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Soil Health Assessment of Forest Soils

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Introduction

Forest sustainability is explicitly tied to soil health, which has been defined as “the capacity of a soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health” (Doran et al., 1996; Sigua, 2018). This definition includes the ability of soil to function effectively as a component of healthy forests (Schoenholtz et al., 2000) and is linked to the soils ability to support physical, chemical, and biological properties while also suppressing plant pathogens (van Bruggen and Semenov, 2000). In broad terms, forest soil health can be defined as a capacity for water retention, carbon (C) sequestration, and plant productivity, or it could simply be defined as the ability of the soil to produce biomass (Schoenholtz et al., 2000). For forested ecosystems to be sustainable, soil health must be maintained. Forest soil health is linked with the amount and composition of surface and mineral soil organic matter (SOM; Harvey et al., 1979; Harvey et al., 1981). In fact, the U.S. Department of Agriculture (USDA) Forest Service requirement to leave 25 to 27 tons ha⁻¹ of coarse woody material greater than 14 cm in diameter. comes from the need to provide ‘parent material’ for decayed wood in many forest ecosystems (Harvey et al., 1981) and ensure a healthy population of ectomycorrhizal fungi. Although mineral SOM is a small fraction of mineral soil mass (1–5%), it is responsible for a majority of soil physical, chemical, and biological properties through plant litter and anthropogenic inputs (Liang et al., 1998; Six and Jastrow, 2002). Since SOM

also improves soil health, it also increases the chances for successful restoration after disturbance (Hagen-Thorn et al., 2004).

Forest and agroforest soils provide many ecosystem services including timber, clean water, flood control, and biodiversity, but maintaining soil health is difficult because of numerous stressors (*i.e.*, climate change, air pollution, altered water tables, intensive harvesting and site preparation, wildfire, invasive species, and overgrazing). No single forest soil health indicator is adequate because changes in one property will likely influence others. Therefore, using a variety of chemical, physical, and biological indicators (properties), land managers can better understand the impacts of stand- and watershed-scale manipulations, temperature and moisture variability, deep soil processes, and invasive species on soil health.

Evaluating forest soil health is difficult because soils are dynamic systems influenced by physical, chemical, and biological properties that are quantifiable using several appraisal techniques, many already being used to assess soil health. For example, the USDA Forest Service Forest Inventory and Analysis (FIA) program collects soil data during its inventory of the Nation's forest resources. Furthermore, many national forests use the Forest Soil Disturbance Monitoring Protocol (Page-Dumroese et al., 2009) to collect short- and long-term data on changes in soil physical attributes after land management, but routine measurements of multiple soil health indicators can be expensive. Therefore, remote sensing (Chaudhary et al., 2012) is often combined with in-field sensors to substitute for more expensive laboratory testing of physical, chemical, and biological properties (*e.g.*, Hemmat and Adamchuk, 2008; Sudduth et al., 2013). Recently, the Comprehensive Assessment of Soil Health (CASH) approach has been used in the eastern U.S. to measure 15 physical, biological, and chemical indicators using a scoring system (Fine et al., 2017). These efforts, and many others, are providing the baseline data needed to test and assess both soil- and ecosystem-health. Currently there is no universally accepted protocol for assessing soil health, but Table 6.1 lists several key soil chemical, physical, and biological properties that are widely used, with some being static (point-in-time) measures and others dynamic (process level) measures.

There are many different indicators that can be used to assess soil health, but those that are simple, easy to measure, relatively rapid to use, cover the largest number of soil types, and sensitive to environmental changes and land management are the most desirable (Doran and Zeiss, 2000; Knoepp et al., 2000). Herein, we discuss how soil health is being assessed in complex agroforest, tropical and temperate ecosystems. Additionally, we present a national perspective using FIA protocols.

Table 6.1 Examples of soil physical, chemical, and biological properties that are used to assess temperate, agroforest, and tropical forest soil health.

Indicator*	Reference	Comment
<i>Soil Physical Properties</i>		
Visual assessment of surface soil changes	Page-Dumroese et al., 2009	Rapid forest soil disturbance monitoring protocol
Soil compaction	Shestak and Busse, 2005;	Soil compaction linked to biological processes
Aggregate stability	Herrick et al., 2001	Rapid field assessment kit
Porosity	Schoenholtz et al., 2000; Udawatta et al., 2006	Including texture, aeration, runoff, infiltration, water holding capacity
Coarse fragments	Page-Dumroese et al., 1999; Jurgensen et al., 2017	Importance of coarse-fragments for calculating nutrient pools and supporting logging equipment.
Water holding capacity	Schoenholtz et al., 2000	Determines water flux, erosion, runoff, infiltration, storage
<i>Soil Chemical Properties</i>		
Active C	Page-Dumroese et al., 2015	Rapid field test
Organic C	Harris et al., 1996	Specific scoring functions for plant productivity
	Busse et al., 2006;	Changes in fungal and bacterial biomass
Organic matter	Sanchez et al., 2006a	
	Gregorich et al., 1994; Laik et al., 2009; Wang and Wang, 2007	Soil organic matter pools respond to changes in plant productivity, climate, and land use
Nutrients		
Nitrogen (organic and mineral)	Doran and Parkin, 1994	A primary indicator of soil health
Base cations (e.g., Calcium, magnesium, potassium) and Cation Exchange Capacity	Merilä et al., 2010	With linkages to plants and soil microbial communities
Integrated physical and chemical measures	Amacher et al., 2007	Integrates 19 measured physical and chemical properties into a single ‘vital sign’ of overall soil quality.

Table 6.1 (Continued)

Indicator*	Reference	Comment
Salinity (electrical conductivity)	Doran and Parkin, 1994	Basic indicator of soil health
Soil biological Properties		
Decomposition of standard substrates	Jurgensen et al., 2006; González et al., 2008	Index of organic matter decay as influenced by biotic and abiotic factors
Fauna	van Straalen, 1998; Knoepp et al., 2000; González and Seastedt, 2001	Bioindicator of soil health
PFLA, DNA or RNA-based techniques	van Bruggen and Semenov, 2000	Microbial diversity and function, species richness, disease suppression
Microbial techniques combined with organic matter and nutrient analyses	Arias et al., 2005	

*Linkages to forest soil health are too numerous to list, only a select few are noted here.

Ecosystem Examples

Agroforestry

Agroforestry (AF) is an intensive land management practice where trees and shrubs are integrated into crop and/or livestock management practices to optimize numerous benefits arising from biophysical interactions among the components (Gold and Garrett, 2009). Five main AF practices are: riparian buffers, alley cropping, windbreaks, silvopasture, and forest farming. Riparian buffers exist around water bodies while upland buffers are mostly located on contours to create alley cropping. Windbreaks protect crops, livestock, and farm structures from wind and snow. Silvopasture is the integration of trees, forage, and livestock and is designed to produce a high-value timber product, while providing short-term cash flow from livestock (Klopfenstein et al., 1997). Furthermore, AF practices were approved by both the afforestation and reforestation programs and under the Clean Development Mechanisms of the Kyoto Protocol for C sequestration (IPCC, 2007; Watson et al., 2000; Smith et al., 2007). However, current literature lacks information on the role of AF practices on soil health. This section will highlight benefits of AF practices on soil health parameters including soil C, physical, biological, and chemical soil properties and a soil's capacity to degrade harmful chemicals and promote biodiversity.

Carbon Sequestration

A decrease of soil C causes degradation of soil health that could lead to a food insecurity and declining ecosystem sustainability (Godfray et al., 2010; Montgomery, 2010). Agroforestry practices increase soil C and reduce greenhouse gases (Schoeneberger et al., 2012a; Udawatta and Jose, 2012; Stefano and Jacobson, 2018) because perennial vegetation stores more C in above- and below-ground biomass, soil, living and dead organisms, and root exudates (Cairns and Meganck, 1994; Pinho et al., 2012) as compared to row crops or grazing. Since both forest and grassland C sequestration and storage patterns are active in AF ecosystems, a higher percentage of C is allocated to belowground biomass through an extended growing season (Schroeder, 1993; Kort and Turnock, 1999; Sharrow and Ismail, 2004; Morgan et al., 2010). Diverse vegetation also promotes diverse soil communities (fauna and flora), development of surface and deep roots, and reduced soil disturbance which, combined, enhance C sequestration potential (Udawatta et al., 2009; Kumar et al., 2010; Paudel et al., 2011; Udawatta and Jose, 2012). In addition, SOM concentrations are greater at the soil surface (0–15 cm) and near the base of trees as compared with soil located greater distances from perennial vegetation or deeper in the soil profile (Seiter et al., 1995; Sauer et al., 2007; Fig. 6.1). Brandle et al. (1992) estimated that 22.2 metric tons of

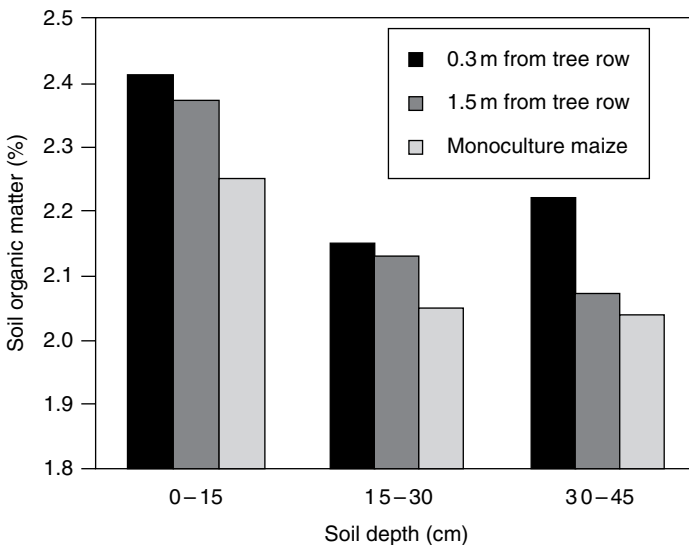


Figure 6.1 Soil organic matter percentage decreased with increasing distance and depth from tree rows for a 4-year old red alder-corn alley cropping system in western Oregon, USA. (Adapted from Seiter et al., 1995).

C is stored on 1.96 million ha of shelterbelts and is a model for enhanced sequestration to mitigate climate change.

In the United States, pasture and grazing lands occupy 266 and 52 million ha, respectively with the potential to sequester as much as 516 Tg C yr⁻¹ just by converting 10% of the pasture lands to silvopasture and 10% of the crop land to alley cropping (Nair et al., 2009). Furthermore, Udawatta and Jose (2012) have estimated that silvopasture, alley cropping, windbreaks, and riparian buffers could sequester 642 Tg C yr⁻¹ in the United States (Fig. 6.2).

Soil Physical Indicators

Climate change is expected to increase the intensity of rainfall in the 21st century increasing soil erosion 16 and 58% (Nearing et al., 2004). This predicted climatic shift emphasizes the importance of soil conservation. By using AF practices, soil health can be increased through improved soil bulk density, aggregate stability, porosity, water holding capacity, infiltration, and limiting sediment movement (Seobi et al., 2005; Udawatta et al., 2006; 2009; 2011a; Adhikari et al., 2014). Aggregate stability is greater in AF soils as compared to soil under row crops or in grazed lands (Udawatta et al., 2008; Paudel et al., 2011, 2012) and can lead to a more stable SOM pool (Novara et al., 2012). Bulk density in AF sites was reduced by 2.3% after six years with a concomitant increase in porosity (Seobi et al., 2005). These changes in soil bulk density, porosity, and SOM also serve to increase infiltration, saturated hydraulic conductivity, water holding capacity, and water storage (Kumar, 2012; Akdemir et al., 2016; Alagele et al., 2018) resulting in enhanced production of food, fiber and, thus, soil health (Balandier et al., 2008; Dimitriou et al., 2009; Udawatta et al., 2011a).

Soil Biological Indicators

Soil fauna composition, microbial activity, microbial biomass, and enzyme activity are good soil health indicators that can be used to predict land management effects on water, microbes, nutrient use efficiency, and disease suppression (Bandick and Dick, 1999; Boerner et al., 2000; Schloter et al., 2003; Brussaard et al., 2007). Soil enzymes are greater in AF soils as compared to row crop and grazed lands (Mungai et al., 2006; Udawatta et al., 2009; Paudel et al., 2011) because of improved litter quality and quantity, diverse vegetation, and root exudates. In addition, a diverse microbial community can sequester eight to ten times more C than monoculture systems (Polgase et al., 2008). These changes imply positive effects on soil biochemical processes and microbial resilience which ultimately leads to greater soil health, resilience, and productivity (Rivest et al., 2013).

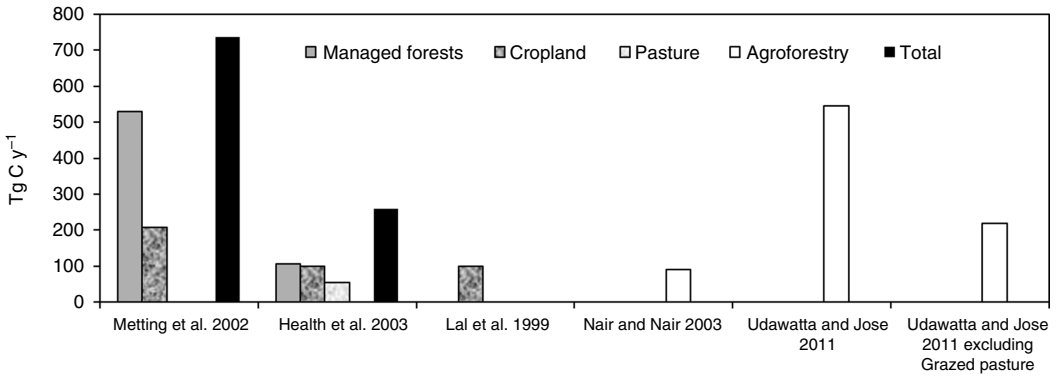


Figure 6.2 Carbon sequestration potential for various management systems in the USA (Source: Udawatta and Jose, 2012).

Soil Enrichment and Decontamination

Nutrient additions, long-term productivity, sustainability, and the reduction of water pollution and hypoxia conditions all enrich soil functions (Jose, 2009; Udawatta et al., 2009; Zomer et al., 2009; Udawatta et al., 2011b). Soil enrichment occurs through filtering of nutrients and sediment within the root zone and in the reduction of water erosion and sediment losses (Udawatta et al., 2011b; Allen et al., 2004). Agroforestry practices retain nutrients and C by filtering nutrients and sediment and reducing water erosion and these properties increase as buffer width increases (Broadmeadow and Nisbet, 2004; Schultz et al., 2009; Udawatta et al., 2011b).

There are many sources of soil contamination (e.g., mining, industrialization, rapid urbanization, herbicides, pesticides, antibiotics, personal care products) and phytoremediation is a cost effective, noninvasive, and socially preferred approach to remove environmental contaminants (Boyajian and Carreira, 1997). Fast growing tree species such as poplars (*Populous* spp) and forage grasses (e.g., *Panicum vigatum*) can produce large amounts of biomass and deep roots that can both tolerate and extract large amounts of contaminants through plant uptake (Dhillon et al., 2008; Gomes, 2012; Zalesny et al., 2019). To date, more than 400 plant species have been identified that can accumulate heavy metals. These plants remove a contaminant from the soil and accumulate the contaminants in shoots and/or roots. This helps reduce contamination in the soil and increase soil health (Paz-Ferreiro et al., 2014).

Key soil health benefits associated with changes in soil properties by AF operations are reduced water pollution, enhanced soil microbial population and diversity, and increased C sequestration which also result in healthier ecosystems and land productivity. Indicators such as bulk density, porosity, infiltration rate, and microbial diversity can help track changes with the AF ecosystems and show reduced water loss and erosion, climate change mitigation, and enhanced ecosystem resilience.

Tropical Forests

As with AF systems, tropical soil health reflects the interaction of physical, chemical, and biological components, but the relative importance of those properties differs depending on local climate and vegetation. There are two types of tropical forests: Moist/wet (2000 to > 8000 mm of precipitation yr⁻¹) and dry (several months of severe drought). Tropical forests occur about 25° north and south of the Equator and have both evergreen and deciduous tree species. In tropical forests rainfall seasonality, distribution, and variability drive soil moisture, litter accumulation and decay, soil respiration, and overall productivity. Further, the length of

the wet season will, in part, dictate the amount of SOM storage (Rohr et al., 2013). Threats to soil health in tropical systems include a changing climate, fire, hurricanes, and land conversion (Jaramillo and Murray-Tortarolo, 2019; Cusack and Marin-Spiotta, 2019).

To understand the drivers of tropical soil health, it is critical to understand the rates of decomposition and incorporation of organic material to determine the capacity of an ecosystem to sequester C and cycle nutrients important for productivity, fertility, and overall ecosystem health. In tropical ecosystems, climate may be less important than the biological regulation by soil macro-fauna (Lavelle et al., 1993; Heneghan et al., 1999; González and Seastedt, 2001). Soil macrofauna are more common in the tropics than in temperate zones while soil microfauna are more abundant in the temperate regions (González and Seastedt, 2000; González, 2002). This latitudinal variation in the types of micro- and macrofauna and their relative importance can have a significant effect on litter breakdown rates. Consequently, biological properties including the diversity of micro- and macrofauna are an important determinant of soil health in the tropics (González and Lodge, 2017). In addition, the abundance of various soil fauna also changes with latitude (Swift et al., 1979).

Environmental Gradients and Future Climate Projections

Henareh Khalyani et al. (2016) assessed different general circulation models and greenhouse gas emission scenarios of downscaled climate projections to inform future climatology and its potential impacts to tropical regions in the U.S., namely the Caribbean islands. Those projections indicate a reduction in precipitation and an increase of 4 to 9°C in air temperature. In addition, they projected a high likelihood of shifts in ecological life zones to drier conditions. The combination of decreased rainfall, increasing variability of rainfall, and higher air temperatures would lead to reductions in soil moisture and changes to SOM dynamics. Though microbial soil processes will likely adjust to changes in rainfall, additional stressors of climate change may lower microorganism diversity or productivity, thus reducing microbial pool resiliency (Silver, 1998). Consequently, tropical forest soils may be affected by the changing climate through increased variability in SOM decay and potential changes to soil biota, oxygen concentrations, and nutrient accessibility (González et al., 2013).

Additional research at the Luquillo Experimental Forest in Puerto Rico suggests that threats of a changing climate to forest soil health vary along elevational gradients. In a field soil translocation experiment, Chen et al. (2017) studied the impacts of decreasing temperature but increasing moisture on soil organic C and respiration along an elevation gradient in northeastern Puerto Rico. Soils

translocated from low- to high- elevation showed an increased respiration rate with decreased soil organic C content, which suggested that the increased soil moisture and altered soil microbes may affect respiration rates. Further, soils translocated from high- to low-elevation also showed an increased respiration rate with reduced soil organic C, suggesting that the higher temperature at low elevations enhanced decomposition rates. Thus, tropical soils at high elevations may be at risk of releasing sequestered C into the atmosphere giving a warming climate in the Caribbean (Chen et al., 2017).

In tropical forests, seasonal soil decomposition is closely tied to wet and dry cycles, suggesting that seasonal adjustments in temperature and moisture due to climate change are likely to affect decomposer communities, soil resource quantity and distribution, and litter quality (Silver, 1998). Decomposer organisms can be key determinants of decay in Puerto Rico (e.g., González and Seastedt, 2001). Yet, the contribution of different groups of decomposers to the decay of coarse woody debris, might vary among the different forest types located along elevation and environmental gradients (González and Luce, 2013). For example, González and Luce (2013) found the decay of coarse woody debris was most strongly correlated with white rot fungi in cloud forests (tropical wet forests) located at the tops of mountains (high elevation). In contrast, wood decay rates in tropical dry forests (low elevation) was related to the high diversity of species and functional groups of wood-inhabiting animals (Torres and González, 2005, González et al., 2008). Thus, the distribution of particular groups of organisms might be more important predictors of wood decay in tropical regions than climatic constraints (González, 2002, 2016).

Tropical Soil Chemical and Biological Properties

Tropical forests are places where large quantities of debris are periodically generated during tropical storms and hurricanes. Such disturbances may increase nutrient losses from the forest depending on how the debris is managed, how the microbiota responds to the disturbance, and the chemical and physical characteristics of the soil (Miller and Lodge, 1997). Canopy disturbances associated with severe hurricane storms dramatically alter the physicochemical environment and the amounts of debris deposited into the forest floor (Lodge et al., 1991; Ostertag et al., 2003; Shiels and González, 2014). In addition, canopy disturbances alter the patterns in litterfall and associated nutrient cycling (Scatena and Lugo, 1995; Lugo and Scatena, 1996); hurricane litter contains a high proportion of green leaves from which nutrients have not been translocated, thus altering the litter quality in the forest floor (Richardson et al., 2010). Cascading effects from canopy openness can account for most of the shifts in the forest

biota and biotic processes, which include increased plant recruitment and richness, as well as the decreased abundance and diversity of several animal groups (Richardson et al., 2010; Shiels et al., 2015). Opening the canopy decreases litterfall and litter moisture, thereby inhibiting lignin-degrading fungi, decreasing litter invertebrate richness, diversity, and biomass, and ultimately slowing decomposition (González et al., 2014; Lodge et al., 2014; Shiels et al., 2015). Yet, modeling exercises relate the long-term effect of hurricane generated debris to a positive effect of decaying large woody debris on soil P exchange capacity (Sanford et al., 1991; Zimmerman et al., 1995).

Decaying wood may impact the physical, chemical, and biotic properties of the underlying soil (Zalamea et al., 2016), stabilize soil temperature (Spears et al., 2003), and contribute to the spatial heterogeneity of soil formation and resultant nutrient cycling in tropical forests (Zalamea et al., 2007, 2016). Further, tree species and decay stage are important factors defining the effect of decaying wood on the distribution of available nutrients (Zalamea et al., 2016). Lodge et al. (2016) found that surface soil on the upslope side of the logs can have significantly more nitrogen (N) and microbial biomass, likely from accumulation of leaf litter above the logs on steep slopes. To summarize, tropical cyclones deposit coarse woody debris on forest floors and significantly alter soil C and N dynamics, which consequently alter soil fertility, soil health, and forest productivity (Lodge et al., 2016).

Earthworms as Bioindicators

The occurrence or abundance of soil fauna can be considered a soil health bioindicator as it can reflect some habitat characteristics. These non-anthropogenic disturbances may increase nutrient losses from tropical forests, depending on how the debris is managed, how soil organisms respond to disturbance, and the chemical and physical characteristics of soil and litter (González and Barberena-Arias, 2017). Earthworms are recognized as indicators of soil fertility and health because they play an active role in organic matter movement and decay, soil formation, and improvement of soil structure by channeling and bioturbation (Fragoso and Lavelle, 1992; Liu and Zou, 2002). Their relatively large size (ranging from 1 to 80 cm, or larger), slow displacement in soil, and ability to re-colonize sites make earthworm concentrations and diversity easy to measure and an attractive bioindicator of soil health (Paoletti, 1999).

Tropical land-use changes affect the abundance and community structure of earthworms. Converting tropical forests to pastures often results in the reduction of aboveground plant litter inputs, causing the disappearance of soil surface litter layer (Zou and González, 1997; Paoletti, 1999). In short-term field experiments,

manipulating plant litter inputs lead to a decrease in anecic worms (those that build permanent burrows in the mineral soil; González and Zou, 1999; Sánchez and Zou, 2004). Furthermore, deforestation and establishment of exotic grasses decreases the diversity of earthworm communities in tropical Oxisols and Ultisols (Zou and González, 1997; Sánchez et al., 2003). Native earthworm communities are often negatively affected by non-native tree species, but they can be preserved in plantations where native tree species are planted (Zou and González, 2001). Conventional practices of site preparation and harvesting favors nonnative soil dwelling earthworms which often have a deleterious effect on native litter-dwelling worms. Therefore, forest management practices can drastically alter earthworm populations and diversity, and yet, maintaining a healthy population of earthworms can further promote forest nutrition and soil health in tropical tree plantations (Zou and González, 2001).

Temperate Forests

Temperate forests, located at mid-latitudes north and south of the Equator, are comprised of both evergreen and deciduous tree species and influenced by strong seasonal temperature shifts and other climate differences. Tree species, climate, parent material, and topography all influence temperate forest soil formation (Binkley and Fisher, 2012), but overstory species often influence soil chemistry (e.g., pH), biology (litter decomposition rate and rooting depth), and soil available water (Adams et al., 2019).

A number of natural and anthropogenic threats make temperate forest soil health vulnerable to degradation. One of the greatest concerns is environmental change due to catastrophic fires, but since temperate forests are often found near population centers, soil health can also be threatened by N deposition, acid rain, and invasive earthworms, plants, insects, and diseases. Forest management affects soil C storage through harvesting and site preparation operations that significantly alter surface and subsurface physical, chemical, and biological properties. However, in temperate and other ecosystems, if the external stress is not too great and the frequency and severity of disturbance are low, many soil properties will return to pre-disturbance conditions if given enough time (Morris et al., 1997).

Similar to AF and tropical forest soils, an important indicator of temperate forest soil health is SOM. This was documented by a 1958 Calhoun Experimental Forest study and gradually became a way to restore forest cover to land previously damaged by agriculture throughout the southeastern U.S (Metz, 1958). The long-term dataset documented the effect trees had on building surface organic horizons and improving soil moisture retention (Richter and Markewitz, 2001). Further, increasing C inputs led to even higher rates of decomposition (Richter

et al., 1999), and also soil changed porosity and nutrient cycling, thus generally improving soil health.

The North American Long-Term Soil Productivity Study

One important program for forest soil health is the Long-Term Soil Productivity (LTSP) study (Mushinski et al., 2017; Powers et al., 2005). This coordinated network of over 100 sites (Fig. 6.3) was initiated to address concerns that SOM removal and compaction were causing declines in temperate forest soil health. In general, loss of branches and twigs from the site did not alter tree growth, but when the surface organic horizons were removed many site experienced declines in productivity. Further, the effects of harvesting, compaction, and SOM removal varied considerably from site-to-site.

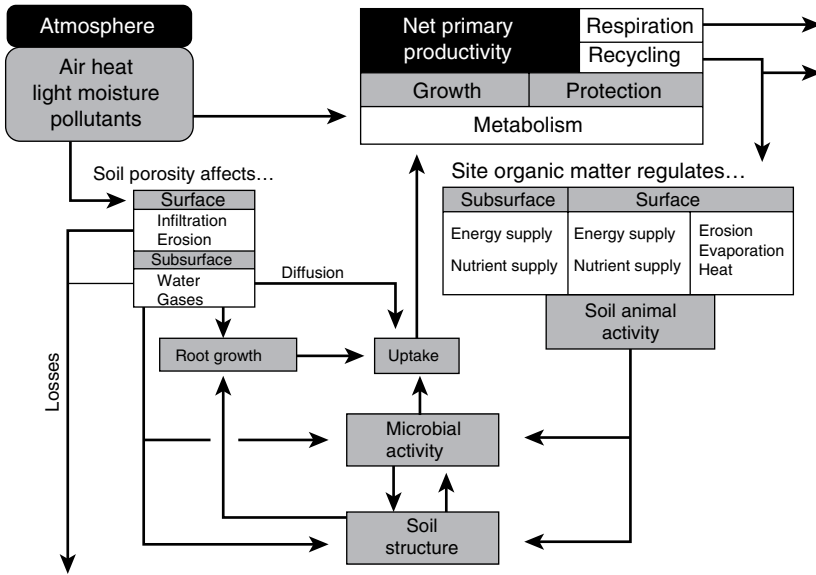


Figure 6.3 Geographic extent of LTSP sites in North America.

Temperate Forest Soil Health

Because temperate forests are widely distributed, a decline in soil health is likely to be of global importance. A rise in temperature of 1–2°C will have regional impacts on precipitation amounts and patterns leading to a changes in soil temperature and moisture properties (Adams et al., 2019). These changes, coupled

with land use change, air pollution, and biotic effects will control forest productivity, SOM decomposition, and the C balance within the soil and in the atmosphere (IPCC, 2003).

Elevated Carbon Dioxide

Rising carbon dioxide (CO_2) is considered to be a major driver of climate change and can significantly affect forest growth, SOM, and soil health. For example, the Free-Air Carbon Dioxide (CO_2) Experiments (FACE) study sites have shown that an increase in CO_2 can increase forest productivity, but there was no evidence to suggest that C storage increased in mineral soils beneath temperate forests (Norby et al., 2002). This is likely due to increased soil respiration (Phillips et al., 2012), root turnover (Bader et al., 2009), and microbial activity (Larson et al., 2002). It has also been shown that litter quality and species changes can change the quality of C inputs to the soil (MacKenzie et al., 2004). Changes in atmospheric and soil C associated with changing climate emphasizes the need to maintain soil bulk density, aeration, surface organic horizons, and other properties that promote soil aggregation and stable nutrient cycling.

Fire

In many temperate forests, wildfire is the most severe threat to soil health. With persistent and recurring drought, often coupled with high temperatures, wildfire risk and severity have been increasing and have resulted in greater loss of all or part of the surface organic horizons and mineral SOM. Those effects cascade into sediment loss, loss of C storage, and degradation of aggregate stability, but they may be partially off-set by creation of pyrogenic C. In fire-prone ecosystems, DeLuca and Aplet (2008) estimate that pyrogenic C inputs may account for 15 to 20% of the total C in temperate, coniferous forest mineral soils, but subsequent harvesting or thinning activities may reduce this amount. A recent meta-analysis noted an overall increase in C in frequently burned forests, but this varies by ecosystem type and burn severity (Pellegrini et al., 2017).

Thinning or Bioenergy Harvests

Many temperate forest stands need restoration because of lack of harvesting, fire suppression, and insect or disease outbreaks have resulted in excess woody biomass within many stands. There is also recent interest in using forests for bioenergy feedstock which may increase harvest operations on many sites. Little is known regarding the impact on soil health of repeated harvest due to forest thinning operations or feedstock extraction, but loss of SOM from periodic stand disturbances can be either negligible (Sanchez et al., 2006b) or significant, depending on soil type, tree species, ecosystem, or climatic regime (Grigal and Vance, 2000).

Conversely, excess biomass left during thinning or bioenergy harvest may provide fuel for uncharacteristically severe wildfires (Page-Dumroese et al., 2010).

Temperate forests supply important ecosystem services and therefore, it is critical to maintain a healthy, productive soil. Many nations have strong forest inventory and monitoring programs that also incorporates soil data collection. These inventories provide an opportunity to be pro-active in response to stressors that may alter forest or soil health.

Using National Forest Inventory and Analysis Data to Assess Forest Soil Health

In 1928, the McSweeney- McNary Forest Research Act (P.L. 70-466) directed the U.S. Department of Agriculture Forest Service to make “. . . a comprehensive survey of the present and prospective requirements for timber and other forest products of the United States. . .” The first inventories were completed in the 1930s and focused on the economic value of the forest by documenting the extent and status of timber resources (Cowlin and Moravets, 1938; Cunningham and Moser, 1938; Spillers, 1939). Seventy years later the Agriculture Research, Extension, and Education Reform Act of 1998 (16 USC 1642(e)) mandated that the Forest Service Forest Inventory and Analysis (FIA) program “make available to the public a report, prepared in cooperation with State foresters, that . . . contains an analysis of forest health conditions and trends.” This Act resulted in the development of comprehensive sampling protocols designed to monitor forest soils (chemical and physical properties), down and dead wood, lichens, ozone damage, tree crown condition, and vegetation diversity (O’Neill et al., 2005a; O’Neill et al., 2005b; Woodall et al., 2011).

Soil sampling conducted by the FIA program differs from the USDA National Cooperative Soil Survey (NCSS) in several critical ways. The FIA program is based on providing a spatially balanced, statistical sample of the landscape (Reams et al., 2005). In contrast, NCSS identifies relatively homogenous map units for the purpose of sampling (Soil Science Division Staff, 2017). Although digital soil mapping provided by the NCSS facilitates the estimation of error or uncertainty associated with soil properties (Kienast-Brown et al., 2017), the design-based framework used by FIA allows calculation of statistically robust estimates of various attributes along with associated estimates of uncertainty (Scott et al., 2005). Additionally, because of the explicit focus on the forest resource, FIA has a much greater sampling intensity across the forested landscape, under both public and private ownerships (Table 6.2). In contrast to the NCSS use of soil scientists who describe genetic horizons and sample the soil using soil pits (Schoeneberger et al., 2012b), FIA field crews collect ocular estimates of soil properties (erosion and rutting) and sample soils adjacent to the field plot by depth (0–10 and

Table 6.2 Forest soil sampling intensity of FIA plots by forest-type group.

Forest-type group	Number of plots
Alder/maple	27
Aspen/birch	447
California mixed conifer	65
Douglas-fir	387
Elm/ash/cottonwood	365
Exotic hardwoods	16
Exotic softwoods	13
Fir/spruce/mountain hemlock	351
Hemlock/Sitka spruce	69
Loblolly/shortleaf pine	356
Lodgepole pine	164
Longleaf/slash pine	75
Maple/beech/birch	718
Nonstocked	229
Oak/gum/cypress	142
Oak/hickory	1573
Oak/pine	253
Other eastern softwoods	33
Other hardwoods	46
Other softwoods	1
Other western softwoods	76
Pinyon/juniper	814
Ponderosa pine	253
Redwood	2
Spruce/fir	307
Tanoak/laurel	18
Tropical hardwoods	135
Western larch	15
Western oak	85
Western white pine	2
White/red/jack pine	163
Woodland hardwoods	328
Grand Total	7528

10–20 cm) by using a slide hammer and volumetric soil core sampler whenever possible (USDA Forest Service, 2011). Both organizations submit their samples to laboratories for physical and chemical analyses. FIA samples soil in association with a comprehensive sample of the aboveground forest resource (USDA Forest Service, 2017) to facilitate our understanding of linkages between soil and forest health (O'Neill et al., 2005b).

Observed soil properties are extrapolated by NCSS using map units. FIA does not define homogenous units for the purposes of sampling or extrapolation. Instead, it relies on two statistical strategies for estimation. The first method uses the base sample and the underlying sample in a design-based framework to convert point observations to estimates (Scott et al., 2005). The second method implements statistical imputation techniques to convert point observations to continuous surfaces (Wilson et al., 2012; Wilson et al., 2013; Domke et al., 2016; Domke et al., 2017).

Soils data collected by the FIA program have been used in a number of different assessments, either in isolation or in combination with other attributes, ranging from regional to national scales.

For example, the Forest Service is responsible for producing the official forest C estimates submitted to the UN Framework Convention on Climate Change (US Environmental Protection Agency, 2018). While soil C stocks have been reported since the early 1990s, they were initially estimated without the benefit of field observations on the FIA plot network. Estimates were based on linkages between FIA plots and NCSS map units (Smith and Heath, 2002, Amichev and Galbraith, 2004). With the addition of the soil indicator to the FIA program in 1999, the foundation was laid for reporting forest C stocks by using continuous, integrated field monitoring (Perry et al., 2009). Forest floor and mineral soil C stocks are currently estimated using an imputation approach (Domke et al., 2016, Domke et al., 2017). In a testament to the value of the inventory, Domke et al. (2016) demonstrated that current Good Practice Guidance for Tier 1 approaches (estimates based on simple methods and default values) overestimate forest floor C stocks. These empirical data have also demonstrated the importance of reforestation for C sequestration (Nave et al., 2018).

In addition to the FIA program, the Forest Service has a Forest Health Monitoring program that plays a role nurturing thoughtful investigations of forest health, including soil data. Their annual National Technical Reports serve as venues to explore nascent trends detected across the monitoring network. Early reports summarized evaluations of soil C and other physical and chemical properties (Perry and Amacher, 2007a; Perry and Amacher, 2007b; Perry and Amacher, 2007c). Building on these assessments of individual soil properties,

Amacher et al. (2007) developed a technique to integrate the multiple chemical and physical observations from FIA plots into an index of forest soil health. Furthermore, FIA data has been used to map the legacy of atmospheric deposition observed in Ca:Al ratios (Perry and Amacher, 2012) and increased mortality of sugar maple (*Acer saccharum*; Perry and Zimmerman, 2012). These myriad analyses illustrate how FIA has become a foundation for national forest resource assessment (Perry and Amacher, 2009).

While there are tremendous strengths in the Forest Service's monitoring of forest soils, it is important to acknowledge known limitations. First, the soil program is considered 'Core Optional', and is not implemented across the nation on a regular basis (Fig. 6.4). This limits the program's ability to map soil health properties of interest (e.g., SOC, N) and document change. Inference from this inventory program is limited by the sampling protocol. Fixed depth sampling is a reproducible method of data collection, but it may yield samples straddling soil horizons and mixing soils of divergent properties (Schoeneberger et al., 2012b); this complicates interpretation of the resulting data and estimates. Finally, sampling frequency could be optimized to detect changes of interest to the Forest Service and partners concerned about managing forest health. The intensity of FIA's field campaigns currently yield a complete sample of the forest resource every 5 to 7 yr in the eastern U. S. and every 10 yr in the western U.S. Because the annual FIA program was implemented in stages beginning in the late 1990s, sampling is not necessarily completed uniformly across years. Sampling is also paused between inventory cycles to increase the likelihood of capturing changes in soil properties.

Despite these limitations, soil sampling conducted by the FIA program represents a tremendously valuable, statistically sound sample of forest soil health. How might it be improved? The most common concerns fit broadly under sampling intensity. First, the mineral soil is sampled to 20 cm by a bulk density sampler where possible. However, IPCC Good Practice Guidance suggests monitoring of soil carbon to at least 30 cm (IPCC, 2003) in agricultural soils. However, deeper sampling (at least to 80 cm) in forested soils should be conducted to better assess deep soil C pools and changes over time (Harrison et al., 2011). Second, only one sample of mineral soil is collected on each plot. This efficient use of limited funds provides landscape-level information, but it provides no detail on small-scale variation in soil health. Third, soil samples are collected only on a subset of the full plot network; originally soil sampling represented a 1/16th subset, but the program is exploring the value of sampling at greater intensities. Another major concern is the narrow focus on physical and chemical properties when microbes are now understood to have a critical role in forest diversity and productivity (van der Heijden et al., 2008). Sampling

Soil Visits in FIADB By Forest-type Group 2018 SEP 25

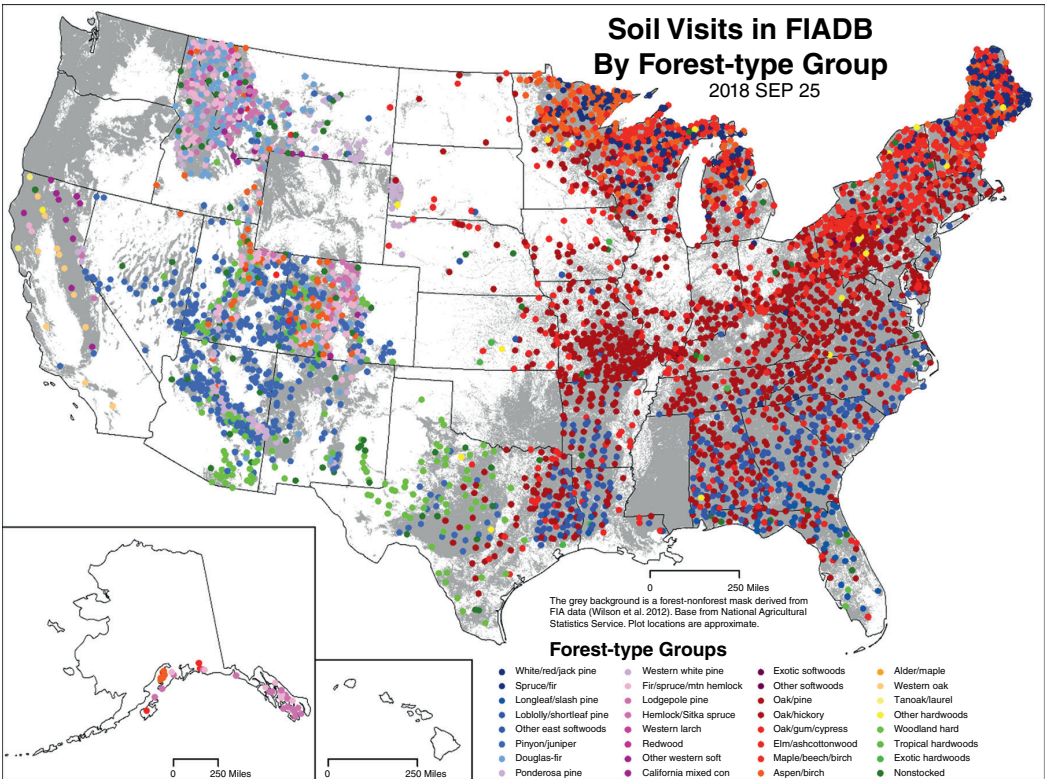


Figure 6.4 Forest Inventory and Analysis samples collected by Forest-type Group.

soil metagenomics to understand fungal diversity may be a relatively inexpensive way to understand this impactful biological property (Tringe et al., 2005, Fierer et al., 2012) and has been piloted on FIA plots in northern Idaho (Ross-Davis et al., 2016).

Forest Soil Health Data Limitations and Management Implications

Often, forest health measurements look at only aboveground responses because they are easier to measure than belowground responses. Numerous studies indicate that forest management and inherent soil factors will elicit differing tree responses (e.g., Greacen and Sands, 1980; Senyk and Craigdallie, 1997; Heninger et al., 2001; Slesak et al., 2017). Ideally, above- and belowground data are needed at a site before harvest operations are conducted so that the magnitude of change and the functions and processes affected can be quantified (Grigal and Vance, 2000). Further, many studies constrain their sampling efforts to the surface mineral soil and to the fine fraction (< 2mm) and omit coarse wood, large rocks, or roots from sampling because of financial or time limitations. Recent studies have pointed out that coarse-fragments (Jurgensen et al., 2017) and deep soil nutrient pools and OM should also be considered to evaluate long-term impacts on soil health (Harrison et al., 2011). Not accounting for these factors could result in faulty soil health assessments (Slesak et al., 2017). In many cases baseline information might not be available. In this case, local specialists, use of the Natural Resource Conservation Service databases, or use of information from similar sites elsewhere may be necessary to make inferences on soil productivity changes. In general, low fertility, coarse-textured soils are at greater risk of nutrient limitations from land management than higher fertility soils with finer textures (Garrison et al., 2000).

Long-term studies are key to being able to link belowground ecological changes associated with land management. Metagenomics, standard decomposition substrates, soil fauna, and microbial biomass are all techniques that help link soil physical, chemical, and biological responses. Data gathered immediately after harvesting are a valuable tool, and there are many examples of developing risk rating systems for forest sites (e.g., Reeves et al., 2012). In fact, best management practices (BMPs) have been developed by some states that minimize or avoid soil impacts considered detrimental to forest productivity. Risk rating tools can provide a framework that, with local calibration, can be used across a wide variety of forested landscapes to depict soils that may be at risk of damage during ground-based harvest activities (Reeves et al., 2012). Use of a standardized soil monitoring protocol is also useful for assessing short- and long-term soil health based on several soil quality indices (Heninger et al.,

2001; Page-Dumroese et al., 2010). This data will be useful for meta-analyses that examine soil and forest health changes.

Climate Change, Fire Shifts, Invasive Species

Three points are clear: soil C is a pervasive material within all forested soils, it is crucial for providing ecosystem services (e.g., soil water quality and quantity), and it is an essential indicator of soil health. Current US federal policy is to harvest forests in a manner that protects soil, watershed, fish, wildlife, recreations, and esthetic resources. Consequently, soil health must also be protected to ensure all other values are maintained. Since soil C is critical, we must begin to assess its vulnerability to climate, fire, and invasive species shifts and to understand these changes more widely.

Fire can affect soil C by changing the quantity and quality of C inputs by mineralizing surface OM and altering mineral soil C (Neary et al., 1999). However, those changes may be offset by creation of black C during wildfires, prescribed burns, or through the addition of biochar to forest soils (Page-Dumroese et al., 2018). Since many public lands have a short window for burning unmerchantable woody material, alternative markets such as bioenergy or bio-based products are one way to reduce the amount of residual woody material while simultaneously conserving C.

Linked to changes in soil C and N is an increase in invasive species. The initial increases in invasive species is caused by a chronic disruption in N, SOM, or nutrient cycling (Hobbs and Huenneke, 1992). Working to adjust these soil imbalances may be one method for restoring microbial-fauna-soil-plant relationships and foster increased soil health.

Conclusion: Criteria and Indicators for Monitoring Forest Soil Health

Monitoring forest soil health is a process to estimate changes in soil conditions that have occurred since the last time it was measured. However, this approach gives no indication of future soil conditions that may result from continuing impacts of degrading processes (e.g., climate change, pollution; Wagenet and Hutson, 1997). Our roles as forest soil scientists should be to anticipate effects in a prospective manner rather than retrospectively (Wagenet and Hutson, 1997; Adams et al., 2000). We must stretch our knowledge of soil data to encompass dynamic processes that underpin soil health assessments. Data from each of the

previous examples can be used to give a qualitative perspective on the impact of management scenarios on soil health and provides resources that further our understanding. Information and data from both short- and long-term studies can be placed into decision trees that help integrate soil property changes into site-specific land management decisions (Wagenet and Hutson, 1997). Current measurements can also be combined with archived samples from numerous sources to help provide additional historical context about how soils are impacted over time by a changing climate and/or land management activities.

Researchers take static measurements of forest soil properties (cation exchange capacity, C, base cations, etc.), but it is imperative to also determine the cause and effect relationships between management and soil properties. These static measurements can also be used to develop risk rating systems. Risk rating systems used to develop BMPs are one way to use data to describe acceptable management retrospectively. Once these relationships are understood we can identify indicators of soil change that could lead to a decline in soil health and forest growth. From the empirical trials we can then move toward forecasting acceptable management across a wide-range of soil types and required ecosystem services.

Use of a standardized soil monitoring protocol is also essential for assessing soil health based on several soil characteristics (compaction, rutting, displacement, erosion, ground cover, burn severity; Heninger et al., 2001; Page-Dumroese et al., 2009). This data can then be used with meta-analyses that examine soil and forest health changes.

Summary

- Sustainable management of temperate and tropical forests as well as AF sites depends on healthy soils and the ability to identify soil change indicators that reflect soil health declines.
- There are no widely-applicable standardized measurements or methods for assessing forest soil health.
- Soil texture influences how compaction or SOM loss can alter soil health.
- Faunal and microbial inventories, and the development of specialized taxonomic expertise, is needed to better describe organisms and biological property changes and links with aboveground changes.
- FIA forest and soil data can be used as an index of ecosystem health.
- Soil monitoring of management practices will help elucidate if we are meeting criteria for sustainability.
- Using long-term trials and archived samples we can begin to forecast ecosystem processes changes that may require a change in management.

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