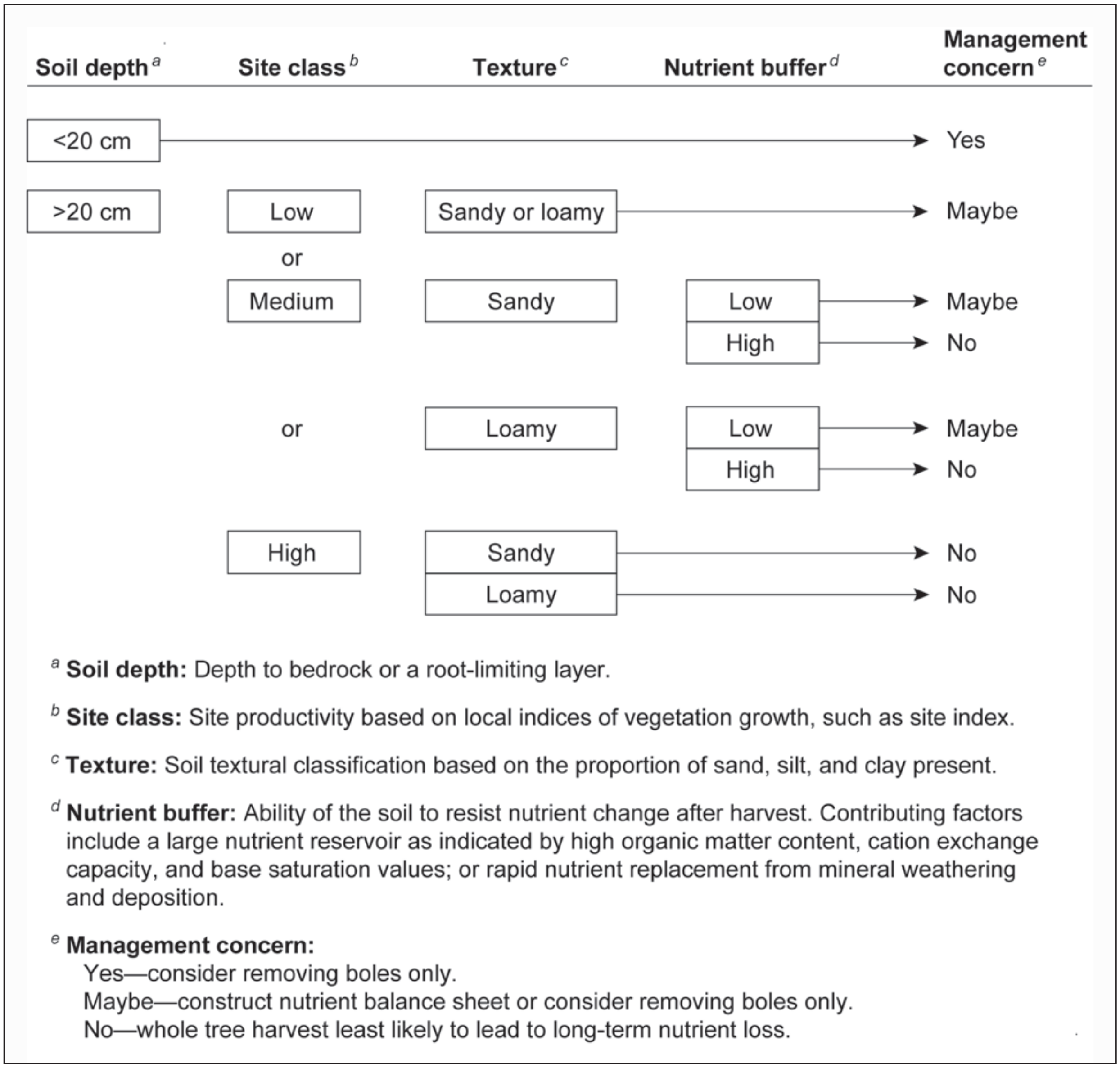


Figure 34—Whole tree harvest decision tool.

loss as whole tree harvested fuel treatments. Several states maintain guidelines for whole tree harvesting, and generally recommend avoiding whole tree methods only in shallow, nutrient-poor soils or in sensitive ecosystems (table 9).

Mitigating nutrient impacts from whole tree harvesting—
There are numerous options to compensate for nutrient losses that result from whole tree harvesting. More nutrients can be kept onsite by harvesting in the fall or winter. Wood becomes more brittle during this time and is more likely to break during thinning activities. Leaving broken branches or tops in the stand will reduce the

Table 9—State-level recommendations for whole tree harvesting and biomass or slash removal under differing forest conditions

Site condition	Management considerations and recommendations	State
General harvest sites	Current guidelines, allowing for slash and biomass removal, are deemed adequate to protect forest resources	Oregon[a]
Moderate to deep loamy soil development; good nutrient status	Bole-only thinning preferred, but whole tree probably okay	Montana[b]
Moderate to deep soil development; moderate nutrient status	Bole-only thinning recommended, but whole tree may be okay Select for trees with moderate to low nutrient demands	Montana[b]
Poor soil development with low-nutrient status	Bole-only thinning recommended, but whole tree may be okay for thinning from below or other light thinning Select for trees with low nutrient demands	Montana[b]
Aspen or hardwood on well-drained sandy soil, or Aspen or hardwood on shallow soil (≤ 20 cm) over bedrock	Avoid whole tree harvest Retain/redistribute slash Avoid short rotations Extend rotation age Fertilize	Minnesota[c]
Brush lands and open lands	Nutrients usually replenished within 10 years	Minnesota[c]
General harvest sites	Retain fine wood < 10 cm diameter created by incidental breakage Retain/scatter fine woody material from 10 percent of trees removed	Wisconsin[d]
Sensitive ecosystems or areas with high conservation needs (old growth, vernal pools, large bogs, seeps, caves), or Shallow soil (≤ 50 cm) over bedrock, or Dry, nutrient-poor sandy soil, or Wetland soil with ≥ 40 cm organic matter and are nutrient poor with low pH (dysic Histosols)	Do not remove fine woody material < 10 cm diameter	Wisconsin[d]
Low fertility site, shallow to bedrock soil, coarse sandy soil, poorly drained site, or erosion prone site	Minimize removal of fine woody material < 10 cm diameter	Maine[e]

[a] Oregon Department of Forestry (2008).
[b] Garrison-Johnston and Johnson (2006).
[c] Minnesota Forest Resources Council (2005).
[d] Herrick et al. (2009).
[e] Benjamin (2010).

nutrients exported offsite. Harvesting trees during their dormant period will also reduce nutrient exports, as leaves retranslocate their nutrients to the roots and other components at senescence (Nambiar and Fife 1991). Thinning deciduous trees after leaf drop will reduce foliar export out of the stand. Marking guidelines that account for species nutrient requirements can also help offset nutrient removals. Thinning trees with high nutritional needs while retaining more frugal species can reduce the overall nutrient demand in the residual stand. For example, Garrison-Johnston and Johnson (2006) ranked forest species in Montana from highest to lowest nutritional needs, but unfortunately did not provide information on the relative levels of nutrients required: grand fir > Douglas-fir > white pine > ponderosa pine > lodgepole pine > western larch. Where species exhibit great differences in nutrient needs, such rankings could become a consideration in marking guidelines for fuel reduction, in addition to crown spacing, shade tolerance, and fire behavior characteristics. Five years after whole tree thinning to reduce fuels and restore stand structure, units where trees were retained based on species preference had different levels of available N in the soil and total N in the forest floor compared with units where retention trees were based on size (Miesel et al. 2008). If these effects persist for decades, the nutrient differences could further affect stand productivity and resilience.

In many cases, whole trees are skidded to a landing where processors such as delimbers remove the nonmerchantable material from the bole. Rather than chipping or burning the slash, skidders could backhaul some or all of it into the unit to redistribute the nutrients onsite. This would allow for efficient harvesting equipment such as feller bunchers to fell trees, while reducing the nutrient losses. Rich (2004) addresses some operational constraints and practices to help make backhauling of slash a feasible option. Consideration should also be given to limit fuel loads in the redistributed slash, however.

An additional consideration to help prevent unsustainable mining of the site is to extend the reentry period between treatments to allow greater nutrient accumulation from deposition, N fixation, mineral weathering, and mineralization. In some cases, the best mitigation for nutrient removals may be to apply fertilizers. This requires knowledge of which nutrients should be replaced and in what proportions. In all cases, the reduction of nutrients from whole tree thinnings can be offset by practical management practices. These treatments should be site specific, based on the unique characteristics of each stand.

Soil Compaction

Key points:

Ecological effects	**Management considerations**
• Soils are easily compacted. Drastic changes in soil and site productivity may follow.	• Adhere to best management practices during fuel reduction harvesting: ▪ Keep machinery off moist soils (harvest when soils are dry or frozen). ▪ Use low ground pressure equipment if possible. ▪ Use designated skid trails or limit the number of random passes. ▪ Harvest on top of deep slash layers for high-risk soils. ▪ Consider subsoiling or scarification to restore compacted soils.
• One size does not fit all. Detrimental effects of compaction are site-specific.	• Identify high-risk soils: ▪ Clay and silt-sized particles are most prone to detrimental compaction. ▪ Rock fragments, soil organic matter, and forest floor cover help reduce compaction potential. ▪ Avoid harvesting on poorly drained soils when possible.
• Ten-year results from the Long-Term Soil Productivity study do not show clear evidence of detrimental changes in site productivity owing to severe compaction.	• Compaction can increase soil water availability on sandy soils, leading to improvements in vegetation growth. • Subboreal aspen was an exception to the rule—soil compaction substantially reduced stand biomass. • Confounding effects of soil mixing and soil displacement were not tested. • The results apply only to well-drained soils.

Few topics in forest soil science have received more attention than the effects of harvesting on soil compaction. More than a half century of research findings, anecdotal observations, and management intervention have been directed at understanding, assessing, and preventing soil compaction. From early findings that identified detrimental impacts of tractor logging on soil physical properties (Munns 1947, Steinbrenner and Gessel 1955) and associated tree growth (Froehlich 1981) to recent advances in logging systems that are more "soil friendly," considerable time and effort have gone into developing practical improvements for limiting soil damage. The list of critical functions at risk in compacted soil is great (e.g., soil aeration, water infiltration, erosion, nutrient and water availability, root growth, site productivity) and it underscores the prolonged attention given to the topic. Several key points have emerged during this period:

- Most soil types are easily compacted by machinery. However, predicting the extent of compaction or its effect on site productivity is quite difficult because of the extreme diversity of forest soils, vegetation types, climatic regimes, and logging systems. One size does not fit all.
- Soil moisture content at the time of harvest is a primary determinate of soil compaction. Countless studies have shown that soil moisture content near field capacity (i.e., wet, but not saturated) is most conducive to compaction (see Greacen and Sands 1980, Howard et al. 1981, McNabb et al. 2001). Saturated soils compact less but are at great risk of detrimental, deep rutting by harvesting equipment. Consequently, it is now common to include specifications or special clauses in harvesting contracts that require low to moderate soil moisture conditions (or frozen soil) before entry. Poorly drained soils, riparian soils, and soils that remain wet for much of the year (e.g., udic and perudic moisture regimes in the coastal Pacific Northwest and Southeast Coastal Plains) are particularly vulnerable to severe rutting and thus require additional preventative or mitigative measures—if harvesting is even appropriate—such as fully suspended logging, skyline harvesting, cut-to-length harvesting on top of thick slash beds, use of low ground pressure equipment with limited entry, or site preparation techniques such as bedding to mitigate damage where water tables are high.
- Soil compaction is substantially affected by relatively few passes by ground-based equipment (McNabb et al. 2001, Parker 2007, Williamson and Neilsen 2000). Because of this, the use (and reuse) of designated skid trails has been highly encouraged and practiced as an option for limiting the aerial extent of site disturbance.
- Use of low-ground-pressure equipment with wide tires or tracks that avoid slippage is well establish in literature and practice as a primary means to minimize soil damage.
- Heavy slash placed on the forest floor during cut-to-length harvesting or mastication operations can reduce soil compaction (Ampoorter et al. 2007, Moghaddas and Stephens 2008). However, the effect is not universal. Benefits to soil physical properties typically require deep slash-beds, large-

diameter slash, and relatively few passes by harvesting equipment (Han et al. 2009, McDonald and Seixas 1997).

- Soil texture (percentage of sand, silt, and clay) alone is not a good predictor of a soil's vulnerability to become compacted (Howard et al. 1981). Although silt-sized particles are most prone to compacting (Greacen and Sands 1980), complicating interactions with other soil properties (organic matter content, rock fragments, clay mineralogy, structure and drainage properties, initial bulk density, moisture content) prevent straightforward predictions.
- Soil recovery is generally a long-term process, particularly for compacted layers at soil depths of 10 to 15 cm and greater that develop beneath skid trails (Greacen and Sands 1980, Labelle and Jaeger 2011, Page-Dumroese et al. 2006). Surface layers may recover faster if soils are exposed to freeze-thaw cycles or if they contain shrink-swell clay.
- Subsoiling and scarification are effective at accelerating restoration of compacted sites, but are not always practical (e.g., wet or rocky sites) and may damage roots in the residual stand.
- The effect of compaction on site productivity is complicated. Tree-growth studies have shown short-term responses ranging from detrimental to neutral to positive (Ares et al. 2005, Froehlich et al. 1986, Gomez et al. 2002, Miller et al. 2001, Parker et al. 2007, Ponder et al. 2012). The net response depends on (1) the degree of compaction, as controlled by many soil properties and machinery specifications; (2) the aerial extent of compaction; (3) soil texture—e.g., compaction may improve soil water availability and vegetation growth in sandy soils; (4) soil resilience, or the time required for recovery; and (5) thinning effects on factors other than compaction such as root severing, bole scarring, and conifer release. As stated by McNabb et al. (2001), "the consequences of compaction on ecosystem processes…is ecosystem specific."

Lessons learned from fuel-reduction thinning studies—
Most information on soil compaction comes from studies of clearcut logging. However, there is a growing body of literature that addresses the effects of thinning on soil compaction (see Ampoorter et al. 2007, Labelle and Jaeger 2011, Landsberg et al. 2003, Moghaddas and Stephens 2007, Parker 2007, Parker et al. 2007, Sidle and Drlica 1981). Drawing an overall conclusion from these studies is difficult, however, because of the variety of harvesting methods and equipment, thinning prescriptions, soils, and techniques used to measure compaction. For example, Moghaddas and Stephens (2008) found that commercial harvesting by chain saw

felling and grapple skidding followed by mastication of understory trees resulted in relatively minor changes in soil properties, whereas Parker (2007) found sharp increases in compaction when testing a variety of harvesters and grapple skidders. Landsberg et al. (2003) also tested a variety of ground-based equipment in an operational thinning study and found a wide range in degree of compaction and surface disturbance, from 17 to 57 percent on an aerial basis, yet they were unable to attribute the differences in compaction to specific equipment types.

Ground-based thinning operations compact soil and result in accompanying soil mixing and displacement when trees are skidded to landings. This is accepted knowledge. A more immediate concern then is whether such changes are detrimental to site productivity. Of relevance here, Parker et al. (2007) examined site productivity on thinned plots that were either compacted (random grapple skidding) or not (no equipment traffic). He found that compaction led to reduced growth of individual trees within 9 m of the most severely affected ground (soil strength about 2,500 kPa). However, when scaled up to the plot level, there was no significant decline in total tree growth or site productivity in 17 years following thinning (Busse et al. 2009a, Parker 2007). The compacted zones in effect served as "hotspots" of reduced tree growth within a much larger unaffected area. This finding suggests that site productivity can be maintained if the aerial extent of severe compaction is kept to a minimum, or, alternatively, if a sufficient network of old root channels, shrinkage cracks, or soil faunal activity exists within the compacted zone to allow for root proliferation (Batey 2009, Nambiar and Sands 1992). Recent findings from the Long-Term Soil Productivity study (see below) support this conclusion, as numerous research sites throughout Canada and the United States showed minimal change in forest productivity even when nearly 100 percent of the ground surface was compacted.

Gaps exist in our knowledge of soil compaction and its effect on site productivity and function. Until the gaps are filled, Parker (2007) outlined several practical, simple steps for fuel reduction thinning practices frequently used by managers.

- Establish realistic, obtainable harvesting goals that address concerns for the local soils and their susceptibility to compaction using the best available science and local experience. Avoid applying regional generalizations about soils and management criteria.
- Include contract specifications and special clauses for acceptable limits of compaction, along with incentives for meeting contract requirements and penalties for noncompliance. This includes selection of appropriate harvesting methods (e.g., cut-to-length harvesting, whole tree harvesting), soil moisture restrictions, and maximum ground pressure that best meet all

environmental, operational, and economic goals.

- Meet on the site with all personnel involved before the thinning operation begins to discuss project goals, concerns, soils issues, and contract specifications.
- Use designated skid trails (echoed by most studies).
- Establish a plan for contract inspectors to repeatedly monitor progress onsite.

Does soil compaction alter site productivity? Recent results from the North American Long-Term Soil Productivity Study—
The Long-Term Soil Productivity (LTSP) program is a large-scale research network of more than 100 forested sites throughout the United States and Canada that examines the short- and long-term effects of soil compaction and surface organic matter retention on site productivity (Powers 2006). Common LTSP treatments, including combinations of soil compaction (none, moderate, severe) and forest floor mass (high, medium, none), were installed following clearcut harvesting of residual forest stands beginning with the first site installation in Louisiana in 1990. The collective findings from the study offer an insightful and somewhat surprising view of compaction and its impact on site productivity. Among the key findings are that soil compaction (1) improved productivity and water-holding capacity of sandy soils in summer-dry climates (Gomez et al. 2002), (2) dramatically reduced productivity of aspen stands regenerating naturally from root suckers (Ponder et al. 2012), and, (3) when averaged across the entire network of sites using meta-analysis procedures, resulted in greater tree growth compared to uncompacted controls (Ponder et al. 2012).

A brief background is helpful to understand these findings. Among the original objectives of the LTSP study were to determine (1) if changes in soil porosity (compaction) have a lasting effect on forest vegetation, (2) whether compaction effects differ by climatic zone or soil type, and (3) whether detrimental impacts, when found, are reversible with time (Powers 2006). All LTSP sites were clearcut harvested, then treatments were applied to 0.5-ha plots prior to planting tree seedlings (or natural regeneration in the case of aspen). Logs were fully suspended during harvesting to ensure that compaction, soil displacement, and soil mixing did not occur prior to treatment. Compaction treatments were applied by repeated driving on the soil surface with heavy equipment (e.g., asphalt roller, bulldozer, grappler depending on local availability and preference) or by compressing with high ground pressure equipment uniformly across entire plots (Page-Dumroese et al. 2006b). No soil displacement or mixing occurred, and thus the results are specific to the effects of soil compaction alone. Severe compaction was targeted at 80 percent of the restrictive bulk density or soil strength for root penetration at each site. Moderate compaction approached the midpoint between severe and no compaction. Interestingly, soils at the 12 LTSP sites throughout California could not be compacted without first removing all surface organics down to bare soil (the forest floor was later replaced on the soil surface), demonstrating the difficulty of compacting some soils.

The LTSP sites span boreal, temperate, Mediterranean, and subtropical climates (fig. 35). Corresponding forest types range from black spruce (*Picea mariana* Mill.), jack pine (*Pinus banksiana* Lamb.), aspen (*Populus tremuloides* Michx.), mixed conifer, ponderosa pine, mixed hardwood–pine, and southern pine–hardwoods, with an equally diverse range in soil parent material, soil texture, soil organic matter content, and site productivity (Ponder et al. 2012, Powers 2006).

In summarizing the LTSP results at year 10, Ponder et al. (2012) stated, “Perhaps the most notable and unexpected result to date is the marked planted conifer growth response to compaction.” Essentially they found positive responses in tree biomass on severely compacted compared to uncompacted plots in their analysis of 46 LTSP installations. Biomass gains were particularly significant at several sites in the southeast and in California. Exceptions to these observations were a strong decline in biomass production for aspen sites in subboreal climates and a moderate decline in productivity for a mixed-conifer site in California growing in a moderately fine-textured soil.

Ponder et al. (2012) offered several explanations for these findings that have relevance to the discussion of fuel reduction thinning and site productivity. Foremost, the majority of LTSP sites are located on well-drained soils. Thus, any negative effects on soil aeration, anaerobic conditions, or nutrient supply that are common in poorly drained soils were presumably inconsequential. As already mentioned, poorly drained or wet soils are a particular concern during thinning operations and require special consideration to avoid soil damage. In addition, about 60 percent of the LTSP sites have sandy to coarse loamy soils, which likely benefitted from improved water-holding capacity when compacted. The authors also reiterated that the results were valid for compaction forces without any additive effects of soil

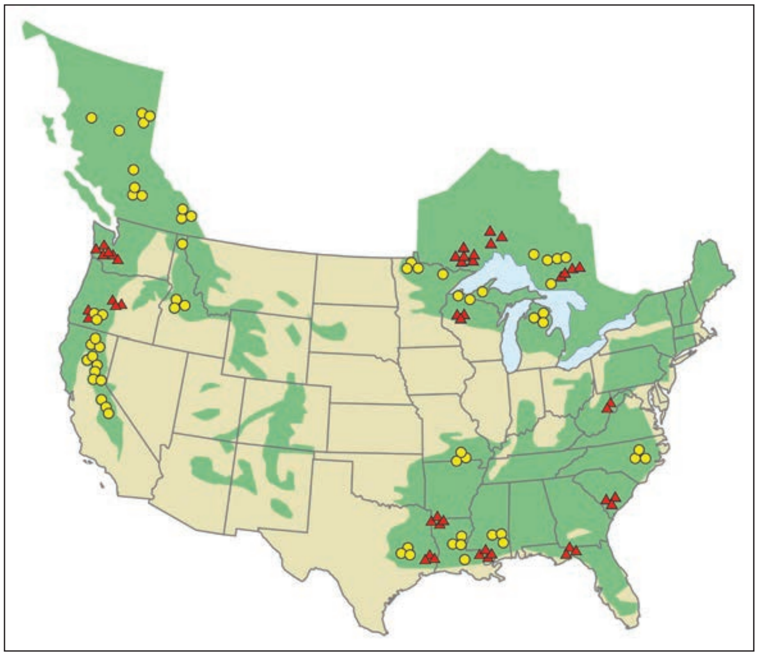


Figure 35—Location of Long-Term Soil Productivity installations. Core sites (yellow circles) have the complete combination of compaction and organic matter treatments; affiliated sites (red triangles) lack some treatments, although most include the soil compaction.

displacement, mixing, or rutting, and that the use of skidding equipment should increase both the intensity and aerial extent of compaction (Han et al. 2009). A final caveat pointed out by the authors was that the findings reflect the short-term response by vegetation to compaction. Whether any long-term losses in site productivity will result are yet to be determined.

Masticated Fuels—New Practices, New Concerns?

Key points:

Ecological effects	Management considerations
• What is vertical becomes horizontal—canopy, ladder, and standing surface fuels are rearranged into woody surface fuel bed dominated by small (1- to 100-hour) fuel classes.	• There is an initial increase in potential surface fire behavior until masticated residues settle and decay. • Nutrients are retained onsite.
• Soil disturbance and compaction can be minimized during mastication.	• Use a boom-mounted rather than rigid-mounted cutting head. • Use low-ground-pressure equipment. • Operate when soils are dry, frozen, or under deep snow. • Limit equipment to designated trails.
• Masticated residues act as a mulch layer. ▪ Soil moisture retention is increased. ▪ Soil heating is reduced. ▪ Seedling establishment may be inhibited extending fuel treatment longevity.	• The net mulch effect is difficult to predict as it depends not only on the residue layer depth and compactness, but on the opposing effects of increased solar radiation and air temperature that result following thinning.
• Short-term microbial response is relatively insensitive to mastication.	• Based on existing knowledge, short-term impacts to microbial communities and processes are not primary drivers for management decisions regarding mastication. • Long-term studies may reveal true biological implications.
• Contrary to expectations, nitrogen (N) immobilization has not been observed following mastication.	• Concerns that mastication will reduce N availability and site productivity are not substantiated by existing research.

There are so few studies of mastication impacts on soil resources that it is difficult to interpret long-term ecological consequences. There is no natural analogue to mastication. The masticated debris is unlike the natural forest floor in terms of particle size, composition, bulk density, and moisture regime, so there are few easy comparisons to natural wildland systems or processes. Short-term studies have assessed mastication impacts on limited physical, chemical, and biological properties of soils, and found few detrimental effects. These are reviewed and discussed in the following sections. Long-term consequences or indirect effects from mastication remain largely unstudied.

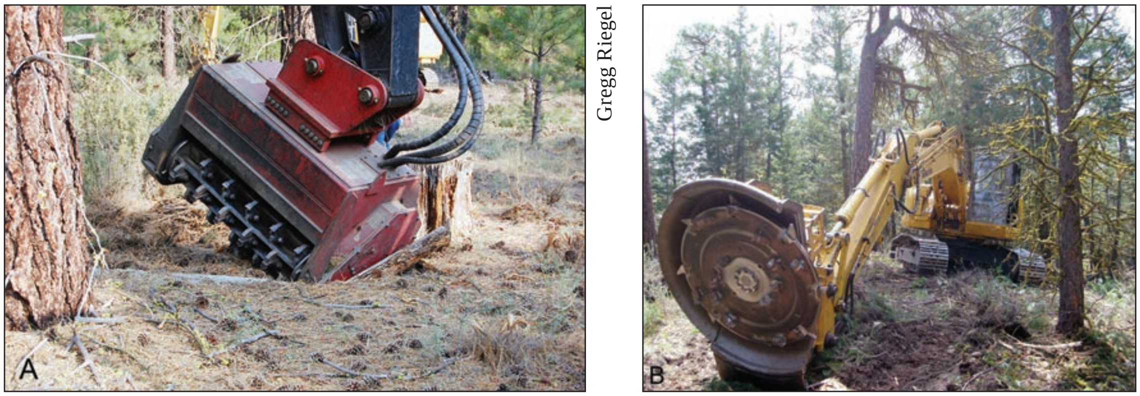
What is mastication?—

Mastication is a mechanical fuel treatment applied to shred, grind, mulch, mow, or chunk woody understory vegetation and small trees. The resulting material is typically broadcast away from the machine as it operates and left onsite. As a fuel treatment, mastication serves to convert canopy, ladder, and standing surface fuels into a woody surface fuel bed—essentially rearranging fuels from vertical to horizontal by shredding and dispersing them on the forest floor.

Mastication equipment is quite variable, and usually consists of a cutting head that rotates on either a vertical or horizontal shaft (fig. 36) (Harrod et al. 2009). The cutting head can be mounted on an excavator, tractor, skid steer, all-surface vehicle, or other heavy equipment, for use in wildland environments. Boom-mounted cutting heads allow the operator to reach up to 9 m in any direction, thereby reducing the need for equipment turns and limiting the associated ground disturbance (Harrod et al. 2009). Rigidly mounted cutting heads require more equipment maneuvering to process fuels, potentially resulting in greater soil disturbance (Coulter et al. 2002).

Physical characteristics of masticated fuel beds: particle size, depth, fuel load—

Characteristics of a masticated fuel bed will differ depending on the project objectives; equipment used; treatment intensity; and the size, age and amount of the fuels (fig. 37) (Kane et al. 2006). As a result, the size of masticated particles may differ from a predominance of fuels in the 1- and 10-hour time lag classes (≤0.64 cm and >0.64 cm to ≤2.54 cm diameter, respectively) (Kane et al. 2006), to predominance of 10- and 100-hour fuels (>2.54 to ≤7.64 cm diameter) (Reiner et al. 2009), to football- or basketball-size chunks (Jain et al. 2008). Masticated biomass is often dispersed unevenly across a site, with some areas receiving deep accumulations of materials while other areas remain unaffected. For example, percentage of cover of residues averaged 8 percent and 17 percent in two ponderosa pine-dominated mastication treatments in the southern Sierra Nevada (Reiner et al. 2009). Harrod (2009) stated that masticated materials can add as much as 5 cm to the forest floor. Maximum depths reported for masticated materials range from 7 cm in a pinyon-juniper woodland (Owen et al. 2009) to 24 cm from a dense shrub understory within a ponderosa pine plantation (Busse et al. 2005), but average depths are typically from 3 to 7 cm. Fuel loadings reported in the literature for masticated sites range from 15 Mg ha^{-1} (7 tons ac^{-1}) of woody fuels (Kane et al. 2006) to 78 Mg ha^{-1} (35 tons ac^{-1}) of masticated biomass (Busse et al. 2005), both at northern California sites. Over time, the masticated layer will settle and decay, and decrease in both depth and mass. Chips, which tend to be more uniform than masticated materials, were found



Gregg Riegel

Richy Harrod

Figure 36—Orientation of masticator cutting heads. Many more examples are available in equipment guides and catalogs. (A) horizontal shaft, (B) vertical shaft.

Eric Knapp

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Figure 37—Masticated debris ranges in size and continuity depending on the vegetation, cutting head, and time spent working the material. (A) fine debris, (B) coarse debris, (C) masticated brush.

to settle 18 percent during the first year after a chipping operation of ponderosa pine biomass in Colorado (Wolk and Rocca 2009).

Masticating without displacing and compacting the soil—
Ground disturbance to the forest floor and mineral soil will depend on the vehicle specifications, soil conditions, and operator experience. Masticator cutting heads requiring higher machine power necessitate the use of heavier equipment (Harrod et al. 2009). Depending on the type of machine used and operating conditions, maneuvers such as turning can displace and churn the organic and mineral horizons. Mastication trials to treat fuels in the Northwest used nine different equipment scenarios at four sites, with vehicles ranging from a 3900-kg (8,500-lb) all-surface vehicle to an 36 000-kg (80,000-lb) excavator to power a variety of cutting heads (Coulter et al. 2002). At each trial, soil disturbance was gauged using a visual assessment method. The greatest soil disturbance was caused by operations designed

to incorporate masticated materials into the soil. Other operations caused soil disturbance to increase from "slight," prior to treatment, to "some" or "moderate" following treatment. In these cases, which included the 36 000-kg excavator, soil conditions changed from "virtually undisturbed" to showing some displacement of the forest floor and evidence of compaction such as platiness or lack of structure. At three sites, no increases in soil disturbance were recorded, owing to operations over deep snow or recent site disturbance from forest thinning that was not increased by the mastication activities.

Compaction resulting from mastication operations can be largely curtailed by selecting appropriate equipment and site-specific operating conditions. Limiting masticators to designated trails or using low-ground-pressure equipment can reduce the extent and intensity of physical soil disturbance. The British Columbia Ministry of Forests and Range (2008) defines low ground pressure as equipment that exerts less than 43.4 kPa (6.3 pounds per square inch [psi]) of total ground pressure. Thinning followed by mastication of mixed-conifer forest in the Sierra Nevada with a 31-kPa (4.5-psi) masticator did not increase the extent of detrimental compaction in heavily managed stands (Moghaddas and Stephens 2008). The boom-mounted masticator was not limited to designated trails, but compaction may have been limited by the cushioning effect of the masticated debris bed. Hatchett et al. (2006) measured few compaction impacts in a masticated fir stand of the Tahoe basin in the Sierra Nevada, after treatment with a 37.9-kPa (5.5-psi) masticator. Any compaction from the operations was limited to a narrow range of soil depth, and detectable only when the soil beneath the masticator travel path was compared with areas far from the machine traffic and were much less disturbed. As in the Sierra Nevada studies, mastication fuel treatments in a Colorado pinyon-juniper woodland did not cause soil compaction (Owen et al. 2009). Although machine specifications were not provided, operations were conducted over snow and frozen ground, which help resist compaction by heavy equipment.

These few studies suggest that compaction can be avoided during mastication operations, but, in fact, little research has been dedicated to this topic. The impacts of heavy equipment on skid trail compaction have been demonstrated for more than 50 years (Dyrness 1965, Steinbrenner and Gessel 1955), but many mastication fuel treatments are fundamentally different than extractive forestry practices. During harvests, impacts tend to be concentrated on trails that receive multiple passes as materials are transported to landings. In contrast, a masticator may track over broad areas to treat fuels, particularly if the cutting head is rigidly mounted rather than boom-mounted. Consequently, a large proportion of the treated area may be subject to a single machine pass. The masticated fuel bed can buffer against soil compaction, similar to a slash mat in cut-to-length harvest treatments, but the debris depth required to offset various levels of machine ground pressure for a variety of soil moisture or texture conditions is not known. Prudence dictates use of low-ground-pressure equipment, particularly on sensitive sites. Windel and Bradshaw (2000) catalogued many equipment options for fuel treatment, including very low-ground-pressure (less than 20.7-kPa or 3.0-psi) machines.

Mastication can substantially modify soil temperature and moisture regimes—Mastication activities leave a residue of woody debris that acts as mulch on the forest floor. The woody mulch insulates against heat gain and moisture fluctuations in the soil. A variety of bark and chip mulches reduced the soil temperature and increased moisture during the growing season in a maple plantation (Iles and Dosmann 1999). Under masticated residues 10 cm deep, Massman et al. (2006) also recorded cooler soil temperatures at 15-cm depth than in control plots. They found that during summer months, the mulch reduced radiant heat gain in the soil and temperatures were about 1 to 9 °C cooler than controls. In late fall and early winter, however, the mulch served to impede cooling, and soil temperatures remained up to 4 °C higher than controls. A different summer mulch effect was observed in a Sierra Nevada study, where Busse (unpublished) measured maximum daily soil temperatures about 5 °C higher for surface-applied masticated residues compared to the control (fig. 38). In this case, the mastication treatment effectively opened the stand and resulted in greater solar radiation and air temperatures at the forest floor surface compared to the densely vegetated control plots. Incorporating masticated fuels into the soil increased soil temperatures. In a study of Rocky Mountain Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), Gower et al. (1992) observed that wood chips applied to the forest floor significantly increased forest floor moisture content, and that the layer of wood chips remained frozen as much as 2 weeks longer than the natural forest floor. While rainfall patterns and woody mulch depth play important roles in soil moisture dynamics, masticated debris acts as a barrier against both water infiltration into the soil and evaporative losses from the soil (Massman et al. 2006).

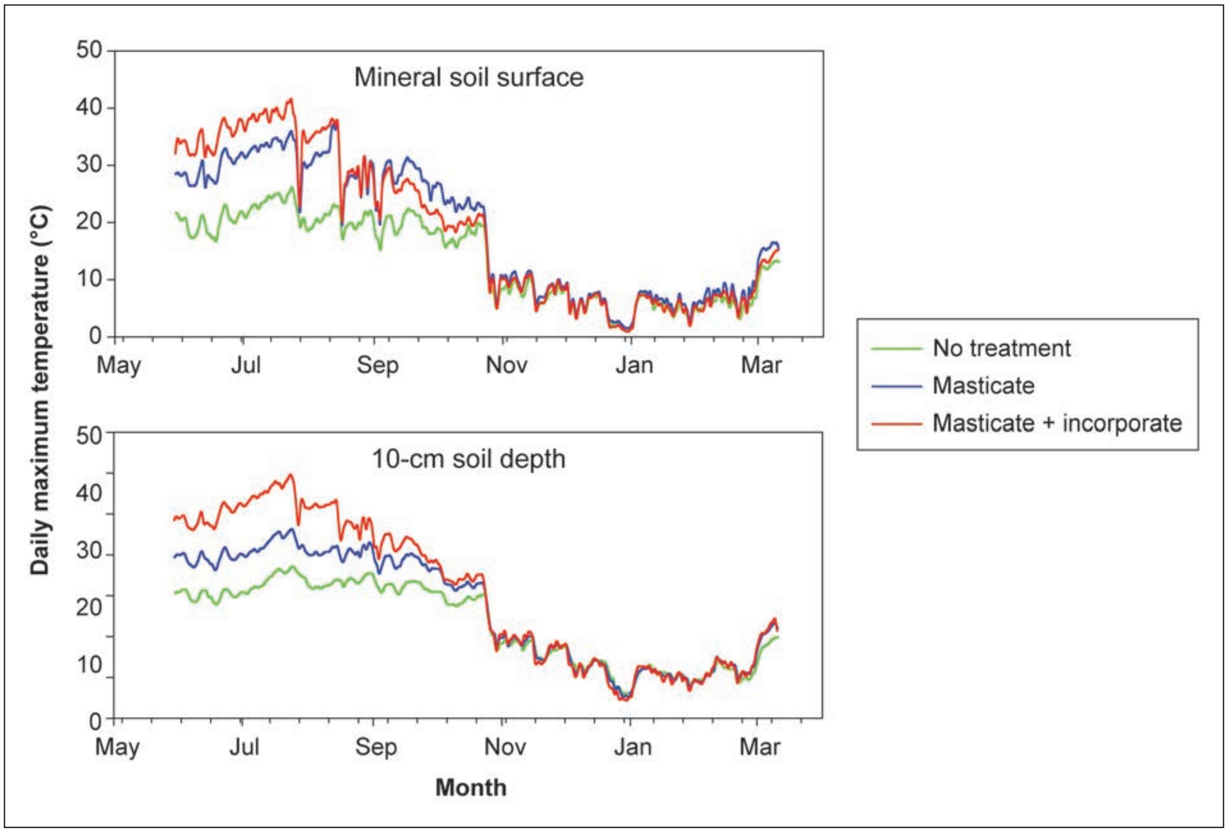

Figure 38—Soil temperature differences among stands of untreated control areas, masticated understory, and masticated understory that has been incorporated into surface soil (Busse, unpublished data).

Mastication has limited short-term effects on soil microbial communities—Forest thinning can allow more solar radiation to the forest floor, causing soil heating. This may enhance microbial activity in the spring, when soils are moist. This effect will be more pronounced in northern forests compared to subtropical or tropical areas closer to the equator. In areas that experience summer drought, thinning can also hinder microbial populations by drying soils earlier in the season. Thinning by mastication, however, adds a woody layer that can act as a mulch, buffering soil heating and moisture changes, with potential impacts to soil microbial processes (Miller and Seastedt 2009). Despite increased substrate inputs following mastication in a ponderosa pine plantation, Kobziar and Stephens (2006) did not detect any resulting changes in mean annual soil respiration rates. They surmise that mastication may have reduced root respiration by shredding vegetation, and that heterotrophic respiration from soil organisms may have increased to make up the difference, but this remains unconfirmed. In a study focused on soil carbon sequestration, Busse et al. (2009b) found fungal and bacterial biomass responses relatively insensitive to forest mastication and mastication followed by incorporation of residues into the mineral soil. They attribute this result to the influence of residue fragment size and differences in decay rates at their study sites. Owen et al. (2009) studied the response of arbuscular mycorrhizal fungi to mastication thinning in a pinyon-juniper woodland. Treatments were designed to reduce overstory 40 to 60 percent, but the masticated mulch served to reduce temperatures and increase moisture in the soil. Despite these environmental changes, mastication did not result in differences in abundance, species richness, or community composition of arbuscular mycorrhizal fungi 2.5 years after treatment.

Soil nitrogen not always immobilized after mastication—
Mastication and the associated addition of woody residues are commonly thought to reduce N availability. By adding mulch that is generally low in N and high in C, such as woody fragments, it is often speculated that microbes will assimilate available inorganic N from the soil in order to decompose the added material. If true, this N immobilization could reduce the amount of soil N available for plant growth. Few studies have examined N transformations and dynamics following mastication, however. In a Sierra Nevada comparison of fuel treatments, commercial thinning followed by mastication did not significantly alter available N or net nitrification rates 2 years after treatment, compared to untreated control stands (Moghaddas and Stephens 2007). In a pinyon-juniper mastication study, Owen et al. (2009) unexpectedly found increased available N (ammonium) 2.5 years after mastication. Similarly, Miller and Seastedt (2009) found no change in N availability in the first 2 years after thinning and wood chip application in a Colorado pine stand, but a 33-percent increase in available soil N in the third growing season. Contrary to common expectations, these studies did not measure N immobilization following mastication treatments.

Mastication is a relatively recent and novel fuel reduction practice. Although many thousands of hectares are being treated with mastication across the United States, particularly in the WUI, relatively little is known about its short- or long-term impacts to soils. Most research examining mastication effects on soils has been conducted in the Western United States, and in California in particular.

Masticated Fuels—To Burn or Not To Burn?

Key points:

Ecological effects	**Management considerations**
• Expect greater fire intensity and aboveground severity than burns in unmasticated stands.	• Allowing masticated residues to settle and compact for 1 or more years can reduce fire behavior. • A prolonged waiting period may also allow emerging understory vegetation to accumulate, increasing the burn effectiveness at reducing surface fuels.
• Soil heating can kill plant roots and soil organisms, but burn prescriptions can limit maximum soil temperatures.	• Burning when soil is moist (at least 20 percent moisture by volume) will limit most soil heating to the top 2 to 5 cm. • The deeper the masticated residues, the more likely heat will penetrate further into the soil. • Spatial variability in residue depth will lead to a range in soil heating, from areas of possibly no heating to hotspots below deep accumulations. • Soil texture is unlikely to affect depth of soil heating.
• Burning masticated residues does not mean more bare mineral, soil.	• Burn prescriptions can be designed to result in similar levels of bare ground for masticated fuel beds or natural fuels.
• Soil nitrogen (N) content and transformations are similar in burned stands, with and without prior mastication.	• High variability in fuel consumption and N movement after burning will produce variable results.

Along with mastication operations, many fuel reduction projects also propose to use prescribed fire as a followup treatment. Mastication can effectively reduce ladder fuels and height to the live crown base in forested stands (Stephens and Moghaddas 2005), which enhances the feasibility of prescribed fire operations. Masticated fuel beds differ from natural fuel beds in their moisture and temperature regimes (Massman et al. 2006), surface area to volume ratios (Kreye and Varner 2007), particle shape, and size class composition (Kane et al. 2009), making fire behavior and effects in masticated residues difficult to predict. For example, Knapp et al. (2011) observed that fire-severity measures in northern California masticated fuel beds exceeded model predictions by nearly a factor of four. Because mastication redistributes large fuels into finer classes, there is potential for more extreme fire behavior immediately after the treatment. For example, a study of prescribed fire in the Klamath Mountains of northern California found that mastication increased both fire intensity and aboveground severity compared to nonmasticated plots (Bradley et al. 2006).

Burning masticated residues with limited soil heating—
Despite the potential for increased fire effects in masticated materials, several northern California studies have shown limited soil heating during prescribed fire in masticated stands by burning residues during spring, when soils are moist. Bradley et al. (2006) found that despite greater fire behavior measures in their masticated stand, no changes in heating at the soil surface were detected between masticated and nonmasticated prescribed fire plots. These spring burns were conducted 6 months after mastication, and the loosely arranged fuel bed was considered a contributing factor to fire behavior. It was expected that, over time, the masticated residues would settle and compact, potentially lowering fire intensity. Similarly, Knapp et al. (2011) observed long flaming duration and high heat content of the masticated woody fuel bed, but soil temperatures at 5-cm depth remained below 60 °C at nearly all thermocouple locations. In this study, prescribed fires were conducted in late spring, 2 to 3 years after mastication (fig. 39). Several studies suggest that over time, masticated residues will compact, leading to reduced fire behavior (Knapp et al. 2011, Kobziar et al. 2009).

Constructed fuel beds burned under controlled moisture conditions can allow researchers to develop guidelines for soil conditions or fuel loadings that may limit soil heating. Busse et al. (2005, 2010) conducted several studies on soil heating in which the masticated depth, fuel load, soil moisture, or soil texture were controlled


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Figure 39—Minimal soil heating occurred during burning of these masticated fuels in a study reported by Knapp et al. (2011).

in constructed test plots to determine effects on the depth of lethal temperatures. They found that soil moisture is the primary factor in limiting soil heating. In dry soils (4 percent volumetric moisture), temperatures exceeded 60 °C from the surface to 10 cm soil depth when the mulch layer was deeper than 2.5 cm. Moist soils (16 percent volumetric moisture), however, required a very deep mulch layer (12.5 cm) to cause lethal soil heating below 2.5-cm soil depth (Busse et al. 2005). Additionally, burning when volumetric soil moisture exceeds 20 percent (common during moist spring conditions) should limit damaging soil heating to the top 2.5 to 5 cm soil depth (Busse et al. 2010). Although differences in soil texture can affect heat transfer, temperature trials in sand, loam, and clay soils did not respond differently to prescribed fire treatments (Busse et al. 2010).

Within a treatment unit, masticated fuels are highly variable in depth, fuel loading, and continuity. This heterogeneity can result in patchy prescribed fires with spatially variable soil heating effects. A wide range of soil heating was measured during fall burns of masticated units in the Cascade Range of northern Washington (Harrod et al. 2008). These were implemented within 6 months of mastication, and took place during cool, moist conditions. As a result, fire intensity was relatively low, with flame lengths generally <1 m. Although many thermocouples did not record any temperature increase resulting from the patchy fires, the duration of lethal heating extended as long as 10 hours at some locations.

Burning masticated stands does not expose more bare soil than untreated areas—

Compared to untreated areas, areas treated with mastication increase in amount and continuity of fuels on the forest floor, which can lead to increased fire intensity, but this does not necessarily mean that more bare mineral soil will be exposed during prescribed fire operations. When comparing fuel treatments including prescribed fire and mastication followed by prescribed fire, Moghaddas and Stephens (2007) and Kobziar (2007) both found no statistical difference in the amount of bare soil exposed after burning. Fuel moisture plays a large role in fuel consumption and subsequent soil exposure. Prescribed burning of masticated fuels during the typical fire season, or summer drought period, resulted in 49 percent (Kane et al. 2010) and 56 percent (Moghaddas and Stephens 2007) cover of bare soil. Following 2 cm of precipitation, bare soil exposure was less than 3 percent (Kobziar 2007). Burning masticated residues during moist spring conditions led to only modest duff consumption, where less than 30 percent was consumed (Bradley et al. 2006).

Prescribed fire yields similar nutrient and microbial responses in masticated and nonmasticated stands—
Fires rapidly oxidize and decompose organic matter, releasing nutrients such as N in available, inorganic form. However, in a comparative study of fuel treatments in the Sierra Nevada, Moghaddas and Stephens (2007) found no significant differences in inorganic N concentrations (ammonium or nitrate), net N mineralization, or nitrification rates between burned stands with or without prior thinning and mastication. High within-treatment variability in fuel consumption and nutrient movement likely influenced the extent and intensity of these N alterations.

In another Sierra Nevada study, Kobziar (2007) measured soil respiration rates in pine plantations that had been burned, with and without mastication. The relative contribution of heterotrophic respiration (from soil organisms) increased in both types of treatments. This appeared linked to a decline in the importance of root (autotrophic) respiration, owing to root mortality and reduced photosynthetic capacity caused by the prescribed burning. The two prescribed fire treatments did not clearly distinguish themselves in terms of heterotrophic versus autotrophic respiration rates.

Thinning and Burning—Early Results From Long-Term Study Sites

Key points:

Ecological effects	Management considerations
• Combinations of mechanical thinning and prescribed fire applied in a variety of U.S. forests using best management practices produced modest short-term changes in soil properties.	• Mechanical removal of 20 to 45 percent basal area exposed more bare soil and increased bulk density at a few study locations, but did not substantially alter soil quality across the national network of sites. • Prescribed fire effects were limited to short-term reductions in forest floor mass and a modest pulse of available nitrogen that increased with fire severity. • Combined treatments have not substantially affected soil quality any differently than have single treatments yet. • These are early findings from a long-term national study. Caution in extending the results beyond the current treatment longevity and range of sites is advised.

Mechanical fuel treatments allow managers to alter forest structure and emulate the effects of fire on the size and species distribution within a stand. Silvicultural treatments, however, may not mimic the ecological functions driven by fire. To address this issue, the Fire and Fire Surrogate (FFS) study compares prescribed fire treatments with mechanical "fire surrogates" across a national network of sites, utilizing a common study design and set of response variables. Study locations represent forests with hazardous fuel loads, but historically short-interval, low- to moderate-severity fire regimes. Sites include western coniferous forests, ranging from the Pacific Northwest to the Southwest, as well as Appalachian hardwood, Piedmont pine, and southern pine forests of the Coastal Plain. Control plots and three manipulative fuel treatments were established at the sites: mechanical treatment, prescribed fire, and combined mechanical and prescribed fire treatments.

Site conditions and treatment intensities were quite variable among FFS sites. For example, helicopters yarded logs in Washington, trees were whole tree harvested in northern California, nonmerchantable trees and brush were masticated at another California site, and mowers and choppers mechanically treated fuels in Florida. Prescribed fire treatments were also highly variable. On average, burn treatments in a Rocky Mountain site in Montana consumed 12 percent of the duff layer (Gundale et al. 2005), whereas those in a California Sierra Nevada locale reduced forest floor mass and depth by more than 80 percent (Moghaddas and Stephens 2007). Prescribed fire was typically applied following mechanical operations, but to meet fuel reduction objectives in the Florida flatwoods, stands were first burned then mechanically chopped (Onokpise et al. 2007). Despite these broad differences in treatment implementation, soil responses in the initial 2 to 3 years after treatment were generally similar across the FFS sites.

In a meta-analysis of 12 sites, Boerner et al. (2009) reported that FFS treatments resulted in only minor, ephemeral impacts to soil physical, chemical, and biological properties compared to the control treatment (table 10). Treatments involving fire significantly increased bare mineral soil. In the first year following treatment, bare soil ranged from 0 to 46 percent cover for prescribed fire only and from 0 to 56 percent cover for mechanical thinning plus fire (Boerner et al. 2009, Moghaddas and Stephens 2007). After 1 to 2 years of litter fall, bare soil exposure at all reporting sites and treatments was less than 10 percent cover (Boerner et al. 2009). Although bare soil facilitates germination of many pines, shrubs, and herbaceous species, there was no clear treatment effect on seedling density across the FFS network (Schwilk et al. 2009). At some study sites, more seedlings were lost as a result of the treatments than were gained through germination. At a Sierra Nevada site where mineral soil exposure exceeded 50 percent, the fire treatments, both with

Table 10—Short-term impacts of the Fire and Fire Surrogate treatments on soil properties

Soil property	Mechanical	Fire	Mechanical + fire	Notes
Bare mineral soil	—	↑	↑	Greater fuel consumption led to greater soil exposure. In year 1, bare mineral soil exceeded 15 percent at three of eight sites. Exposed soil was <10 percent 1 to 2 years later.
Bulk density	—	—	—	Operations highly variable, including conventional harvest, whole tree harvest, helicopter yarding, and mastication.
pH	—	—	↑	Significant pH increases were transient, and limited to sites with greatest fuel consumption.
Available calcium (Ca), magnesium, potassium	—	—	—	No significant changes at network scale, but Ca was increased in Oregon 1 year after treatment, and in Oregon and Ohio 1 to 2 years later.
Available Phosphorus	—	—	—	No significant changes at network or individual site scale.
Total soil nitrogen (N)	—	—	—	No significant changes at network or individual site scale.
Total inorganic N (TIN)	↑	↑	↑	On average, TIN increased by 1 to 6 kg ha^{-1} in year 1, but changes were not significant 1 to 2 years later.
Rate of N mineralization	—	—	—	Greater variation in N mineralization at western sites compared to eastern sites.
Soil carbon (C)	—	—	—	Most soil C impacts are limited to top few centimeters. By looking at top 15 cm, this study did not measure any significant soil C changes.
Soil C:N ratio	—	—	—	Modest effects on C and N led to nonsignificant changes in C:N ratio at network scale.
Forest floor C	—	↓	↓	Prescribed fire consumes C-rich surface litter.

"—" indicates no significant change in effect size relative to control treatment at network scale.
"↑" indicates significant increase in effect size relative to control treatment at network scale.

and without mechanical operations, increased the density of Douglas-fir seedlings (Moghaddas et al. 2008). Despite this increase, no significant differences in understory plant abundance were detected among treatments (Collins et al. 2007). The FFS treatments did reduce species richness of native plants relative to the control in the first year at this site, but the decline was modest, about one fewer species on average (Collins et al. 2007).

The meta-analysis also showed that prescribed fire, mechanical, and combination treatments increased inorganic N in the mineral soil compared to the control (table 10). This was not unexpected as (1) forest floor-bound N is typically released in inorganic form during burning, and (2) thinning plus burning may alter the soil microclimate, enhancing N mineralization rates and, consequently, the release of inorganic N. The measured spike in inorganic N in the first year after treatment was fairly moderate, however, and short lived.

Prescribed fire, alone and combined with mechanical treatment, significantly reduced total C storage in the forest floor by an average of 7.3 and 6.5 Mg C ha^{-1}, respectively. These losses were mostly due to litter consumption and typically returned to pretreatment levels in a short period (Boerner et al. 2008a). In comparison, C stored in soil organic matter (SOM) to 30 cm depth was not significantly different than control values, with soil C changing less than 10 percent across the network (Boerner et al. 2008a). Short-term patterns of total N loss varied by treatment and region (fig. 40). On average, 84 percent of N lost from prescribed fire was from the forest floor pool, whereas 92 percent of N lost from mechanical treatment was from vegetation. Nitrogen losses resulting from combined mechanical and fire operations were closely split among these pools, with 55 percent from forest floor and 45 percent from vegetation. Overall, the FFS treatments removed less than 10 to 15 percent of the total N in these stands, primarily from forest floor and vegetation layers. Mineral soil storage, which accounts for more than 80 percent of total N at these sites (Boerner et al. 2008b), was unaffected and will likely provide an adequate buffer against soil quality or productivity impacts.

Cumulative effects research—

Results from the FFS study are in their infancy. Still, they offer early evidence of soil resilience to both single and combined treatments when applied using best management practices. As additional entries and burns are planned for most FFS sites, the study will ultimately provide long-term evidence of either soil tolerance, resilience, or susceptibility to cumulative practices across forest types and regions. In support of the FFS results, other studies of more limited geographical scope have demonstrated only modest changes in soil or site quality following repeated fuel reduction treatments (Binkley et al. 1992, Busse et al. 2009a, Hart et al. 2005a, McKee 1982). Still, our knowledge of cumulative soil effects from fuel reduction and forest restoration practices is incomplete and does not allow for broad-sweeping conclusions. Will repeated harvesting traffic eventually result in extensive damage to soil structure and greater soil erosion? Will repeated fire or whole tree harvesting operations deplete nutrients to suboptimal levels? Will the indirect effects of

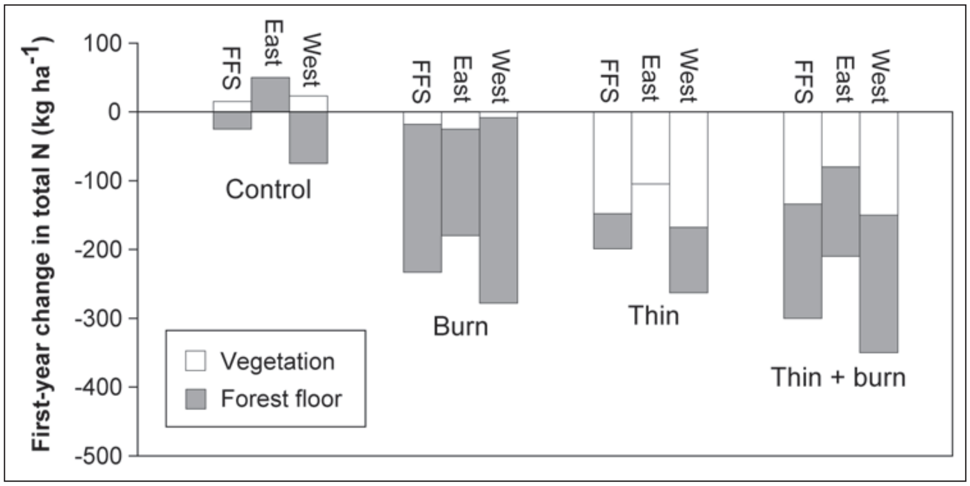

Figure 40—Net changes in total nitrogen (N) following fuel reduction treatments. Study site location: Fire and Fire Surrogate (FFS) = network average, East = Eastern United States, West = Western United States. Adapted from Boerner et al. (2008b).

repeated treatments (modified microclimate, variable forest floor depths, heterogeneity in forest structure) have a greater impact on long-term soil quality than any pulse effects of fire or harvesting? And which soil types and site conditions are most vulnerable to repeated disturbance?

Although the findings from most single-entry studies reported in this synthesis do not indicate any major soil concerns when using well-planned fuel reduction practices, little reassurance is afforded for addressing soil responses to multiple-entry, site-specific treatments. Resource managers can help safeguard against unwanted soil damage by continuing to document treatment intensity and extent, landscape pattern, and resulting soil quality of all repeated treatments until additional knowledge from studies like the FFS advances. More information on the FFS study can be found at http://frames.nbii.gov/portal/server.pt/community/fire_and_fire_surrogates_study/363.

Soil in a Changing Climate

Key points:

Ecological effects	Management considerations
• Sequestering or "hoarding" soil carbon (C) reduces atmospheric carbon dioxide concentrations and helps mitigate climate change.	• Current science to suggests that this process will play only a minor role in climate change mitigation. • Most evidence shows that soil C content is relatively stable following fuel reduction treatments, and that attempts to increase C storage will be limited to previously degraded land. • Until a better appreciation of the balance between sequestering soil C versus using it to fuel essential ecological functions is met, soil C sequestration might be best thought of as a lesser goal of forest soil management.
• United States soils will adapt to changes in annual precipitation, temperature, snowpack, vegetation, and wildfire risk (they have no choice).	• How well they adapt to environmental pressures is obviously unknown, as is the role of fuel reduction practices in helping buffer our soils against environmental change.

Soil's ability to function effectively in forest ecosystems wil be put to the test in the next century by our changing climate. Exactly how U.S. soil will be affected is uncertain; however, widely variable conditions and responses will likely be the case. Factors such as regional differences in climate change severity, latitudinal and elevational hotspots, soil vulnerability to climatic pressure, and complex interactions between soils and changing vegetation patterns add to the uncertainty. These unknowns also instill a sense of urgency in the natural resources community to identify and evaluate potential mitigative actions and adaptive strategies. For example, how does soil fit in the climate change equation? Does it offer any leverage, perhaps providing an untapped sink for long-term forest C capture (C sequestration), effectively lowering greenhouse gas concentrations? Or, are the benefits of relative insignificance compared to other socioenvironmental options. Here we briefly introduce soil C sequestration as an option for mitigating climate change and explore the projected impacts of climate change on soil quality.

Soil carbon sequestration and mitigating global climate change—Soil C sequestration is the process of removing CO_2 from the atmosphere (via photosynthesis) and storing it as SOM. Usually a portion of incorporated plant residues are stored in long-term, decay-resistant humus compounds (Six et al. 2002), which not only serve to sequester C and reduce greenhouse gases but also to improve soil quality. The remaining residues are more easily respired, returning CO_2 to the atmosphere in a relatively short-term turnaround process. Thus, soil C is sequestered and atmospheric CO_2 levels are lowered when the C input rate from residue incorporation is greater than the output rate from heterotrophic respiration. This is not as easy as it sounds, however, since C input and output rates are often comparable on a decadal timeframe (Janzen 2006).

Soil has an amazing ability to store C. Approximately 80 percent of all terrestrial C, or about 1500 gigatons (Gt) of C, is found in soil (Batjes 1996). This explains, in part, why soil continues to receive considerable scientific attention for its potential role in climate change mitigation (Kimble et al. 2003, Powlson et al. 2011). Atmospheric CO_2 levels increased from 315 parts per million (ppm) in 1960 to 383 ppm by 2009 and are expected by some to approach 800 ppm by 2100 (Solomon et al. 2007). By increasing soil C sequestration, more CO_2 would be withdrawn from the atmosphere, resulting in a negative feedback that reduces the effects of climate change. On the other hand, C losses through forest disturbance (e.g., wildfire) would cause more CO_2 to enter the atmosphere in a positive feedback that promotes climate change (Philander 2008). The germane questions become (1) how much more C can soils store to help offset rising greenhouse gas concentrations, and (2) can forest management practices assist in this process?

Most evidence suggests that land conversion from abandoned agricultural fields to grasslands or forests is the most viable means for increasing soil C storage (Post and Kwon 2000, Richter et al. 1999). Historical tilling and erosion of these lands led to substantial soil C loss which, in turn, has been shown to slowly aggrade to near-original levels once native vegetation is restored. Whether forest management practices such as thinning and prescribed burning modify soil C storage is in question, however (Hoover 2003). At present, most studies suggest that these practices do not substantially affect mineral soil C storage (Boerner et al. 2009, Johnson and Curtis 2001), and thus would not be expected to either mitigate or exacerbate atmospheric CO_2 levels. No confirmation of this with long-term data exists, however.

So the debate continues. Can soils store more C and play an important role in climate change mitigation? Lal (2004) suggests that global soils can sequester an additional 0.4 to 1.2 Gt of C each year, which is the equivalent of 5 to 15 percent of global fossil fuels emissions. Countering this optimistic view, Powlson et al. (2011)

argued there has been an overemphasis on the potential benefit of soil C sequestration and that it may detract from other more effective measures (e.g., slowing global deforestation). They reasoned that (1) soil does not accumulate C indefinitely (soil C storage is finite), and (2) the process of C sequestration is easily reversible unless the change in management practices that led to increased C is continued indefinitely. Instead, they discussed the value of a "no regrets" soil policy, one that first encourages management practices that increase soil C content for the benefit of soil functioning and, second, reaps the benefits of climate change mitigation if actualized.

Janzen (2006) offers keen insight into the debate on soil C in asking "shall we hoard it or use it?" The dilemma, as he points out, is that the ecological value of soil carbon is derived primarily from its decay (releasing nutrients, fueling biological processes, providing glues for soil structure). Now we are asking systems to accrue C while simultaneously allowing them to decay. Is such a "win-win" proposition realistic or not? He suggests that we must find approaches to (1) increase C inputs, (2) optimize the timing of decay to match ecosystem needs, and (3) better understand the input and output flows of soil C, not just evaluate C stocks. Forest management approaches for dealing with soil C sequestration are clearly in an experimental phase.

Adapting to global climate change—
Regional changes in the frequency and severity of precipitation events, heat waves, drought, floods, and hurricanes remain a key uncertainty in future climate change. Intensified storm events will affect soil erosion rates. Higher moisture and temperature will result in deeper, more leached soil profiles with clay eluviation to lower horizons: however, these effects are slow and will be overshadowed by erosion events and changes in management practices (Paul and Kimble 2000). Heat waves and droughts will heat soils, increase soil evaporation rates, and reduce soil moisture content. Drier soils will stress trees, increasing tree dieoffs from insect infestation. Dead trees and longer fire seasons will result in an expansion of areas subject to wildfires. Table 11 provides a qualitative, first-approximation of projected impacts of climate change on U.S. soils and the environment.

Table 11—Projected soil impacts based on predicted U.S. regional temperature, precipitation, snowpack, wildfire risk through 2099[a]

	Projected environmental change				
Region	**Temperature increase**	**Annual precipitation**	**Snow-pack**	**Wildfire risk**	**Projected soil impacts**
	°C	*Centimeters*	*- - - Percent*	*- - -*	
Southern California	3 to 4	12 to 25 ↓	↓	40 ↑	Warmer temperatures, lower precipitation, reduced snowpack, and earlier snowmelts will result in reduced soil moisture and increased soil temperatures. Drought periods will be extended and interspersed with El Nino-driven intense rain events. Soil erosion will increase. Heavy winter rains will increase fine fuels, leading to greater fire hazard.
Northern California	4 to 6	12 to 25 ↓	↓	55 ↑	Warmer temperatures will reduce available soil moisture, especially from mid to late summer, and increase the length of wildfire season in some middle and upper elevation forests. Fires will be fueled by increased tree mortality and summer flammability. Some project a 30 to 50 percent decrease in forest productivity; however, increased atmospheric carbon dioxide (CO_2) concentrations may result in a 10 to 20 percent increase in forest growth and an increase in soil and forest floor organic matter.
Sierra Nevada, California	↑	↓	90 ↓	↑	Reduced snowpack and soil moisture along with hotter air temperatures will increase the number and severity of wildfires, affecting soil heating and erosion. An increase in rain-on-snow events will also lead to greater streambank erosion and runoff.
Oregon/ Washington	2 to 5	↓	80 ↓	↑	Declining snowpack will lead to reduced summer streamflows. Higher summer temperatures and earlier snowmelt will increase potential for wildfires and insect outbreaks and alter forest species composition.

Table 11—Projected soil impacts based on predicted U.S. regional temperature, precipitation, snowpack, wildfire risk through 2099 (continued)

Region	Projected environmental change: Temperature increase	Projected environmental change: Annual precipitation	Projected environmental change: Snow-pack	Projected environmental change: Wildfire risk	Projected soil impacts
	°C	*Centimeters*	*- - - Percent - - -*		
Alaska	3 to 7	5 to 25 ↑	↓	↑	Although precipitation is projected to increase, longer summers and higher temperatures will lead to decreased soil moisture and greater thawing of permafrost. Earlier warming of soil will increase the growing season, but will also increase wildfires and insect outbreaks.
Arizona/New Mexico	4 to 6	10 to 30 ↓	75↓	↑	Soil moisture will decrease as drought periods are projected to increase. The severity of monsoon storm events is expected to increase owing to warming land and water temperatures. Current drought (1999–2010) has resulted in substantial die-off of pinyon pines. A decline in riverflow will alter riparian vegetation and increase the loss of wetlands.
Rocky Mountains	2 to 4	↑	↓	↑	Less snow and more winter rain will lead to earlier snowmelt and lower summer streamflows. Changes will result in longer summer drought, reduced soil moisture, more wildfires, and greater insect infestations. Forest carbon storage will decrease during prolonged drought seasons.
Great Basin, Nevada/Utah	2 to 5	↑↓	↓	↑	Projections for precipitation will vary across the Great Basin on account of large differences in topography. Higher levels of CO_2 may favor cheatgrass and other forb and shrub invaders. Increased temperatures, winter precipitation, and fine fuels (cheatgrass) will increase the wildfire season and potentially lead to greater soil erosion and loss of biological crusts and soil nutrients.
Texas/ Oklahoma	3 to 6	↑	NA	↑	Projected increases in precipitation are based on localized intense rain events and are unlikely to offset decreasing soil moisture and water availability because of rising temperatures, drier spring and summers, lower humidity, and aquifer depletion. Changes in precipitation patterns will alter wetland ecosystems. Increases in temperature and drought severity will multiply the risk of grass and shrub fires. An increase in wildfires could deplete soil organic matter (SOM) reservoirs overtime.

Table 11—Projected soil impacts based on predicted U.S. regional temperature, precipitation, snowpack, wildfire risk through 2099[a] (continued)

Region	Projected environmental change				Projected soil impacts
	Temperature increase	Annual precipitation	Snow-pack	Wildfire risk	
	°C	*Centimeters*	*- - - Percent - - -*		
Great Plains	3 to 6	13 ↑	↓	↑	Northern areas will experience the greatest change in temperature. Projected increased precipitation will be highly variable over the region. Increased evaporation from rising air temperatures will likely overwhelm the extra moisture from precipitation, causing soil moisture to decline throughout much of the region. Areas of increased soil moisture may show gains in SOM and C sequestration. Increased severity of storm events will lead to greater flooding and erosion.
Great Lakes	4 to 8	10 to 25 ↑	↓	↑	Longer warm seasons will increase wildfire risk. Soil moisture is projected to increase up to 80 percent during winter in some areas, but decrease by up to 30 percent in summer and fall. This will result in some wetland ecosystems drying up entirely during summers. Increased soil moisture will result in increased C loading and sequestration.
Northeast	2 to 3	↑ ↓	↓	↑	Precipitation is projected to increase by 14 percent during the winter, but decrease by 2 to 3 percent during the summer. Extreme storm events will increase resulting in greater flooding and erosion. Warmer coastal areas may allow hurricanes to survive longer and gain more strength.
Southeast/ Central	2 to 6	↑ ↓	NA	↑	Warming waters will increase in the intensity of hurricanes. Fall precipitation will increase and summer precipitation will decrease. Declines in forest growth will result from heat stress and reduced soil moisture.
Florida	1 to 3	Little change	NA	↑	Warmer air temperatures will affect the biogeochemical cycles of C, nitrogen, phosphorus, sulfur, and redox cations in wetland soils and sediments. Higher air temperatures will lead to increased evapotranspiration and reduced soil moisture content.

NA = not applicable.

[a] Climate ranges are based on Grassland Fire Danger Index A2 and B1 scenarios (low-to-high temperature scenarios) and are adapted from Adams et al. 2009, Cayan et al. 2008, Chambers and Pellant 2008, Field et al. 1999, Hayhoe et al. 2006, Kling 2003, Romanovsky et al. 2007, Running 2009, van Mantgem et al. 2009, Westerling and Bryant 2008.

Conclusion

Most fuel reduction projects are driven by the desire to reduce or rearrange hazardous fuels. Although obvious, in practice this means that fuel priorities may take center stage while other affected resources fade into the background to absorb the impacts. Soils can be profoundly affected by mechanical operations, prescribed burns, and combination fuel treatments. However, with thoughtful planning and careful implementation, reducing fuels and proactively managing soil resources can be complementary objectives. This report provided a synthesis of soil chemical, biological, and physical responses to fuel treatment practices, as well as tools to help predict these effects. This information can be used to help prevent detrimental soil impacts. For example, compaction can be minimized by operating heavy equipment when soils are dry, and soil heating can be minimized by burning when soils are wet. But beyond preventive measures, we hope this information is also used to create desired soil conditions through fuel treatment practices. For example, prescribed fires can be used to create coarse woody debris (CWD), or mechanical thinning can create soil microclimate conditions that promote faster nutrient cycling, if those are important ecological objectives locally.

Soil ecosystems are dynamic, and we can anticipate both short- and long-term effects from treatment operations. For example, the rate at which fuels accumulate and decay will influence not only how long a fuel treatment is effective (Keane 2008), but also the depth and composition of the forest floor. Without continued treatment or wildfire, litterfall will accumulate in the form of falling leaves, cones, branches, and snags for decades (Keifer et al. 2006). Snags are often created by prescribed burning treatments, which serve as recruitment material for downed woody materials over several years to decades (Stephens and Moghaddas 2005). Snag retention and creation during fuel treatments will contribute to meeting longer term woody debris objectives. Whether N is removed during burning or thinning, forest managers might find it beneficial to monitor the response of N-fixing plants. These species can play a crucial role in restoring site N and maintaining site productivity. Knowing if a treatment will enhance or diminish N fixers can help managers determine how long it will take to replenish the N lost from a site. This may be an important consideration when planning followup treatments.

Under historical fire regimes, wildfires burned until available fuels were consumed or moisture conditions extinguished them. In the dry West, these fires typically occurred during late summer when ground and surface fuels were at their driest. Many fires resulted in patchy fuel and highly variable forest floor conditions, likely ranging from exposed mineral soil to areas that did not burn. In many ecosystems, extended fire exclusion has led to increased fuel accumulations, prompting the need for fuel reduction treatments. In designing these treatments, consideration should be given to the historical fuel conditions that likely occurred in the treatment area—in the variability of load, depth, and distribution across space and time.

This synthesis highlighted soil effects and concerns applicable to a broad variety of fuel treatment scenarios, soil types, and vegetation communities. The take-home messages that can be applied across most landscapes include:

1. Low- to moderate-severity burning results in nominal damage from soil heating, N loss, exposed mineral soil and erosion potential, or root and soil microbial mortality. In particular, burn prescriptions that encourage incomplete duff consumption generally produce benign effects on soil quality and functioning. Alternatively, mixed-severity burning that results in forest structural and compositional heterogeneity, variable forest floor consumption, and some changes in soil properties may be highly appropriate for meeting desired fuel conditions and safeguarding soil quality.

2. Severe burning and harvesting based compaction are high risk factors for detrimental soil damage **if** applied across a large percentage of the land. Resulting soil erosion, nutrient loss, and soil quality decline in these cases may be acute.

3. Burn piles that contain a high percentage of large-diameter wood generate extreme soil temperatures of lasting duration. This is of limited concern on a site or landscape basis **if** the piles occupy only a small percentage of the land surface. In comparison, burning of slash piles containing a mix of fuel sizes (e.g., precommercial thinning slash) generally does not produce excessive soil temperatures or changes in soil functioning.

4. Whole tree harvesting removes a relatively small percentage of a site's nutrient capital, and, in most cases, postharvest nutrient reserves remain adequate to meet ecosystem needs. Exceptions to this rule include harvesting in extremely nutrient-poor forests or in northern U.S. aspen stands. Simple apriori calculations of nutrient export are encouraged to gain perspective on anticipated site nutrient loss.

5. Soils are easily compacted by mechanical harvesting. Whether this translates to a loss in soil quality or reduced site productivity is uncertain, however, based on results from recent compaction studies. Therefore, a cautious approach when planning fuel reduction thinning operations is to first identify high-risk soil conditions (e.g., seasonally wet; clayey texture; low rock

content; low organic matter content; minimal forest floor development), then select best management practices based on site-specific restrictions (e.g., avoid harvesting when soil moisture content is near field capacity).

6. The effects of repeated fuel reduction treatments on soil quality are poorly studied. Although findings from most single-entry studies indicate few soil concerns when best management practices are adhered, any translation to multiple-entry, site-specific treatments is premature until more results from long-term studies become available. As a precaution, soil scientists can develop simple, qualitative accounting systems that track local nutrient loss (or gain) or changes in soil physical properties for consecutive entries of fire or thinning. Then, adjustments to the frequency of burning and harvesting can be suggested as appropriate.

7. Soil properties can vary widely across all spatial scales, from broad landscapes to individual microsites. Not only is heterogeneity the norm, but it is a desirable attribute that can be augmented by fuel reduction practices. For example, using burning techniques to create an uneven mosaic of forest floor consumption is sufficient to limit erosional loss, even in the presence of fire-induced soil water repellency. Similarly, acknowledging the natural variability in CWD content at the landscape scale helps avoid the need for strict managerial control on a treatment-area basis, which may offer little benefit to soil productivity anyway.

8. Several soils topics such as the long-term consequences of masticated fuel beds or fuel treatments in riparian areas remain largely unstudied, making rule-of-thumb assessments unverifiable.

Bear in mind that soil properties and functions are inextricably linked to the vegetation, fuels, land use, disturbance history, and climate conditions in which they are found. In practice, the most important knowledge first comes from a local understanding of how these factors vary and interact across a landscape. When coupled with local knowledge, the information provided here will allow for a more powerful interpretation and assessment of fuel treatment effects on our soil resources.

English Equivalents

When you know:	Multiply by:	To get:
Centimeters (cm)	0.394	Inches
Meters (m)	3.28	Feet
Hectares (ha)	2.47	Acres
Kilograms (kg)	2.205	Pounds
Tonnes per hectare (Mg/ha)	0.446	Tons per acre
Kilograms per hectare	0.893	Pounds per acre
Tons	907	Kilograms
Gigatons (Gt)	1×10^9	Tons
Degrees Celsius (°C)	1.8 °C + 32	Degrees Fahrenheit

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Appendix: A Soil Quality Primer

This section introduces soil quality as a conceptual tool for managing soil and provides examples of soil quality indexes, visual disturbance classes, and risk ratings used by land managers. Our intent is to provide supplemental information on soil quality principles, which themselves help define the limit of disturbance for long-term changes in soil function (Page-Dumroese et al. 2006a), in support of the 12 fuel reduction topics covered in this synthesis report.

Soil quality represents the ability of a soil to function as part of a healthy ecosystem (Karlen et al. 1997, Schoenholtz et al. 2000, Seybold et al. 1999). It is often used synonymously with the term soil productivity, although subtle differences exist between the two (Burger et al. 2010). Basic attributes of soil quality or soil productivity include (1) providing for plant growth, water retention, and carbon storage; (2) maintaining and sustaining biological activity, diversity, and productivity; (3) governing and partitioning water and solute flow; (4) filtering, buffering, decomposing, immobilizing, and detoxifying organic and inorganic materials; and (5) storing and cycling of nutrients and other elements within the Earth's biosphere (Burger and Kelting 1999, Karlen et al. 1997).

The present day concept of soil quality with an emphasis on holistic and sustainable soil management began to evolve in the early 1990s with substantial input from the Soil Science Society of America (Doran and Parkin 1994, Karlen et al. 2003). The development of guidelines to assess forest soil quality at the national and international levels then began in earnest as part of the United Nations Conference on Environment and Development (O'Neill et al. 2005). This led to the Santiago Declaration of 1995 and produced the Montreal Process, consisting of seven criteria and 67 indicators applicable to the conservation and sustainable management of temperate and boreal forests. Criterion 4 addresses the conservation and maintenance of soils, with indicators focused on best management practices for limiting soil degradation.

The origins for managing lands using the soil quality concept were already well established before the 1990s. For example, the U.S. Forest Service (USFS) developed soil guidelines in the early 1970s and identified threshold levels which, if exceeded, were considered detrimental to soil productivity (Page-Dumroese et al. 2000). Similar guidelines and thresholds were adopted shortly thereafter by other public land agencies. The National Forest Management Act (NEPA) of 1976 gave legal credence to these efforts by requiring the development of resource management plans that ensured the productivity of forested lands be sustained in perpetuity. In support, the Forest Inventory and Analysis and Forest Health Monitoring programs of the Forest Service now provide systematic monitoring of soil properties across all forested regions of the United States and help develop benchmarks for regional, national, and international reporting on sustainable forest management and soil quality (O'Neill et al 2005).

Present day objectives of the USFS are to "maintain or restore soil quality" and manage soils "to sustain ecological processes and function so that desired ecosystem services are provided in perpetuity" (USDA FS 2010). Agency guidance encourages both qualitative and quantitative methods to evaluate soil quality, yet specifies that qualitative measures are generally sufficient to meet soil management objectives. Forest Service regions and local management units supplement this guidance with specific standards, guidelines, and threshold values as needed. In practice, the Forest Service advocates three elements for managing its soils (USDA FS 2010):

(i) Soil quality assessments. Local assessments incorporate past and ongoing management activities and are used to establish benchmark (or pretreatment) soil conditions.

(ii) Soil quality analyses. Soil analyses use a combination of local monitoring results, local knowledge, and scientific literature to predict future soil conditions. The purpose is to assist managers when writing environmental documents (such as NEPA analyses) and proposing treatment options.

(iii) Soil quality monitoring. Monitoring is required to evaluate changes in soil properties with time. Repeated measurements provide an understanding of the extent, intensity, and duration of management impacts on soil.

Evaluating Soil Quality

No universal standards exist for measuring soil quality or specific ratings against which all soils can be compared (Karlen et al. 2003). Soil properties vary greatly at national, regional, and landscape scales owing to differences in parent material, climate, vegetation type, topography, age, and land use history (box 4). With these differences come a variety of responses by soils to disturbance (e.g., tolerance, resilience, or detrimental damage). Therefore, establishing soil quality indexes that would be a standard throughout the U.S. is futile (Burger and Kelting 1999, Burger et al. 2010, Page-Dumroese et al. 2000, Schoenholtz et al. 2000). Instead, soil scientists must select appropriate soil quality indicators for given locations and treatments. As an example, soil quality issues during harvesting differ between a well-drained, infertile soil where nutrient depletion is a primary concern versus a fertile clay soil that is susceptible to compaction, rutting, and displacement (Burger

Box 4

Soil-Forming Factors

1. **Parent material** is the primary material from which soil is formed. Examples include bedrock, organic material, a stream deposit (alluvium), wind deposit (eolian material), glacial deposit (till), volcanic ash, or material moved down slope by gravity (colluvium). The composition and fragmentation of parent material affects the rate of soil formation and the chemical and mineralogical properties of soil. For example, rocks with small-grained minerals tend to undergo faster chemical weathering than rocks composed of larger grains.

2. **Climate** forces such as heat, rain, ice, snow, and wind facilitate the breakdown of parent material and determine how fast or slow soil formation proceeds. For example, wet climates of the Eastern United States produce highly weathered soils by promoting rapid chemical and biological weathering, whereas weathering processes and soil development are slower in the dry Southwestern United States.

3. **Living organisms** include all plants, animals, and microbes living in or on the soil. Vegetation type and density influences the soil by providing cover, organic matter, and belowground biomass. Organisms affect decomposition and transport of soil materials (e.g., preferential flow through root pores and bioturbation). The dead remains of plants and animals are recycled by micro-organisms into organic compounds that enrich the soil. This is evident when comparing the nutrient-rich soils of wet, northwestern Washington with the low-nutrient-status soils in southwestern pinyon/juniper forests.

4. **Topography** is the shape of the land surface in which a soil occurs. For example, soils tend to be shallow on steep slopes and exhibit little development of soil horizons because of the constant movement of soil downhill. Generally, soil temperatures are warmer on south-facing slopes than on north-facing slopes; soil moisture content is greater on north slopes and less moist on south slopes (moisture accelerates chemical and biological weathering); and vegetation is denser on north slopes as compared to south-facing slopes.

5. **Time** plays a role in all of the above factors. This role can assert itself over hundreds or thousands of years. With time, physical, chemical, and biological weathering processes result in the development of soil horizons that differ in color, texture, and chemical makeup.

Adapted from Jenny (1941).

and Kelting 1999). Fuel reduction treatments present their own unique concerns related to soil quality. Mechanical thinning can result in detrimental soil compaction, whereas severe burning can lead to unwanted soil heating and nutrient loss. Therefore, the goal of monitoring soil quality is to develop an indexing procedure that can be adjusted for different soils and be used to enumerate ratings that are site specific for a variety of ecosystems (Karlen et al. 2001).

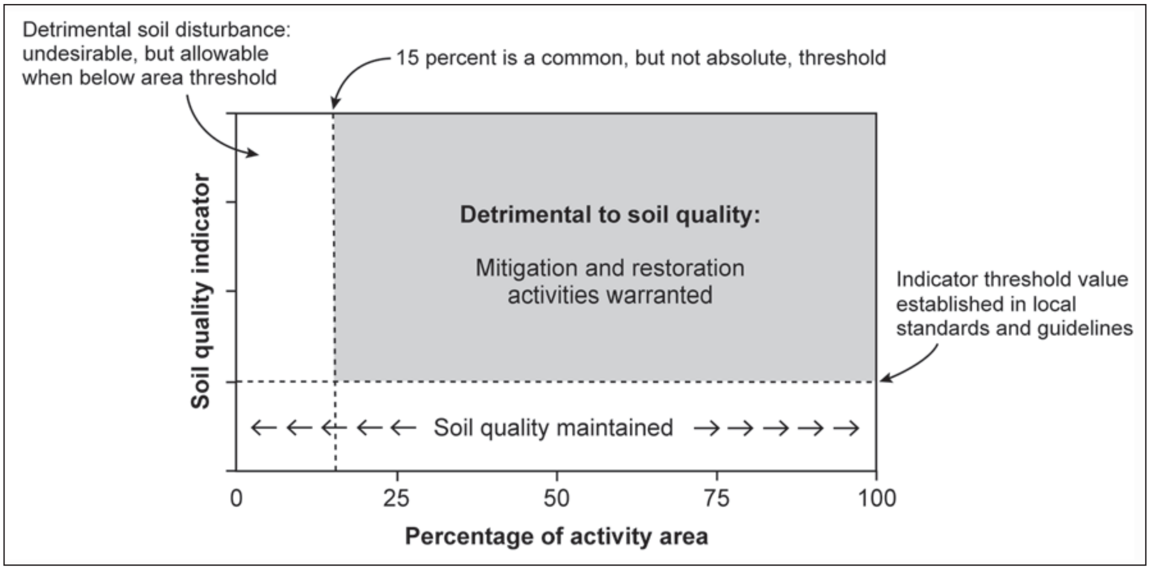

Figure 41—Thresholds for detrimental soil disturbance. Detrimental soil conditions exist when indicator values are above the threshold level (horizontal dashed line). An activity area is considered detrimentally impacted when the extent of detrimental soil conditions exceed the area threshold (vertical dashed line), gray area of chart.

For decades, the USFS has used soil properties that are easy to quantify and of known sensitivity to management practices as indicators of soil quality, including soil compaction, rutting, displacement, erosion, mass movement, groundcover, and burn severity. These properties are measured either visually or with rapid and inexpensive methods, and their cumulative change is assessed within an activity area using either a random or systematic sampling procedure (Page-Dumroese et al. 2009). Threshold values for each soil quality indicator have been established to predict when conditions become detrimental to soil quality. For example, a 10-percent reduction in soil porosity has been used as a threshold for detrimental compaction (USDA FS 1995). Additionally, a threshold for the cumulative extent of detrimental disturbance within a treatment area is set as a management trigger to determine if mitigation activities are warranted. Traditionally, a value of 15 percent of an activity area was used as a disturbance threshold as this was thought to be the lowest value that will manifest as observable changes in soil productivity (fig. 41) (Powers et al. 1998).

Traditional soil quality indexes—

***Soil compaction*—**Soils become compacted when they are compressed to a smaller volume by surface traffic such as harvesting equipment, livestock, wildlife, or humans. This causes a rearrangement of soil particles with a concomitant reduction in the number and continuity of large pores (Greacen and Sands 1980). Common effects of compaction on soil quality include:

- Decreased soil porosity, water infiltration, hydraulic conductivity, and air exchange.
- Increased erosion by surface runoff.
- Restricted rooting area and inhibited root penetration (fig. 42).
- Reduced rates of organic matter decomposition and nutrient cycling.
- Reduced water storage and nutrient availability in clayey soils.
- Improved water storage in sandy soils.
- Greater potential for anaerobic, waterlogged conditions.

Absolute changes in these properties vary greatly depending on specific site conditions such as (1) soil moisture—moist soils compact easier than dry soils, (2) soil temperature—frozen soils resist compaction, (3) soil organic matter (SOM) content—organic matter is more difficult to compact than soil minerals, (4) rock fragments—rocky soils are resistant to compaction, (5) snow or forest floor cover—surface cover helps diffuse compaction forces, (6) number of passes by machinery—substantial compaction may occur after only a few passes by high ground pressure equipment, and (7) extent of ground coverage—designated skid trails produce less aerial disturbance than random movement. Because of this complexity, the task of assessing compaction severity is challenging. Several procedures are currently used, yet no method is ideal (inexpensive, quick, reliable, reproducible, sensitive to regional variability). Both ocular and tile-spade estimations are used frequently since they are rapid and inexpensive, although their ultimate usefulness depends on an operator's experience and interpretational skills. Bulk density and cone penetrometer measurements are recommended when quantitative measures are required.

Figure 42—Soil compaction following a thinning operation.

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Figure 43—Severe rutting from harvesting on top of moist soil.

Rutting and puddling—Rutting occurs when heavy equipment is driven on wet soil (fig. 43). Evidence of rutting can remain for many years, altering soil hydraulic flow and decreasing site productivity (Han et al. 2006). Deep, rutted areas can keep soil temperatures cool, inhibit root growth, and slow organic matter decomposition. Even shallow rutting can cause degradation in site quality by altering the flow of soil water and gasses. Rutting of clayey soils may lead to soil puddling, which further changes soil hydrologic function and adversely affects site productivity.

Reliability of rutting as a visual indicator of soil disturbance is considered high because it is easily detected and measured. However, applying the same standards for rutting across regions may be misleading. For example, shallow rutting in arid environments may persist for many years, whereas the effects of deep rutting in moist southeast forests can be short lived when ameliorative practices such as bedding and providing drainage channels are used (Kelting 1999).

Soil displacement—Soil displacement is the movement of surface soil horizon (mineral or organic) from one place to another by erosive or mechanical forces (fig. 44). Displacement can also include soil mixing which occurs when the topsoil and subsoil are mixed during logging operations, or by ripping, tilling, or windthrow. Soil displacement degrades site quality by exposing the nutrient-poor subsoil and altering slope hydrology. Lateral displacement of soil may decrease productivity by disruption of water distribution, damage to root systems or, in extreme cases, uprooting and toppling of trees (Clayton et al. 1987). On marginal, low-nutrient-status sites, the loss of the organic layer alone can be detrimental. On steep slopes, displacement of soil can change subsurface and surface hydrological flow resulting in soil slips and mass failures.

Figure 44—Soil displacement.

Because of its qualitative and subjective nature (ocular assessment and dependence on the observer's field experience), the reliability of this indicator may be considered moderate. In forest soils, for example, it is sometimes difficult to determine the boundary between the forest floor and the mineral soil. In addition, the depth of the surface horizon can vary widely across the landscape.

***Erosion*—**Soil erosion is the detachment and movement of soil by gravity, water, wind, or ice. It is a natural process that, in the absence of human intervention, often matches the rate of soil formation in a geological timeframe (about 1 to 4 Mg ha^{-1} yr^{-1}). Detrimental soil erosion occurs when soils are exposed (e.g., wildfire, road building, thinning operations, severe prescribed burning) and the rate of erosion exceeds the rate of soil formation.

The primary types of erosion in wildland systems are dry erosion and water erosion, which result when plant cover or forest floor cover is removed, leaving hillslopes bare. Dry erosion, or ravel, is the unconsolidated flow of soil owing gravity (Rice 1974) (fig. 45). Any disturbance can initiate downslope movement of ravel when slopes are greater than 55 percent. Below 55 percent, wind acts as the catalyst to initiate movement of material downslope. Ravel is typically redistributed on the hillslope and eventually delivered to the drainage channel. Water erosion is caused by surface water runoff when either (1) rainfall intensity exceeds the soil infiltration rate (Horton 1945), or (2) the water storage capacity of the soil is exceeded by rainfall (saturation overland flow). Once overland flow begins, the extent of erosion is governed by numerous site factors including the intensity and duration of the rain event, slope steepness and length, surface cover, and soil texture, structure, and antecedent moisture content (Moffett et al. 2007).


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Figure 45—Dry ravel erosion.

Rills, gullies, channels, pedestals, and landslides are easily observed and symptomatic of excessive soil and nutrient loss from a site. Rills are the result of the cutting of numerous small water channels by concentrated surface runoff. As concentrated flows continue to cut into the hillslope, rills become larger, forming gullies or large channels. Pedestals are remnants of former surfaces that are indicative of interrill erosion (or sheet erosion). Landslides occur typically in steep, unstable landscapes when clayey soils become saturated. The weight load on the slope increases as previously empty pore spaces and fractures fill, which leads to increased pore pressure, reduced shear strength, and, ultimately, slope failure and landslides.

Unlike visual assessments of these erosional processes, quantitative measurements of soil loss are exceedingly difficult to implement on an operational scale. Soil erosion models offer an alternative approach for semi-quantitative assessments of erosion. The “water erosion prediction project” model (WEPP; http://www.ars.usda.gov/Research/docs.htm?docid=10621) was developed to predict wet erosion within small watersheds and hillslopes, and its predictive accuracy has been further improved with a spatial erosion modeling tool, GeoWEPP, which uses geo-referenced, site-specific input parameters and defaults.

Ground cover—Ground cover, including plants and forest floor residues, serves many soil quality functions. These include reducing the risk of surface soil erosion, regulating soil temperature and moisture, and storing carbon (C) and nutrients. In addition, forest floor residues function as a fuel source for fire and as a restrictive layer for plant germination and survival if excessively deep. Despite these diverse functions, soil quality guidelines are often primarily concerned with the role of ground cover in protecting against soil erosion (fig. 46). Loss of cover exposes bare soil to raindrop impact, causing the deterioration of soil structure and the onset of sheet erosion. Generally, the degree of patchiness of forest floor cover determines its effectiveness in intercepting rainfall and preventing surface runoff and erosion. Robichaud et al. (2000) noted that erosion is effectively controlled at 60 percent plant cover, even during high-intensity rain events.

Burn severity—Soil quality is affected when burning is severe. Critical issues include soil erosion from loss of forest floor cover, change in soil pH and nutrient status, and loss of SOM. Severe heating within the mineral soil profile may occur beneath concentrated fuels such as large slash piles, downed logs, and thick masticated fuel beds (fig. 47). For example, Ulery et al. (1993) observed a 1- to 8-cm-thick reddened layer beneath large burned out logs that they attributed to heat-induced iron (Fe) oxide transformations. In comparison, low- to moderate-severity burning results in blackened soil as a result of charring of organic matter, which may temporarily enhance soil quality resulting from an increase in base cations and available nitrogen (N).

Visual observations can identify color changes in the mineral soil and loss of surface cover, but they do not serve as effective surrogates for changes in soil chemical and biological properties. For example, the color of the mineral soil may naturally be red owing to the presence of hematite in weathered soils. In this case, color cannot be used as an indicator of severity class.

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Figure 46—Insufficient ground cover leading to surface wind erosion.

Terry Craig

Figure 47—Visual scar left from high-severity pile burning.

Visual soil quality disturbance classes—

Traditional guidelines and thresholds have been used effectively to evaluate soil changes associated with management practices. However, they can be cumbersome and costly to implement. Further, the lack of a standard, nationwide reporting system prohibits the development of rigorous statistics to help explain the effects of treatments such as fuel reduction activities on soils. These drawbacks led Page-Dumroese et al. (2009) to propose a monitoring protocol based on a visual disturbance rating system that is both practical to implement and can provide standardized information over a wide range of ecosystems and soils. Four soil disturbance classes and their identifying features developed by Page-Dumroese et al. (2009) are shown in table 12. The disturbance classes essentially account for the same set of soil quality indices as the traditional methods (compaction, rutting, displacement, erosion, ground cover, burn severity), using visual observations taken at numerous gridpoints within a treatment area to develop an aggregate score between 0 (no disturbance) and 3 (highly disturbed) for an entire unit. This assessment is particularly suitable for fuel reduction practices with its emphasis on tractor-derived compaction, forest floor integrity, and burn severity. Further, the expectation is that all disturbance scores will be entered into a nationwide database (in development), allowing resource managers to better understand trends in soil responses across landscapes, soil types, and treatments prescriptions.

Table 12—Soil disturbance classes based on Page-Dumroese et al. (2009)[a]

Severity class	Identifying features
0	• Forest floor undisturbed • No evidence of past equipment operation • No soil char or evidence of litter and duff burning • No mineral soil displacement or erosion • Water repellency at background levels
1	• Forest floor intact • Faint wheel tracks and ruts • No mineral soil displacement and minimal mixing with forest floor • Slight increase in soil compaction • Light burning may be evident • Repellency similar to preburn levels
2	• Forest floor partially intact or missing • Wheel tracks (5 to 10 cm deep) created by one or more passes • Compacted, platy structure to a soil depth <30 cm • Moderate erosion • Burning moderate, depth of char <5 cm • Increase in surface water repellency
3	• Forest floor missing • Wheel tracks >10 cm deep • Majority of soil surface is displaced • Compaction extends beyond 30-cm depth • Severe burning—duff fully consumed; soil char depth >5 cm • Increase in surface water repellency

[a] Severity of impacts increases from class 0 to class 3. Severity class is determined at multiple gridpoints, systematically located within a treatment unit to determine a mean severity class rating.

Visual disturbance classes offer simple and standardized soil quality assessments. Their drawbacks are that they are highly qualitative in nature (Napper et al. 2009) and the observations are dependent on the expertise of the observer and may be difficult to interpret. For these reasons, it may be necessary to supplement visual observations with quantitative measures of soil quality on a subset of treatment units that are of particular concern or are representative of benchmark soils and site conditions.

Soil Risk Ratings

Risk ratings offer managers a decision-support tool for planning purposes that identifies vulnerability to disturbances such as compaction, erosion, and severe burning (Curran et al. 2005). Table 13 provides a simple example of a risk rating developed by USFS Pacific Southwest Region (California) soil scientists to identify

Table 13—Compaction risk ratings based on texture class and coarse fragments

Compaction hazard	Texture class	Coarse fragments > 2 mm
Low	Sandy Any texture	Any amount Greater than 70 percent
Moderate	Loamy texture Clayey	Any amount Any amount
High	Silty	Less than 35 percent

Source: Adapted from USDA FS 2006.

at-risk soils. This rating system provides a first-approximation of compaction risk based on soil texture and rock content, and is similar to a compaction rating system used in British Columbia (Curran et al. 2007). It essentially provides a rough assessment of site conditions where caution may be required and, conversely, where soil conditions are likely tolerant to disturbance (e.g., compaction in rocky soils of any texture). In turn, the success of this or any risk rating system relies on quality data provided by extensive field observations and research findings, along with a commitment to update the rating system as new information becomes available.

An added benefit of risk ratings is that they offer an implicit decision tree for selecting best management practices for sustaining soil quality. For example, manageable options for harvesting on top of clay soil with low rock content (high-hazard rating in table 13) might emphasize selection of the appropriate harvest timing to avoid wet soil, use of low-impact equipment, or the presence of thick slash and forest floor layers.

While soil risk ratings offer a valuable "triage" assessment for project planning, they may be overly simplistic for some uses. For example, an assessment of compaction risk based only on texture and rock content fails to acknowledge countless other site factors such as slope, soil water content, forest type and assumed effects on site productivity, and cumulative effects from previous entries. Recognition of this limitation has led to recent advances in soil risk ratings through the use of soil and geology databases linked visually through geographic information systems (Kimsey et al. 2011, Reeves et al. 2012). This evolving technique holds promise as a user-friendly, cost-effective tool for planning purposes that can (1) identify sites with vulnerable soil physical, chemical properties, and (2) help select best management practices to protect the resource.

Quantitative Indicators of Soil Quality

Quantitative assessments of soil chemical and biological properties provide definitive evidence of changes in soil quality. Thus, they can be used to validate the findings of adverse soil conditions identified by qualitative (ocular) methods or to provide benchmark data for environmentally sensitive areas (e.g., riparian zones) or for analyses of cumulative treatment effects. Although far from a comprehensive list, we identify below several options for chemical and biological indicators of soil quality. Most soil chemical indicators can be analyzed at commercial laboratories for moderate cost, whereas there are few commercially available or affordable biological measures of soil quality. These, instead, fall in the category of research-based indices.

Chemical indicators—

All fuel reduction practices affect soil chemical properties to a certain degree. Thinning removes nutrients from a site, alters plant competition and demand for available soil nutrients, and can alter soil hydrological properties that indirectly affect soil nutrient availability. Prescribed burning also affects soil nutrient availability and organic matter content—either positively or negatively—depending on fire severity. Useful chemical indicators include SOM, total and available N, soil pH, cation exchange capacity, total or available phosphorus, and heavy metals. For a more complete description of these and other potential indicators, see review articles by Doran and Jones (1996), Pankhurst et al. (1997), and Weinhold et al (2004).

Soil organic matter. Soil organic matter is critical to the biological, physical, and chemical health of all soils and is considered a cornerstone measure of soil quality. Long-term monitoring of SOM may be warranted in some cases since as burning and thinning can reduce SOM levels if sufficiently severe or aggressive.

Total nitrogen. Nitrogen is the second most limiting soil constituent (following water) that constrains plant growth and net productivity in most ecosystems. Like SOM, soil N can be altered by fuel reduction practices, justifying a long-term monitoring plan of forest floor and mineral soil N when cumulative treatment effects are anticipated.

Soil pH. Soil pH is a key indicator of soil quality. It controls the availability of many plant nutrients and toxic compounds, and regulates most microbial processes. Calcium, magnesium, and oxyanions such as sulfates and phosphates become more available as pH increases to alkaline levels. Conversely, many micronutrients and heavy metals become more available as pH decreases. Prescribed fire generally raises pH depending on the base cation content of ash, although the increase may be minimal or short lived (Murphy et al. 2006).

Other chemical indicators. Other options for chemical indicators include cation exchange capacity (useful for fire effects monitoring); available N (ammonium, nitrate; ephemeral measurement that can vary considerably with season of year); phosphorus (total P or available P); additional nutrients and heavy metals. These are generally high-cost analyses and may be most appropriate for research-level monitoring of fuel reduction practices.

Biological indicators—

Soil biological activity is an important attribute of a healthy soil. Soil organisms regulate nutrient cycling, decompose SOM, protect against pathogens, and mediate heavy metal toxicity. They also help improve soil structure through the binding properties of fungal mycelium, which, in turn, improves soil porosity, water infiltration, and plant-available water content.

Biological indicators include individual species, functional groups, and their ecological processes (McGeoch 1998). They have proved to be useful tools for detecting changes in the environment because of their rapid response to disturbance. Drawbacks exist, however, such as their inordinate numbers, taxonomic challenges, general unfamiliarity, and ineffectual methods of detection (Andersen et al. 2002). Criteria for potential indicator organisms or processes include that they (1) play a key role in the functioning of the soil ecosystem, (2) occupy a wide range of ecosystems enabling comparisons between systems, (3) are abundant and easily recognized, (4) are easy to sample and analyze, and (5) are not too sensitive to environmental stresses that they become extinct (Edwards et al. 1996). Table 14 lists common soil organism groups that function as indicators of soil quality.

Table 14—Soil organisms as bioindicators for determining soil ecosystem health

Organism group	Use as bioindicator	Soil function
Macrofauna:		
Earthworms	General indicators of soil quality (presence is typically limited to nonacidic soils, hardwood forests, and moist climates	• Mixing and transport of surface organic matter deep into the mineral soil • Improve soil structure, porosity, and water movement • Improve soil fertility
Termites	Limited primarily to subtropical and tropical climates	• Mixing and transport of surface organic matter into the mineral soil • Improve soil structure, porosity, water movement • Digestion of soil wood
Ants	Strong indicators of habitat disturbance	• Transport organic matter and minerals from lower horizons to the soil surface • Increase soil porosity and lower soil bulk density • Govern soil hydrological processes • Break down of soil wood
Millipedes	Indicators of soil physical and chemical perturbation in moist areas	• Assist in the breakdown of surface organic materials such as leaves, twigs, and roots
Carabid beetles	General indicators of habitat alteration. (populations are often reduced following fire or mechanical treatments)	• Chewing and mixing of forest floor organics
Mollusks (snails)	Indicators of heavy metal pollution	
Mesofauna:		
Springtails Mites	Key indicators of soil food web health	• Predators of bacteria, fungi, and algae • Help decompose surface organics
Enchytraeids (potworms)	Indicators of chemical stress from pesticides, heavy metals, and fertilizers	• Decompose and transport of surface organic matter into the mineral soil
Microfauna:		
Nematodes	Indicators of soil nitrogen (N) cycling and assorted soil chemical and physical processes	• Predators of bacteria • Plant pathogens
Protozoa (ciliates)	Indicators of soil food web health	• Enhance nutrient cycling processes • Predators of bacteria • Facilitate organic matter decomposition

Table 14—Soil organisms as bioindicators for determining soil ecosystem health (continued)

Organism group	Use as bioindicator	Soil function
Micro-organisms:		
Fungi, bacteria, algae, viruses	Traditional indicators of soil quality and disturbance response in all ecosystems	• Nutrient cycling • Organic matter decomposition • Improve soil structure (fungi) • Improve plant nutrient and water uptake (mycorrhizal fungi) • Sequester carbon • Release greenhouse gases • Symbiotic relationships with plants • Degrade toxic chemicals • Suppress plant pathogens • Pathogens

Source: Adapted from Chen et al. 2006, Deka et al. 1983, Didden and Rombke 2001, Esquilin et al. 2007, Foissner 1999, Joschko et al 1989, Kobziar and Stephens 2006, Li et al. 2004, Luo and Zhou 2006, McCarthy and Brown 2006, Neher 2001, Rainio and Niemela 2003, van Straalen and Verhoef 1997.

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