

Sustaining forest soil quality and productivity

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Forest soil sustainability is not an ideal, fixed, or static goal (Nambiar, 1996; Lal, 2005). Therefore, for this chapter, we define *sustainable soil* as a product of the interaction of soil physical, chemical, and biological properties that function to produce biomass, store carbon (C), bioremediate wastes, regulate water quantity and quality, and promote above- and belowground biodiversity. Ecological and climatic changes, along with pressure from urban growth, all influence soil properties in large and small ways (e.g., land use, soil biodiversity). Because soils are dynamic systems, it is critical to understand the links among soil properties, land management, and forest sustainability (Burger and Kelting, 1999; Page-Dumroese et al., 2009). There are a number of severe forest soil impacts that can occur from management [e.g., coal-mining (Burger et al., 2017), gas pads (Drohan et al., 2012), shale-gas infrastructure (Fink and Drohan, 2015)], but these sites are often spatially limited, or have limited data about restoring and understanding dynamic soil processes (Fink and Drohan, 2015), and are not addressed in this chapter. The focus of this chapter is on if, when, and how forest harvesting and site preparation can affect soil physical, chemical, and biological properties and highlighting several management considerations that help ensure long-term soil productivity (LTSP) and forest sustainability. We also discuss how soil monitoring can provide long-term data to inform adaptive management and contribute to our understanding of healthy forests and soils.

3.1 Soil health for sustainable forest management

Productive soils are essential to productive forests and are a nonrenewable resource that should be protected, restored, and enhanced to preserve forest production and ecosystem services, such as water quality and quantity, and decreased erosion, and help mitigate climate change through C sequestration (Powers et al., 1998). Recently, there has also been widespread interest in using forest biomass for energy production, fire hazard reduction of overstocked stands, and restoration of forests decimated by drought or insects (He et al., 2014; Miner et al., 2014), which increases the need to understand the relationship of soil properties to site productivity before and after forest operations. Understanding soil impacts requires a dedicated soil monitoring effort. There are numerous international monitoring acts and laws that have been enacted to ensure sustainable practices [e.g., Helsinki Process, 1994, Montreal Process, 1995, Sustainable Forestry Initiative (SFI), Forest Stewardship Council (FSC), American Tree Farm System, state BMP guidelines, and Forest Practices Acts] and which define forest management success as the presence, density, or biomass of plants at a point in time (Ruiz-Jaen and Aide, 2005), but they do not include a metric for soil recovery, sustainability, or improved function (Aronson et al., 2006).

Sustainable forestry at the stand level includes optimizing the harvest unit size, a minimum “green-up” period after harvesting, stocking level, genetic material, wildlife corridors, and the maintenance of water quality (Burger and Kelting, 1999). Similar to international efforts, US guidelines and policy for stand-level forestry also target maintaining soil sustainability. Although many forest criteria address the issue of soil sustainability, there are many complicated factors that make it difficult to determine if soil productivity is affected after harvest operations as discussed later.

3.1.1 Important factors for soil sustainability

Soil loss is irreversible on the human timescale and, therefore, soil conservation is a key to maintaining forest productivity. In 1993 the United Nations convened an international seminar in Montreal, Canada on sustainable development of temperate and boreal forests. This conference led to several multicountry initiatives (e.g., Montreal Process, Helsinki Process) and resulting in criteria and indicators for sustainability which specifically concern soil impacts from forest management. For example, indicator 22 states “area and percent of forest land with significant compaction or change in soil physical properties resulting from human activities.” These criteria also include maintaining the C cycle and ecosystem services such as water yield. Soil-based indicators such as erosion, diminished soil organic matter (OM), and compaction are evaluated and are meant to communicate what participating countries want to see as the condition of their forests.

Generally, all ecosystems will recover from disturbance given enough time (Morris et al. 1997, 1983). For example, forest sustainability can be achieved if timber harvesting rotations are longer than the time it takes the soil to recover. If the stress is not too great and the frequency of disturbance is low, the system will eventually return to a predisturbance condition; however, it is critical to understand the starting state of forest conditions (Morris et al., 1997). Successful recovery and future productivity in a changing climate will rely heavily on soil OM and nutrient conservation (Wells and Jorgensen, 1979; Kimmins, 1977).

3.2 Soil organic matter

Soil OM is critical for sustaining soil productivity because of its role in nutrient availability, gas exchange, and water supply (Jurgensen et al., 1997; Powers et al., 1998) and is essential for micro- and macrofauna that are active in nutrient cycling, soil aggregation, and disease incidence or prevention (Harvey et al., 1987). Soil C and nutrient pools as well as soil physical properties, such as water-holding capacity and bulk density, are directly correlated with soil OM content (Vance et al., 2018). Forest floor materials also influence vegetation reproduction, erosion, fuel type and fire behavior, and susceptibility to disease, but the exact amount needed to sustain soil productivity varies by site and soil type (Grigal and Vance, 2000).

3.2.1 Loss of soil organic matter during management

Many forest management operations can affect soil OM content by altering the inputs and distribution of surface litter, root turnover, and microbial activity. Studies on the effects of harvest operations on soil OM and C formation, however, have produced mixed results. In general, timber harvest operations, especially clearcutting, may alter mineral C pools (Johnson and Curtis, 2001), although the majority of C loss is from the surface organic horizon (Powers et al., 2005; Nave et al., 2010). Some researchers point out that changes in OM are site-specific and often produce a cascade of secondary changes (Clarke et al., 2015). For example, harvest operations that remove woody litter and downed wood may disrupt the input of decomposing C into the mineral soil layer resulting in higher soil temperatures which increase decomposition rates (Covington, 1981), or changes in soil moisture availability (Gordon et al., 1987; Prescott et al., 2000) that could increase or decrease decomposition rates. Harvesting also supports the development of a new aggrading stand that may increase litter C inputs as compared to preharvest levels (Fleming et al., 2006), whereas mixing soil OM into the mineral soil by equipment might also increase decomposition (Jandl et al., 2007). Any reductions in soil OM are likely to persist until the new forest has fully occupied the site

(Powers et al., 2005). On the North American LTSP study sites, OM removal treatments during clearcutting operations (bole only, whole tree, and whole tree + forest floor removal) resulted in little net difference in soil C after 5 years except at the North Carolina study site (Table 3.1). The dramatic increase in soil C in North Carolina was similar to that observed following harvest on the South Carolina Piedmont (Van Lear et al., 1995) and was attributed to root decomposition (Sanchez et al., 2006). However, longer term data from North Carolina indicated that whole tree harvesting + forest removal, resulted in declines in soil C with time were attributed to the loss of the forest floor and its role in supplying C inputs (Powers et al., 2005).

3.3 Soil nutrients

It is well established that soil nutrient limitations constrain plant growth. For example, calcium (Ca) and phosphorus (P) deficiencies limit growth in many tropical areas, whereas nitrogen (N) limitations are fairly common in temperate and boreal systems. Countless research studies on nutrient availability and cycling processes further confirm the importance of site-specific assessments to fully understand the effects of ecosystem disturbance on soil nutrient status (there is no magic or one-size-fits-all solution). In this regard, forest fertilization studies provide a useful mechanism to evaluate site-specific nutrient deficiencies, postharvest changes, and the relationship between soil nutrients and tree growth.

Table 3.1 Mean soil carbon ($\pm$ SE) and changes in bulk density for several long-term soil productivity locations immediately after harvesting and at year 5.

Location	Carbon (Mg ha^{-1}) ^a		Bulk density change from preharvest (%) ^b	
	Immediate postharvest	Year 5	Immediate postharvest	Year 5
Idaho	19.7 (2.1)	19.3 (0.7)	22	19
Louisiana	16.4 (0.2)	16.5 (0.3)	2	– 4
North Carolina	12.0 (0.4)	28.2 (1.7)	24	9
British Columbia	29.4 (0.9)	32.5 (1.9)	13	15

^aData from Sanchez et al. (2006).

^bData from Page-Dumroese et al. (2006).

3.4 Forest fertilization

Soils in the US (Jokela et al., 1991a; Hopmans and Chappell, 1994; Littke et al., 2014), Chile (Donoso et al., 2009), Brazil (Campoe et al., 2010), Australia (Turner, 1983), and many other locations around the world have benefitted from fertilizer additions for increasing stand growth and value (summarized in Binkley et al., 1995). For example, Albaugh et al. (2007) reported that about 0.567 million ha of pine plantations in the southeastern US have been fertilized annually since 1999. Depending upon the region and soil, N applications alone or in combination with P have been commonly applied at planting or mid-rotation age. On some sites potassium (K) and even micronutrients [i.e., manganese (Mn), copper (Cu)] may limit stand growth after N and P demands have been met (Jokela et al., 1991b; Mandzak and Moore, 1994; Jokela, 2004; Kyle et al., 2005; Vogel and Jokela, 2011; Carlson et al., 2014).

Fertilization resulted in short-term soil changes that minimally altered soil C levels in some ecosystems (Johnson and Curtis, 2001; McFarlane et al., 2009; Nave et al., 2010) but not in others (Harding and Jokela, 1994). This reflects site-specific factors such as background soil fertility and the short-term nature of most studies that do not typically quantify all ecosystem nutrient pools (e.g., soils, vegetation, and forest floor). Furthermore, Vogel et al. (2011) found that fertilization with macro- and micronutrients increased aboveground C relative to controls plots by 40% for loblolly (*Pinus taeda*) pine and 20% for slash pine (*Pinus elliottii*), but only had a marginally significant impact on soil C. Harding and Jokela (1994) reported that, after 25 years, the application of P fertilizers to a P-deficient site increased aboveground biomass and litter accumulation by about 1.5-fold over the untreated control but had no significant effect on soil OM to a depth of 91 cm. McFarlane et al. (2009) examined ponderosa pine (*Pinus ponderosa*) forests of varying fertility and reported that soil C and N increased after fertilization as a function of site quality, with the strongest response on poor sites. Therefore it is important to recognize that fertilization is generally for the benefit of trees, not soil.

3.5 Soil compaction

Soil compaction is often the most prevalent issue that arises from forest harvest operations (Batey, 2009). Compaction from trafficking reduces air-filled pores, thereby increasing soil bulk density (Johnson et al., 2007). Increased bulk density leads to reduced water infiltration, water-holding capacity, and hydraulic conductivity leading to waterlogging or standing water on flat terrain or surface runoff and erosion on steeper slopes (Waggenbrenner et al., 2016; Cambi et al., 2015; Hartemink, 2008). Although all soils are highly susceptible to compaction, forest

soils, in particular, with their loose, friable structure and high porosity are particularly at-risk for long-term impacts.

Fine-textured soils and those with low bulk density are more prone to compaction than coarse-textured or higher bulk density soils. It is not uncommon to find compaction in fine-textured soil deep in the mineral soil (50 cm; [Hakansson, 1985](#); [Page-Dumroese et al., 2006](#)) and persisting for decades or centuries ([Hakansson et al., 1988](#)). Soil OM on the soil surface (slash, forest floor, or mineral soil OM) may reduce the susceptibility of soil to compaction by increasing the resistance to deformation. Ruts are a singular and troubling form of compaction often found in the middle or the sides of skid trails. Typically, the bottom of ruts is sealed, keeping water on the soil surface and creating preferential flow paths that move water and soil downslope.

>Results from the LTSP study showed relatively rapid recovery from harvest-caused compaction in coarse-textured soils, particularly at the soil surface (0–10 cm). Slower recovery was common for fine-textured soils ([Table 3.1](#); [Page-Dumroese et al., 2006](#)) and may take as long as four decades to recover to predisturbance levels ([Froehlich et al., 1985](#)). Overall, climatic regime; soil texture; and the degree, extent, and severity of compaction determine the rate of recovery ([Page-Dumroese et al., 2006](#)). LTSP study sites with coarser-textured soil and warmer temperatures (Louisiana and North Carolina) had much greater recovery of soil bulk density within 5 years as compared with finer-textured soils in cooler climatic regimes, despite some freeze-thaw activity. Interestingly, on 183 clear-cut harvested units on the Kootenai National Forest in northwestern Montana (US) with soil compaction considered detrimental to forest productivity in 1992, 86% of these soils had recovered to predisturbance levels by 2013 ([Gier et al., 2018](#)). Soil texture was an important factor in this study, as soils that did not recover were fine-textured.

The consequences of soil compaction can be extreme. Changes in gas exchange, loss of OM, altered root growth, decreased microbial communities, modified greenhouse gas exchange, and ultimately a loss in aboveground growth and production are possible outcomes ([Batey, 2009](#)). Further, the time needed for recovery varies depending on the extent of soil physical, chemical, and biological changes induced during the harvest operations and local site factors such as slope, aspect, and climatic conditions. Therefore avoiding or reducing soil impacts during management activities has been recognized as an increasingly important step to maintain forest growth ([FAO, 2004](#)).

Degradation of soil properties can be expected during timber harvest operations and subsequent site preparation, although the impact on soil productivity depends on climate regime, soil texture, plant species, and severity of soil changes ([Powers et al., 1998](#)). Therefore soil data that focus on (1) baseline soil conditions, (2) the interrelationships of soil type and soil function, and (3) the relationship of soil disturbance severity will help determine how management-induced changes can alter above- and belowground forest productivity. As shown in the following sections, many soil properties are resilient to land management

practices while others are more sensitive, with the extent of their response is often dictated at a site-specific level (Burger, 1997).

3.6 Management impacts on soil properties

3.6.1 Harvest operations

There are few definitive studies that show soil productivity has declined under forest management (Morris and Miller, 1994; Powers, 1990) because site-specific declines in tree growth are often complicated by factors other than—or in addition to—soil disturbance, such as absence of repeated burning (Sheffield et al., 1985) or variable weed competition associated with a redistribution of OM in windrows (Stransky et al., 1985). However, many anecdotal findings from around the world also suggest that management can reduce site productivity through changes in soil properties, especially OM content (Powers, 1990; Walmsley et al., 2009).

3.6.1.1 Precommercial thinning

Precommercial thinning to decrease the risk of wildfire can impact soil physical properties (compaction) or soil fertility (nutrient removal), which may result in reduced residual tree growth or susceptibility to insects or disease attacks. Recent changes in land management methods, particularly in the western US, now favor precommercial thinning to restore fire regimes, watersheds, and wildlife habitat (Jiménez et al., 2011; Churchill et al., 2013). Thinning is often used to restore ecological function in naturally regenerated stands (Edmonds et al., 1989; Brown et al., 2004; Han et al., 2006) and usually generates a significant amount of biomass that must be disposed of to reduce wildfire risk (McKeever and Falk, 2004). Thinning residues can be transported to a bioenergy facility, dispersed across the harvest site by masticating, converted to biochar, or piled and burned (Jones et al., 2010; Creech et al., 2012), each of which can impact soil productivity (Callaham et al., 2012; Scott and Page-Dumroese, 2016). In the US, biomass from precommercial thinnings is usually piled and burned because there are few markets for this material. However, burn piles can be large and may create air quality problems (e.g., smoke and particulates) or degrade soil properties when burned. Local (in- or near-harvest site) production of biochar from nonmerchantable woody residues is an appealing alternative from a soils perspective and has received growing attention in recent years as a method to sequester C, reduce open burning, and avoid soil impacts from pile burning (Busse et al., 2013; Page-Dumroese et al., 2017).

Periodic soil disturbance (e.g., from repeated entries as for thinning) can impact tree growth (Sanchez et al., 2006; Page-Dumroese et al., 2010; Jurgensen et al., 2012) depending on soil type, tree species, and climatic regime (Henderson, 1995; Grigal and Vance, 2000). For example, when various thinning rates were implemented over a 57 year period in a northern Minnesota (US) red pine (*Pinus resinosa*) stand growing on a sandy soil, soil compaction increased

with thinning intensity. In contrast, no evidence of soil compaction was found on a rocky, loam soil in a northern Wisconsin (US) sugar maple (*Acer saccharum*) stand that was periodically thinned over a 50 year period (Tarpey et al., 2008). Increased removal of thinned biomass for bioenergy, rather than leaving the residues on the soil surface, can exacerbate short-term nutrient and C losses (Sinclair, 1992), increase soil temperature, or decrease water availability (Harvey et al., 1976; Covington, 1981; Wang et al., 2018). For example, in young balsam fir (*Abies balsamea*) stands in Quebec, Canada, increased soil temperatures after thinning increased cellulose decomposition and nutrient availability (Thibodeau et al., 2000). In a Chinese pine (*Pinus tabulaeformis*) plantation, increased wood decomposition in the mineral soil and on the soil surface was noted after thinning to restore ecosystem functions (Wang et al., 2018). However, Jang et al. (2016) found few impacts on soil properties 50 years after biomass removal and prescribed burning in a western Montana mixed conifer stand. Thinning could also negatively impact soil productivity by providing fuel for uncharacteristically severe wildfires if too much biomass is left on-site (Page-Dumroese et al., 2010).

3.6.1.2 Salvage logging

Salvage logging removes dead and damaged trees left after wind storms, wildfire, drought, disease, or insect attack. This management practice captures economic value and lowers the risk of high-severity ground fires by reducing the amount of coarse wood “jackpiled” on the soil surface (Simon et al., 1994; Passovoy and Fulé, 2006). Such “hot spots” have the potential to kill regeneration (Kemp, 1967) and decrease soil OM and C stocks (Page-Dumroese et al., 2015). Salvage logging can also reduce biodiversity, degrade wildlife habitat, or riparian conditions (Noss et al., 2006). Since these operations are often more intensive and widespread than traditional forms of logging, they exacerbate the amount and severity of soil disturbance following wildfire (McIver and Starr, 2000; Van Nieuwstadt et al., 2001), as is shown in Fig. 3.1. Post wildfire salvage logging can also increase soil erosion, flooding, and impacts on water quality (Lewis et al., 2018), but careful placement of logging debris on skid trails may reduce sediment loss (Spanos et al., 2005).

The season of logging is an important consideration for limiting undesirable soil effects. Soil disturbance considered detrimental to forest growth is typically nominal when conducted on frozen (winter) or dry (summer) soils using equipment matched to the site (Reeves et al., 2012). These two factors (harvest season and equipment used) are relatively easy options to limit the amount, extent, and severity of soil disturbance (McIver and Starr, 2000).

3.6.1.3 Tethered logging

Ground-based harvesting is usually restricted to slopes no greater than 35%, as anything steeper relies on manual felling and raises concerns for human safety (Sessions et al., 2017). In addition, disturbance on steep slopes may increase surface soil displacement, resulting in a greater likelihood of severe erosion and a



FIGURE 3.1

Extensive ruts and compaction after salvage logging a harvest unit killed by beetles.

Photo credit: USDA Forest Service.

decrease in LTSP. Newer logging systems, such as cable-assisted or “tethered” logging operations which uses a wire rope anchored upslope to assist traction (Fig. 3.2), are now being used to harvest steep slopes in mountainous areas of North America, New Zealand, and Europe to reduce soil disturbance (e.g., Sessions et al., 2017; Belart et al., 2019). While equipment slippage may be minimized in tethered logging, compaction (increased ground pressure) or soil displacement can still occur (Visser and Stampfer, 2015). For example, stream and soil disturbance in Oregon and Washington, US conifer stands were greater after tethered logging than after conventional logging operations, although overall detrimental soil impacts were below regulatory thresholds (Chase et al., 2019).

3.7 Harvest operations considerations

- Maintaining organic horizons during and after harvest operations is important and having a known baseline of soil and forest conditions to compare postharvest conditions is essential.



FIGURE 3.2

Tethered logging system in which the back of the harvester is attached to a stump or other equipment.

Photo credit: USDA Forest Service.

- Single-entry thinning operations likely have few impacts on soil sustainability. However, monitoring of soil compaction and OM displacement is encouraged where repeated stand entries are expected.
- Forest residues are important as a long-term soil OM source, whereas forest floor material (litter and duff) is important for short-term turnover of OM. Both types of OM source should be conserved, with the specific amounts dependent on site characteristics and economics.
- Soil compaction is a threat to soil and site sustainability and should be minimized. Selection of the appropriate harvest equipment, season, and slopes will determine the areal extent and depth of compaction and/or ruts.

3.8 Postharvest site preparation

3.8.1 Residue management

Logging residues left following harvests are a variable but important source of soil OM. For example, in the southeastern US approximately 12% of total pine

stand volume was left after clear-cut harvesting in Alabama (Schultz, 1997), but in California, US, mixed conifer and broadleaf stand approximately 20%–40% of the stand volume remained as residues (Kizha and Han, 2015). This variability is important to understand and highlights that the amount and distribution of residues at a given site will depend on desired species composition, soil properties, physiographic province, past management practices, harvesting method, understory competition levels, and economics (Nelson et al., 2013). Knowing how much large wood should remain is critical for long-term OM inputs which benefit water-holding capacity and ectomycorrhizal fungi (Jurgensen et al., 1997) and can determine the flora and fauna trajectories of forest and soil restoration activities (Berkowitz, 2013). For example, Harvey et al. (1976) recommended leaving 25–37 Mg ha⁻¹ of large woody material on the soil surface in cool-moist ecosystems to build mineral soil OM over time, but the amounts of coarse wood left on-site could range from 10 to 125 Mg ha⁻¹ (Jurgensen et al., 1997). Understanding preharvest conditions and the desired stand conditions will dictate the amounts left behind.

The successful establishment of a healthy forest with sustained biodiversity and C sequestration usually requires postharvest management of large-diameter (> 7.5 cm) woody residue (Liu et al., 2012; Nave et al., 2019). Depending upon regional preferences and policy, residual logging slash can be left or chipped (masticated) on-site, piled and later burned (Fig. 3.3), or removed for bioenergy production where facilities exist. Residue management generally improves future stand yields and economic



FIGURE 3.3

Slash pile with tree boles up to 6 cm in diameter formed after precommercial thinning a lodgepole pine (*Pinus contorta*) stand in Montana.

Photo credit: USDA Forest Service.

returns, as seedling growth and survival are increased from greater water and nutrient availability and enhanced root growth (Morris and Lowery, 1988; Allen et al., 1990; Lowery and Gjerstad, 1991; Brown et al., 2003). But, caution is advised and the use of appropriate equipment and site preparation methods are encouraged to avoid impacting site productivity by compacting the soil or removing a large amount of organic residues off-site. For example, the placement of nutrient-rich surface litter and mineral soil into adjacent windrows is no longer practiced in the US because of the negative impacts on soil productivity (Morris and Pritchett, 1983).

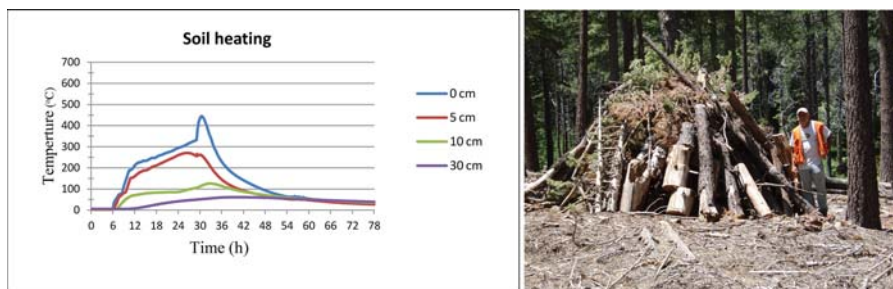
3.8.1.1 Mechanical site preparation

Intensive forest management practices have increased the use of heavy machinery for improved postharvest seedling establishment and growth (Gent et al., 1984). Mechanical treatments used prior to planting may include disking, roller drum chopping, and subsoiling. These methods are effective at restoring bulk density in the surface 0–15 cm of mineral soil, but equipment pressure may extend compaction below this depth (Reaves and Cooper, 1960) and away from the skid trail (Solgi et al., 2020). Subsoiling may be used on soils having high mechanical resistance (compaction), or with soils having cemented horizons that reduce root penetration, growth, and occupancy within the lower mineral soil profile (Löf et al., 2012). Chopping breaks up existing woody vegetation and crushes the logging slash into smaller pieces to enhance the effectiveness of prescribed burning (Knoepp et al., 2004) and decomposition (Gholz et al., 1985). Similarly, disking may be used to ameliorate surface soil compaction along skid trails, loading ramps, and retired forest roads. As with any surface soil disturbance treatment, an evaluation of the site's erosion potential must be integrated into the prescription process.

3.8.1.2 Pile burning

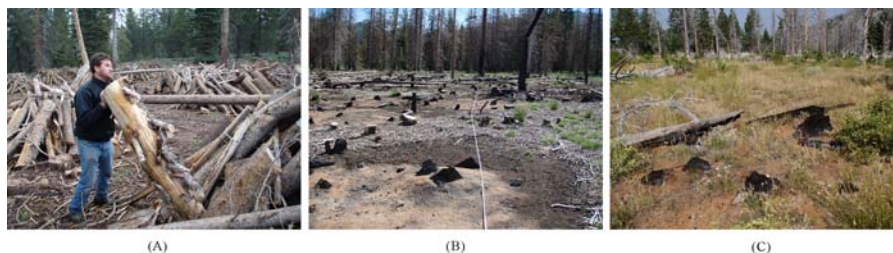
Slash pile burning is a traditional practice used throughout much of the world for fuel reduction and disposal of harvest residues prior to planting. Nonmerchantable woody residues are moved into piles which can range from small, hand-built piles to imposingly large, tractor-generated piles on log landings, and then burned on-site. This practice is recognized for its effectiveness, low cost, limited risk of fire escape, and simplicity (Rhoades and Fornwalt, 2015). The fuel composition of piles, the amount of soil contained in a pile, and the pile structure and moisture content all influence burn characteristics, and collectively dictate the amount of soil damage that may occur (Massman et al., 2008). Excessive heat from burning large-diameter piles (> 15 m), particularly those that contain large-diameter wood, can have large impacts on soil properties (Busse et al., 2013, Smith et al., 2016), as can slow, smoldering fires in poorly cured (wet) piles or in piles containing excessive soil that traps heat and increases the downward heat pulse.

Loss in soil quality is a concern beneath pile burns (Korb et al., 2004; DeSandoli et al., 2016). Soil properties are modified in proportion to fire severity, maximum soil temperatures, and heat duration reached during burning.

**FIGURE 3.4**

Temperature and heat duration extremes during burning of a moderate-sized pile containing a mixed composition of conifer fuels.

Photo credit: USDA Forest Service.

**FIGURE 3.5**

Long-term recovery after severe pile burning on a moist site. Example of a site (A) with many handpiles before burning, (B) 8 months after burning, and (C) 8 years postburning.

Photo credit: USDA Forest Service.

Established thresholds for soil change are typically based on maximum temperature alone, as determined in controlled-setting experiments (Hungerford et al., 1991), whereas the effects of heat duration are poorly quantified. Temperatures that result in the destruction of soil OM, microbial death, or nutrient transformation are often exceeded in the surface mineral soil, even when burning small piles (Busse et al., 2010; 2013). However, since soil is a porous matrix and a poor conductor of heat, extreme heat does not travel much below 5–10 cm in the profile (Fig. 3.4), suggesting that any changes in soil properties will mostly be surficial. Soil moisture content at the time of burning is also a key consideration; moist soils are more effective at quenching the downward heat pulse than dry soils and produce lower maximum temperatures of shorter duration and, consequently, less soil damage. Further, moist sites have been shown to recover vegetation cover fairly rapidly following severe pile burning (Fig. 3.5).

Recent studies have identified a variety of post-fire soil effects, yet they have given few definitive recommendations other than concluding that small-

diameter piles are generally benign across landscapes (Tarrant, 1956; Meyer, 2009; Hubbert et al., 2015; Jang et al., 2017). In comparison, soil damage from moderate-sized, tractor-generated piles (5–15 m diameter) is more difficult to generalize and is probably best considered on a site-specific basis that considers the areal extent of pile coverage within a treatment unit (i.e., do the piles occupy a large proportion of the ground surface?). Several studies with moderately large piles showed declines in total soil C and N in the surface 0–15 cm soil layer with concomitant increases in available N (Korb et al., 2004; Esquilín et al., 2007; Rhoades et al., 2015), whereas detrimental effects on soil microbial communities were shown by some (Korb et al., 2004; Esquilín et al., 2007), but not all studies (Smith et al., 2016). York et al. (2009) measured plant growth—a general indicator of sustainable soil health—and found greater 10-year conifer growth within the perimeter of moderate-sized pile scars compared to adjacent unburned sites, which they attributed to fire’s positive effect on long-term soil N availability.

3.9 Drainage and bedding

Stand growth and survival are constrained by excessively wet soils, as are access for harvesting and other silvicultural operations. As a result, forest drainage (ditching) became a common mechanical site preparation treatment on both mineral and organic soils in the southeastern US from the 1930s through the mid-1980s (Skaggs et al., 2016). Treated sites were typically very poorly drained (e.g., pocosins), had precipitation amounts that exceeded evapotranspiration rates, and had natural drainage systems that were inadequate for removing excess water. Saturated hydraulic conductivity represents an important soil property influencing drainage, and was used to guide the establishment and spacing between drainage ditches. Growth responses of planted pines were large following drainage, ranging from 80% to almost 1300% (Terry and Hughes, 1975). Yet, despite research conducted in the 1950s–70s that demonstrated the positive impacts of soil drainage on tree growth and site accessibility (Miller and Maki, 1957; Maki, 1960; White and Pritchett, 1970; Terry and Hughes, 1975), the expansion of drainage practices ended in the US in 1990 because of federal wetland protection regulations on jurisdictional wetlands (Fox et al., 2007a; Skaggs et al., 2016). However, forest operations still accrue benefits from existing, “historical” drainage systems through ditch maintenance, replanting, and nutrient management efforts. Today, bedding is now commonly used on such sites to modify the microsite and aid stand establishment and survival.

Raised beds (mounds), commonly made with a bedding plow (Fig. 3.6), are used to improve soil microsites, control competing vegetation, and increase seedling growth (McKee and Shoulders, 1970; McKee and Shoulders, 1974; Crutchfield and Martin, 1983). Single or double-pass bedding operations are

**FIGURE 3.6**

Mechanical site preparation can be done (A) using a subsoiler to prepare a site for planting and (B) a poorly drained, bedded site post-thinning.

Photo credit: Eric Jokela.

conducted in the spring to mid-summer months, which allow the beds to settle before planting, as seedling mortality can occur if roots come in contact with unsettled air pockets. Contour bedding is used on sites with slopes ($> 35\%$) to reduce the risks of soil erosion and sediment transport. “Spot” (or bucket) mounding (inverting mounds of soil to create a double organic layer within the mound; [Londo and Mroz, 2001](#)) is used rather than continuously mounded rows, thereby reducing runoff while increasing nutrient retention at the planting site ([Costantini and Loch, 2002](#)). Bucket mounds are also common in Scandinavia, Canada, and the upper Great Lakes, US region ([Takyi and Hillman, 2000](#); [Londo and Mroz, 2001](#)).

Bedding is commonly used to increase tree survival and growth on poorly drained soils characteristic of many lower Coastal Plain sites in the southern US and wetland sites in the northern US ([Gent et al., 1984](#); [Gale et al., 1998](#)). High water tables lead to anaerobic soil conditions, which is a problem in young, developing stands that have low leaf area and correspondingly low rates of evapotranspiration. Bedding increases soil aeration by mixing surface organic horizons with mineral soil ([Gale et al., 1998](#)). When bedding is combined with burning and harrowing it can also increase nutrient release. For example, this combination of site preparation treatments resulted in a ten-fold increase in soil solution P as compared to burned and chopped site preparation ([Gale et al., 1998](#)), and it also resulted in increased K ([Burger and Pritchett, 1988](#)). Bedding has also been used in the interior western US and Australia as a method to increase soil water-holding capacity ([Cornish, 1993](#); [Page-Dumroese et al., 1997](#)) on drier sites, where it can increase soil OM, cation exchange capacity, N, and decrease soil bulk density. In South Carolina, US soil bedding increased wood decomposition by both microorganisms and termites and this impact on OM content can also alter site sustainability ([Jurgensen et al., 2019](#)).

3.10 Herbicide applications

Herbicides are used to control competing vegetation, including unwanted invasive plants, and their application is a common practice in plantation forestry. Over the last several decades, the evolution and use of more intensive silvicultural practices across forested landscapes in the US and Europe (e.g., including the use of genetically improved growing stock, site preparation, fertilizer additions, and understory competition control) have contributed to substantial gains in growth and forest productivity (Fox et al., 2007b; McCarthy et al., 2010). However, the impacts of herbicides on soil and ecosystem processes are variable, but important for understanding their long-term effects on forest development and sustainable soil management practices.

Direct effects: Commonly used forest herbicides are reasonably plant specific and generally act by disrupting or modifying photosynthesis (e.g., hexazinone, atrazine), protein synthesis (glyphosate, imazapyr, sulfometuron methyl), or plant hormone production (triclopyr, 2,4-D). Therefore most of these compounds have a little lethal effect on soil organisms (Busse et al., 2004). For example, increased soil bacterial growth was found when glyphosate, one of the most widely used compounds around the world (Thompson and Pitt, 2003), was added at high concentrations. This result was because organisms metabolize soil-bound glyphosate as a source of C, N, and P (Fig. 3.7; Ratcliff et al., 2006). Most evidence to date points to the tolerance and resilience of soil organisms to traditional forestry herbicides, although scrutiny is still necessary as new products come on the market.

Indirect effects: Herbicide application can produce lasting changes in plant community structure and tree growth, which could be a factor in the retention of soil nutrients and, therefore, in determining their suitability for sustainable of

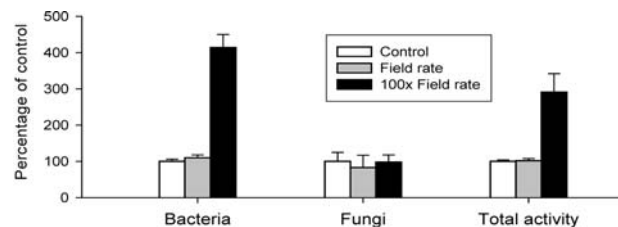


FIGURE 3.7

Effect of glyphosate application on soil bacteria (total number), fungi (hyphal length), and total activity (respiration). Measurements were taken 30 days following glyphosate application to a ponderosa pine plantation soil in northern California.

Based on results from Ratcliff, A.W., Busse, M.D., Shestak, C.J., 2006. Changes in microbial community structure following herbicide (glyphosate) additions to forest soils. *Appl. Soil Ecol.* 34, 114–124.

forest management systems (Vogel et al., 2011). In southern US pine forests, complete and sustained understory competition control were associated with decreased soil N and C content (Echeverria et al., 2004; Sarkhot et al., 2007; Sartori et al., 2007; Rifai et al., 2010; Vogel et al., 2011). This could potentially impact nutrient retention and long-term stand productivity. Loss of soil C under intensive weed control treatments has been attributed to reductions in woody and herbaceous fine root growth and turnover (Albaugh et al., 1998; Shan et al., 2002). Subedi et al. (2017) examined the inter-rotational effects of sustained weed control (i.e., repeated herbicide treatments during the first 5 years of stand establishment) on the richness, diversity, and composition of understory plants in young loblolly pine stands growing on low-fertility Spodosols in North Central Florida, US. The competition control treatment used in the first rotation effectively reduced the shrubby understory component in the second rotation. The absence of shrubby species such as gallberry (*Ilex glabra*), a dominant manganese (Mn) accumulator (Jokela et al., 1991a; Subedi et al., 2014), may affect nutrient cycling and accentuate subacute deficiencies of this and other nutrients and influence site productivity. However, because the silvicultural treatments used in that study were intensive (sustained weed control and fertilization), the authors noted that soil responses would likely be less pronounced under operational treatments (Subedi et al., 2017).

Post-fire herbicide use, while beneficial to stand establishment and growth, may lead to increased soil erosion and sedimentation rates by reducing plant recruitment and total plant cover (Certini, 2005). Consequently, it is critical to evaluate site erosion potential and factors such as slope, soil cover, anticipated storms, and proximity to rural communities to determine if herbicide applications to remove soil cover will have unintended impacts. Furthermore, the long-term effects of vegetation control on soil sustainability have not been clearly documented, but the LTSP study suggests that there could be a moderate impact of vegetation control on soil nutrients (Busse et al., 1996, 2001; Ponder et al., 2012; Zhang et al., 2013).

Whether herbicides substantially modify forest soil properties or processes depends on the interactions of site, herbicide, and soil characteristics (Nelson and Cantrell, 2002). Soil responses will change depending on herbicide chemistry, soil-herbicide binding potential, herbicide application rate and frequency, vegetation type, soil fertility, and site productivity (Minogue et al., 1991). Clearly, on the one hand, herbicide control of competing vegetation benefits seedling survival and tree growth and may be an important tool for reducing wildfire risk where grasses or shrubs form a dominant understory layer (Ponder et al., 2012; Zhang et al., 2013). On the other, herbicide movement in subsurface water leading to potential contamination of water bodies, increased soil erosion due to loss of vegetation cover, and long-term changes in soil fertility due to a shortfall of understory roots and litter input may lead to unwanted changes to the forest environment.

3.11 Site preparation, bedding, and herbicide considerations

- Determine which site preparation treatments will cause a minimal amount of soil disturbance while still achieving management goals.
- Soil bedding (mounding) can increase planted tree growth by improving soil aeration, concentrating soil OM, and increasing nutrient availability.
- Small burn piles have fewer soil impacts than large or moderately sized piles. Burning when soils are moist will help lessen soil heating and potential damage beneath piles.
- Herbicides reduce competition and increase tree growth, but their indirect effects may alter C pools erosion potential, soil fertility, and water quality.

3.12 Soil sustainability monitoring

As we have pointed out, forest soils are heterogeneous and their responses to harvesting and site preparation are varied. This makes soil monitoring an integral part of forest management so that the data can inform adaptive management by contributing an understanding of the limits of a particular soil to particular management activities and the efficacy of treatments (DeLuca et al., 2010). But, as also previously noted, solid baseline data are needed to link soil type and function changes with management and determine if thresholds indicating a drop in site productivity have been crossed. Monitoring soil sustainability represents a key consideration for both environmental policy and resource management. Soil monitoring refers to the repeated collection of soil data used to describe soil conditions at points in time, requires archiving the data (Page-Dumroese et al., 2009), and can provide information on early changes in soil quality that affect the health or growth of managed forests (Morvan et al., 2008). Within the context of a changing climate, soil monitoring data becomes even more important for proper management of forested landscapes. However, the consistent collection of monitoring data is neither inexpensive to collect nor short-term. Temporal changes associated with changes in soil C, compaction, soil structure, etc. and dynamic processes (e.g., hydrologic function) must be integrated into monitoring efforts to provide detailed soil data to promote continued harvesting or restoration of public and industry lands. Furthermore, soil monitoring can determine unintended consequences of harvest activities and provide data for returning an ecosystem to the intended trajectory (DeLuca et al., 2010).

3.12.1 Descriptive soil quality measures versus functional approaches

Soil monitoring has been made relatively easy with the use of visual indicators (Heningner et al., 1997; Curran et al., 2005; Page-Dumroese et al., 2009) that

describe soil displacement, loss of OM cover, and soil mixing, among other visual properties. These visual indicators are a surrogate for soil quality changes associated with land management. However, the change in soil quality must also be related to soil function (support of natural ecosystems, food and fiber production, a base for buildings, etc.; [Vegter et al., 1988](#)) and monitoring for these two things (quality and function) can be performed simultaneously. Managers and scientists must understand the limits of individual soils to provide these functions ([Nortcliff, 2002](#)) and, as previously noted, these functions will be site-specific ([Burger and Kelting, 1999](#)) and require site-specific validation ([Page-Dumroese et al., 2012](#)). By using a common soil monitoring protocol on public lands, it is relatively easy to determine where monitoring is needed, enlist citizen scientists in the monitoring effort, get highly detailed and statistically robust data, and conduct analyses of the data. In addition, linking these data to processes such as infiltration or nutrient leaching will provide insight into how soil functions.

3.12.2 Soil functional integrity

Soil functional integrity is the intactness of soil and native vegetation patterns and processes, which indicate whether that soil is either resistant or resilient to disturbances. Healthy soil will maintain inherent functions (e.g., water regulation, plant growth) provided by forested ecosystems ([Greenland and Szabolcs, 1994](#)), but they are variable and depend on the interaction of soil physical, chemical, and biological properties ([Schoenholtz et al., 2000](#)). Soil OM, function, and resilience are all interconnected. However, the influence of OM on soil properties depends on its spatial distribution (horizontal and vertical), composition (biochemical and functional), and microbial populations ([Herrick and Wander, 1997](#)). Therefore properly functioning soil must have intact soil physical, chemical, and biological properties that support: (1) biological activity, (2) water flow, (3) filtering and detoxifying organic and inorganic materials, and (4) storing and cycling nutrients ([Seybold et al., 1999](#)).

3.12.3 Ecosystem stability as measured by soil indicators

As we noted previously, most ecosystems will return to predisturbance given enough time and the right climatic, site, and soil conditions. Ecosystem stability (return to an equilibrium after disturbance; [Redfearn and Pimm, 1987](#)) is related to the intrinsic properties of OM because both physiochemical and biological properties from the surrounding environment dictate the rate of OM decomposition. Although OM is inherently unstable, it can persist in soils for thousands of years, and it is an ecosystem property that enhances stability ([Schmidt et al., 2011](#)). Clearly, land managers influence the amount of OM left on a site, the surrounding environmental properties that govern decomposition rates, vegetative growth, and hydrologic function. Soil monitoring will inform managers about the rates at which soil OM and other soil properties are changing (declining, staying

the same, recovering). Since soil OM is often vulnerable to perturbations, an understanding of which ecosystems are resistant to OM losses or resilient (recover OM losses) are key to maintaining both soil and site sustainability.

3.13 Soil monitoring considerations

- Soil monitoring is a crucial activity to determine the long-term success of management activities.
- Repeated monitoring provides an understanding of soil changes over time.
- Policies that encourage monitoring and data archiving in a shared database facilitate data collection and prioritization of ecosystems in need of monitoring.
- Site-specific validation of monitoring data is critical for establishing linkages between soil and site productivity.

3.14 Conclusion

The responses of soil and vegetation to harvest operations and site preparation are complex, making the development of soil sustainability guidelines a balancing act. Environmental values and ecosystem services must be maintained to protect long-term productivity. Adaptive management in forest ecosystems is dependent on soil monitoring to inform the success of treatments and reduce unintended long-term consequences. In addition, long-term field trials, such as the LTSP effort, will provide data on the extent to which a site can tolerate harvesting and site preparation and not lose productivity. Much of our long-term data are based on approximately 30 years of careful measurements which means that as the stands grow we must continue to evaluate whole ecosystem responses to disturbance and repeated stand entry impacts. Of course, changing climates, equipment, and ecosystem services needed will produce varying impacts on soil OM, functions, and sustainability, but careful management can minimize soil physical, chemical, and biological impacts.

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