8 Soil Management and Restoration

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Introduction

> The destruction of soil is the most fundamental kind of economic loss which the human race can suffer.—The Essential Aldo Leopold: Quotations and Commentaries

Soils sequester carbon (C), store and regulate water, cycle nutrients, regulate temperatures, decompose and filter waste, and support life (Dominati et al. 2010). We depend, and will continue to depend, on these ecosystem services provided by soils, services that are products of interactions between and among abiotic and biotic properties and that are the foundation for self-maintenance in an ecosystem (SER 2004).

But, soil is a limited resource. It takes thousands of years to develop soil, yet it can lose its productive capacity and ecological integrity in a fraction of that time as the result of human activities or natural events (Heneghan et al. 2008; Hillel 2004). The impacts of management actions and natural events can remain on the landscape for decades and longer, leaving land use and historical legacies (Foster et al. 2003; Morris et al. 2014) that can cause profound ecological and economic consequences from lost farm, pasture, or forest productivity. Furthermore, climate shifts and environmental stressors affect soil properties and functions, both directly and indirectly. Rises in temperature affect decomposition and nutrient cycling, biological populations, and soil hydrologic functions. Flooding is a natural disturbance in riparian and floodplain ecosystems, but flood sizes and frequencies have been altered by human influences through damming and channelizing rivers, draining wetlands, and deforesting floodplains, so that most flooding now often exceeds the natural range of variation.

As natural resources become limited, the value of managing and restoring aboveground and belowground processes becomes more important. Sustainable soil management involves the concepts of using, improving, and restoring the productive capacity and processes of soil (Lal and Stewart 1992), and we can use ecological restoration, which is intimately linked with soil management, to ameliorate degraded and disturbed resources, reverse the trends of soil degradation, and enhance soil properties to regain ecosystem health. Ecological restoration is one of several actions that can ameliorate degraded and disturbed soils, defined as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER 2004). The practice of ecological restoration draws from and integrates many disciplines from agronomy to wildlife management and from engineering to indigenous knowledge.

Concerns about ecosystem services (e.g., food, water, energy, biodiversity conservation) within the context of a changing climate lead to calls for action, research projects, and eventually the development of new management and restoration techniques (Adhikari and Hartemink 2016; McBratney et al. 2014). This chapter begins with a summary of historical forest and rangeland management with respect to soils and is followed by an overview of the shifts in policy and planning and advances in management and restoration. We highlight a few case studies, discuss monitoring, and end with key findings and information needs.

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Context

> Humans are now an order of magnitude more important at moving sediment than the sum of all other natural processes operating on the surface of the planet.—Wilkinson (2005)

Historical Forest Soil Management

The early history of forests in North America does not include a record of soil impacts, but we can infer some aspects of this from land use and population factors. As Europeans began arriving in North America during the mid-seventeenth century, total forest area was estimated at 4.14 million km^2 (1023 million acres) (Oswalt et al. 2014). Native peoples were soon decimated by disease, and much of their agricultural land reverted to forest naturally (Lewis and Maslin 2015; Mann et al. 1988). The extent of forest cover in what is now the United States, as reconstructed from saw timber inventories, was likely greater from 1650 to 1700 than in any other period (Birdsey et al. 2006). Williams (1989) suggested that forests may have covered "at least four-fifths of the land area east of the Mississippi River." Population increased slowly during the 1700s but jumped from 5.3 million to 76 million people in the 1800s (Fedkiw 1989; MacCleery 2004). To support the increase in population growth, settlers cleared approximately 0.77 million forested km^2 for farms and pastures between 1850 and 1910, more area than had been cleared in the previous 250 years (Williams 1989). Following the period of intensive land clearing, the remaining 3.05 million km^2 of forest (Oswalt et al. 2014) was largely saved by technological advances; land used to feed draft animals became available for other crops as animal labor was replaced by motorized equipment, and agricultural production per area boomed as a result of plant breeding, irrigation, and fertilizer. In 2012, forested area has increased only slightly to 3.1 million km^2 (766 million acres) since 1910 (Oswalt et al. 2014).

Until recently, forest soil management has been characterized primarily by inattention or risk avoidance. Inattention prevailed throughout the 1800s in the absence of regulations, conservation practices, or foresters (Fedkiw 1989; MacCleery 1993); this period of rapid land clearance was also a time of high demand for wood. Land clearing and extraction from remaining forests resulted in an alarming 75% drawdown of sawtimber stocks between 1800 and 1920 (Birdsey et al. 2006), leading to the first forest conservation policies in the 1930s (MacCleery 1993). Wood was enormously important during the nineteenth century as a fuel source for heat and steam power; it was the only material for building fences up until the mid-1800s (MacCleery 1993); and it also served a significant industrial demand for charcoal (Foster and Aber 2004). After 1850, lumber production increased quickly as cities were constructed and farmsteads were built in the Great Plains; railroad expansion also used a large proportion of harvested wood (MacCleery 1993).

Timber extraction prior to the Civil War typically focused on removing high-value trees close to waterways, as logs were heavy and difficult to move (Fedkiw 1989). Logging was mainly accomplished by hand-felling trees and skidding the logs with oxen or horse teams. It was a cumbersome process that left most of the forest unaffected, although damage to streambanks and streams was serious, and effects still exist today (Sedell et al. 1991). When railroads came into widespread use in the latter half of the 1800s, destructive logging practices affected larger areas. The most widespread soil damage of that period was caused by fires ignited by sparks from locomotives. Wildfires ripped through logging slash, destroying the organic horizons of soils and leading to postfire erosion, sedimentation, and nutrient loss (Fedkiw 1989; MacCleery 2004). In the western United States, general policies to pile and burn logging slash were adopted to prevent such wildfires (Lyman 1947). However, slash pile burning comes with its own set of short- and long-term consequences to soil properties that long-term research would later identify (reviewed in Rhodes and Fornwalt 2015). In some regions, farming was attempted on unsuitable logged-over soils that eventually reverted to forest.

Soil fertility, as a component of forest site quality, was gradually recognized during the 1920s and 1930s, and research efforts during the following decades focused on matching forest species to site. The use of site index as a rough measure of productive capacity and the construction of yield tables led to the acceptance of the two soil facts: Soils differ widely in their ability to support tree growth and vegetative growth overall; and forest management as well as other types of resource management, such as wildlife management, could be informed and made more productive by a knowledge of soil (Leopold 1933; Wilde 1958).

Forestry operations moved toward mechanization after World War II (WWII) and progressed in stages as technology changed. Simmons wrote in 1949 that the "tractor, the power saw, and the motortruck are becoming commonplace throughout the country, even on small logging jobs." Truck hauling replaced railroads for moving logs to mills, and ground skidders replaced horse teams for dragging logs to pickup points. Road and skid trail layout became important, utilizing engineering techniques to stabilize roadbeds and manage hydrology. This was primarily to maintain longevity of the roads but also served to limit soil movement and loss. With the advent of heavier tractors and specialized logging equipment, soil compaction became a widespread problem. In the 1960s, researchers began to study soil compaction as a factor responsible for decreased forest growth. Insights as to how soil and ecosystem properties may have been changed by historical logging, fire, and in some locations subsequent tillage or grazing are gradually becoming available. The recognition that soil porosity was a nationwide forestry concern led to including compaction treatments in the North American Long-Term Soil Productivity (LTSP) study initiated in 1989 (Powers et al. 2005), a research effort which has gathered long-term data on the effects and interactions of compaction and organic matter removal relative to forest growth.

Another forest management issue in the post-WWII era was the realization that removing trees from the forest was akin to harvesting crops and that forest soils, as well as agricultural soils, might be susceptible to nutrient loss. As new laboratory techniques developed, researchers were able to quantify amounts of nutrients in various parts of trees, and tree species, and calculate nutrient budgets that estimated removals in timber harvesting (e.g., Perala and Alban 1982). This research led to guidance on limiting certain types of harvests and on the use of forest fertilization in economically viable situations. One of the first reported forest fertilization successes was at the Charles Lathrop Pack Demonstration Forest in Washington, where researchers documented a positive response using potassium fertilization on red pine (*Pinus resinosa*) (Heiberg et al. 1964). Fertility concerns, along with improved laboratory equipment and techniques and the advent of soil surveys in forested areas, led to further examination of forest soils and an increased recognition of the importance of organic material in the forest litter layer and upper mineral soil layers. Nutrient changes have been studied in many locations, notably at Hubbard Brook Experimental Forest in New Hampshire, and studies elucidating soil microbial functions have recently provided insights on historical impacts and considerations for future soil management (Jangid et al. 2011).

Historical Rangeland Soil Management

In the United States, rangelands occupy approximately 35% of the land area (Reeves and Mitchell 2011); major rangelands include the Great Plains, the Desert Southwest, the Great Basin, and the Intermountain Plains and Valleys. Rangeland is "land on which the indigenous vegetation (climax or natural potential) is predominately grasses, grass-like plants, forbs, or shrubs and is managed as a natural ecosystem. If plants are introduced, they are managed similarly. Rangeland includes natural grasslands, savannas, shrublands, many deserts, tundras, alpine communities, marshes and meadows" (SRM 1998). The majority of rangelands are categorized as drylands, which are lands limited by soil water (Hassan et al. 2005) with soils having, in most cases, low organic matter, low fertility, high accumulations of calcium carbonate, and low nutrient resources, such as available nitrogen (N) and phosphorus (P) (Sharma et al. 1992; Stott and Martin 1989). Despite these limitations, rangelands are expansive and heterogeneous, supporting a diversity of ecosystems that provide ecological, social, and economical services.

Livestock grazing was, and remains to this day, a primary land use of rangelands during the nineteenth and twentieth centuries. Several rangeland research stations, including the Jornada Experimental Range in New Mexico (circa 1912), were developed in the early twentieth century to address and keep pace with the unprecedented and intensive livestock grazing practices in the western United States. Early management was built on assumptions of equilibrium ecology and steady-state management (i.e., the range condition model) (Clements 1916; Dyksterhuis 1949; Sampson 1923), which presumed that livestock grazing controlled plant succession, such that the species composition of plant communities was a linear response to grazing intensity (Briske et al. 2005). For 50 years, the range condition model was the standard protocol and worked moderately well in grasslands dominated by perennial herbaceous forbs and rhizomatous grasses. At the turn of the twentieth century, however, rangeland conditions were considered poor (Gardner 1991), mostly due to improper livestock grazing practices and uninformed management. The inability of the range condition model to account for complex vegetation dynamics such as woody plant encroachment and establishment and spread of nonnative plant species (Westoby et al. 1989), coupled with advances in resilience and state and transition concepts and theories (Briske et al. 2003; Friedel 1991; Holling 1973; Westoby et al. 1989), led to comprehensive reviews of the rangeland profession and management of rangelands. Reviews by the Natural Research Council (1994) and the Society for Range Management (1995) called for and outlined the standardization of monitoring and replacement of the range condition model with a model that could account for multiple states within and across plant communities (i.e., state and transition models) (Briske et al. 2005; Westoby et al. 1989).

Progressive Shifts in Policy and Planning

> The starting point must be the soil, or at least the substrate into which plants must establish and root, for although soil can exist without plants, there are few plants that can exist without soil.—Bradshaw (1987)

As we look back on the history of forest and range management, it is apparent that a number of changes in soil management and protection have arisen from new information made possible by advances in research and technology. In recognition of the relationship between aboveground and belowground processes, and with a better understanding of management impacts as well as natural disturbance and recovery processes, several approaches and even shifts in policy have been discussed and developed. The soil ecological knowledge (SEK) approach, for example, acknowledges interactions among principal components of the soil systems as well as feedbacks between aboveground and belowground ecosystem processes (Heneghan et al. 2008).

Currently, forest and rangeland soil management includes approaches that protect soils while seeking to avoid and minimize soil damage. This is still a viable and efficient management approach, as rehabilitation and restoration processes are often costly and impractical, constrained by concerns for tree damage and impeded by accessibility, substrate, and terrain. Statutes, regulations, and guidelines at the federal and state levels have been developed to address many aspects of soil resource protection and management. A more proactive approach to the management and rehabilitation of forest soils has gradually been forming, assembling a variety of approaches that use soil properties to guide land management actions as well as taking direct action toward restoring desirable soil properties.

Forest Service Policy

Limiting the loss of soil productivity has been the focus of soil management on National Forest System (NFS) lands since the 1980s. Under the National Forest Management Act of 1976 (NFMA), all national forests are required to assess the impacts of management actions to ensure that they "will not produce substantial and permanent impairment of the productivity of the land." The NFMA did not define "land productivity," but the USDA Forest Service (hereafter, Forest Service), with guidance from the US Office of General Council, defined it as the capacity of a soil to produce vegetative growth. While land productivity is generally perceived as being broader—including timber, wildlife, watershed, fisheries, and recreation values—soil productivity is essential to the sustained production of all other ecosystem goods and services (Powers et al. 2005). When considering how to monitor land productivity, NFS soil scientists noted the difficulty in detecting a change in productive potential and decided that a change of approximately 15% would be a detectable threshold. Each NFS region developed soil quality standards to detect changes, and some regions developed management guidance that limited soil disturbance to no more than 15%, or 20% on an area basis, reasoning that this would protect land productivity.

Forest Service Manual (FSM) Chapter 2550 Soil Management directs soil resource management on NFS lands (USDA FS 2010). The manual was revised in 2010 to provide a greater focus on ecological functions, with an objective of maintaining or improving soil health on NFS lands "to sustain ecological processes and function so that desired ecosystem services are provided in perpetuity." The FSM defines soil quality as "the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation and ecosystem health" (USDA FS 2010). Soil function is any ecological service, role, or task that soil performs. The FSM identifies six soil functions: soil biology (see also Chapter 5), soil hydrology (see also Chapter 3), nutrient cycling (see also Chapter 4), C storage (see also Chapter 1), soil stability and support, and filtering and buffering. In order to provide multiple uses and ecosystem services in perpetuity, these six soil functions need to be active and effectively working.

With the shift of National policy in the 2010s, several NFS regions and forests have adapted soil quality guidance that reflects a greater focus on managing NFS lands to maintain soil ecological functions as a foundation of planning management actions instead of focusing on soil disturbance as a proxy for maintaining productivity. For example, new regional guidance was created in 2012 for the Eastern Region (Region 9). This guidance directs soil scientists to look at the landscape and determine the site-specific soil properties that are at risk for an ecosystem response from management actions (USDA FS 2012). When high risk is found, actions on those soils are mitigated to protect soil quality and ecosystem function. Monitoring of the soil and ecosystem response on these sites is then fed back into the planning process to better inform the next round. The Colville National Forest also took a similar approach in its recent Forest Land and Resource Management Plan (USDA FS 2018). The forest linked important soil properties to the six soil functions identified in the FSM with the goal of maintaining soil function on the landscape.

Use of Ecological Sites and Associated Information

A more proactive approach to the management and rehabilitation of forest and rangeland soils has gradually been forming, assembling a variety of techniques that use soils and other ecosystem properties to guide management actions, as well as taking direct action toward restoring desirable soil properties. The need for an ecological approach has been recognized for some time. Rowe (1996) stated that, "recognition of land/water ecosystems in a hierarchy of sizes can provide a rational base for the many-scaled problems of protection and careful exploitation in the fields of forestry, agriculture, wildlife and recreation."

The ecological site concept was formed from an integration of previous land classification systems implemented in parts of Canada and Europe during the 1900s (Barnes et al. 1982), along with those previously utilized in the United States, including the Land Systems Inventory (Wertz and Arnold 1972), habitat type classifications pioneered in Finland (Cajander 1926) and widely applied in the

Northwestern United States (Daubenmire 1968), the US Soil Survey, and the Forest Service's ecological classification system (Bailey 1987; Cleland et al. 1997).

An ecological site is "a conceptual division of the landscape that is defined as a distinctive kind of land based on recurring soil, landform, geological, and climate characteristics that differs from other kinds of land in its ability to produce distinctive kinds and amounts of vegetation and in its ability to respond similarly to management actions and natural disturbances" (Caudle et al. 2013). Ecological site descriptions (ESDs) are linked to soil survey polygons and associated soil and site information, such as vegetation type and structure, disturbance processes, successional states, and rates of change, and these attributes inform management decisions for a variety of uses (Herrick et al. 2006b). Within this standardized methodology, ecological sites are the basic land classification units for documenting soil, site, and biological characteristics for current and potential future conditions (USDA 2003). Ecological site descriptions are being developed for rangelands and forests across the United States and provide a standardized communication and planning tool for land managers to assess site function and develop projects.

State and transition models (STMs) are interpretations linked to ecological sites, modeling the plant communities that typically develop in response to ecosystem drivers (Bestelmeyer et al. 2003). The STM concept is an outgrowth of concepts of ecosystem succession developed over the last century and summarized by Christensen (2014), who stated that, "We now understand that there is no single unique or unifying mechanism for successional change, that successional trajectories are highly varied and rarely deterministic, and that succession has no specific endpoint." State and transition models reflect some of the complexity involved in succession by recognizing typical disturbances and feedback loops, continuous and noncontinuous transitions between states, and alternate stable states for each ecological site. As such, they are a valuable tool in providing a framework for the probable outcomes of management decisions. State and transition models and ecological sites are linked to soil map units in a one-to-many relationship, i.e., an ecological site may occur on multiple soil map units, thus providing spatial information needed for land management planning. However, limitations of polygon-based soil maps (Zhu 2000) must be considered when applying ecological site information.

Agencies including the USDA Natural Resources Conservation Service (USDA NRCS), the Department of Interior's Bureau of Land Management (BLM), and Forest Service have jointly adopted ecological sites and STMs to aid in rangeland management (Bestelmeyer et al. 2017; Pellant et al. 2005). The agencies have developed a systematic rangeland assessment, detailed in the Interagency Ecological Site Handbook for Rangelands (Caudle et al. 2013), that incorporates STM concepts of ecosystem function specific to ecological sites.

Advances in Management and Restoration

When exposed to various management practices, soils may lose, retain, or improve their capacity to sustain plant, animal, microbial productivity, health, and vitality while also maintaining balanced hydrologic, C, nutrient cycles and ecosystem functions (Heneghan et al. 2008). Departure from natural ranges of variability due to, for example, loss of organic matter—a key driver of ecosystem function—warrants restoration. Traditional approaches focus on manipulating a single chemical, physical, or biological factor to improve soil function, such as by establishing plant cover on highly degraded sites to prevent erosion. Improved technology since the 1980s has led to rapid and significant advances in understanding soil processes and interactions with other ecosystem components, such that restoration of structure and function requires the integration of multiple factors (physical, biological, and chemical). For example, an understanding of belowground biology and processes precipitated an emphasis on soil health and the capacity of a soil to function as a vital, living ecosystem that sustains water, air, plants, animals, and humans (Doran and Zeiss 2000). Soil health is a term used to emphasize and convey that soil functions are "mediated by a diversity of living organisms that require management and conservation" (Doran and Zeiss 2000). Soil health is similar to the older term "soil quality" but places a greater emphasis on the biological components of soils and their roles in ecological processes, especially the cycling of organic matter and nutrients.

Increased knowledge of the legacies of past land use also led to recognition of degraded steady states and appropriate adjustments of land management planning objectives. The recognition of soil-mediated legacy effects can be useful for developing realistic and practical restoration goals (see Foster et al. 2003; Morris 2011; Morris et al. 2011, 2013, 2014). Another current focus is the recognition of soils as part of an ecosystem, taking on properties from air, water, plants, and minerals and contributing influences to the system. This approach is evident in planning and modeling applications and in developing indicators for sustainability (Jonsson et al. 2016). Although there is much left to discover, the new knowledge contributes in many ways to our ability to manage and restore soil functions. In the following sections, we discuss soils-based management approaches; changes in forest, fire, and mine reclamation practices; invasive species and soils; and innovative techniques in biochar, seed coatings, and soil transplants.

Soils-based Management

Properties observed in soils are a repository of information, referred to as a "soil memory" or "pedomemory" (see references in Nauman et al. 2015a). Biotic and abiotic interactions, such as those between plant communities and soil, over time can develop specific soil properties indicative of site history. Linking soil properties with historical reference plant communities is a foundation for soils-based management frameworks, such as ESDs. Ecological sites and STMs are being used to guide forest and rangeland management and restoration across the United States (Caudle et al. 2013). Mapping soil morphology and ecological sites to estimate historical plant composition has provided guidance for the restoration of historically disturbed red spruce-eastern hemlock (*Picea rubens-Tsuga canadensis*) forests in the central Appalachian Mountains (Nauman et al. 2015a, b). And in the western United States, mapping ecological sites has been useful in managing and restoring shrub-steppe habitats (Williams et al. 2011, 2016a). In the following sections, we briefly discuss other soils-based management approaches, such as resistance and resilience, soil security, and soil sensitivity.

Application of Resistance and Resilience Concepts

A framework for assessing the resistance and resilience of sagebrush (*Artemisia* spp.) and pinyon-juniper (*Pinus - Juniperus*) ecosystems to invasive species and fire threats has been developed for the western United States (Chambers et al. 2014, 2017; Miller et al. 2014). Resistance is "the ability of an area to recover from disturbance, such as wildfire or drought," and resilience is the "ability of an area of land to remain largely unchanged in the face of stress, disturbance, or invasive species" (USDOI 2015) (also see Box 5.1). The concepts are useful to land planning and assessment. Sagebrush communities on cool, moist, more productive sites are more resistant and resilient to drought and species invasions than those at the drier, warmer end of the spectrum (Chambers et al. 2014, 2017; Maestas et al. 2016). While soil differences related to drought resistance are well known in agricultural crops, the concept has not been commonly applied to natural vegetative communities. In the southern Rocky Mountains, particularly in the Four Corners and Upper Rio Grande ecoregions, sagebrush communities are likely to have higher resistance and resilience than those in the Great Basin (Chambers et al. 2014). A substantial portion of the southern Rockies is classified as having moderate to high resistance and resilience due to climate. The monsoonal precipitation pattern also promotes perennial forbs and grasses, leading to higher resistance to invasive species invasion (e.g., Bradford and Laurenroth 2006).

A similar approach is applied to managing pinyon-juniper woodland ecosystems (Miller et al. 2014). Woodlands are long-established, complex ecosystems with a canopy cover of 40% or less and a well-developed understory (Thomas and Packham 2007). Trees in woodlands are often distributed unevenly, with highly variable spatial patterning. Woodland structure is vital to the continued existence of many organisms such as birds and butterflies, and woodlands also provide ecosystem services such as water quality and quantity. Woodlands, like forests, are valued for their wood products, and tree removal could result in changes in soil nutrients, hydrology, and biodiversity.

Soil Security

The concept of soil security has emerged in the past decade as a way to communicate and elevate the urgency of maintaining and improving soil resources to support human needs, biological diversity, and ecosystem structure and function (McBratney et al. 2014). Soil security concerns are placed on par with water, food, and energy security, among other factors, that contribute to sustainability. Addressing soil security requires sustaining the capability or long-term productive capacity of soils, as well as maintaining or restoring their condition. A monetary or capital value placed on soils would ensure their consideration in land allocations based on economics. Stewardship, or connectivity of people to the land, is another aspect of achieving soil security, as is public policy and use regulation. As such, soil security includes considerations of soil health and quality but also adds human elements and synthesizes an overall approach within the context of global sustainability. An example of placing value on soils is the assessment of forest land by Oregon's Department of Revenue (ORS 321.805–855). Large tracts of privately owned forestland are classified into productivity groups using information found in the NRCS Soil Survey. Productive soils receive higher productivity classes and are assessed as more valuable for forest management.

Soil Sensitivity

Several adaptation assessments are using new information derived from soil vulnerability models. Peterman and Ferschweiler (2016) identified soil sensitivity factors that indicate increased vulnerability to drought-related erosion and vegetation loss and predicted changes in vegetation functional groups as a result of climate change. Changing temperature and precipitation regimes directly influence soils, plant productivity and phenology, and ecosystem resilience to invasive species and wildfire. Soil sensitivity is a measure of the ability of a soil to endure or recover from a disturbance. Peterman and Ferschweiler (2016) created vulnerability maps that rank soils according to the presence of vulnerability factors and the potential for change in vegetation type under future climate scenarios. Loss of plant cover and productivity due to drought and warm temperatures can lead to increased erosion and loss of important soil properties, such as water holding capacity, stability, organic matter, and nutrients (Breshears et al. 2003; Munson et al. 2011). Vulnerability maps can help managers identify priority areas for restoration and conservation. For example, Ringo and others (2018) developed a geospatial soil drought model for the Pacific Northwest using soil properties and climate information. These data are being used to examine forest resiliency, wildfire risk, and potential management actions to mitigate undesired ecosystem trajectories.

Forest Management

Forest management has changed in the modern era due to the development of new technologies as well as a better understanding of the ecological impacts of management actions. One major technological change is the mechanization of forest operations, with equipment replacing manpower in harvesting, site preparation, and planting (Silversides 1997). When applied properly, new technology can decrease forestry impacts on soils; for example, the use of low-pressure tires and tracked vehicles produces less compaction, and equipment like the “shovel logger” can reach out as far as 100 m from a road to fell and move trees. Other technology being used in the United States includes tethered logging systems, which can harvest on slopes up to 80% in a ground-based system and can provide access to sites with greater limitations.

Compaction is still the most serious issue today on many forest sites because it alters soil porosity, reduces infiltration, and can increase erosion, thereby leading to reduced movement of water, air, and nutrients through the soil; it also impacts microbial populations and can reduce tree growth (Brussard and van Faasen 1994; Bulmer and Simpson 2005; Page-Dumroese et al. 2009; Stone 2002; Thibodeau et al. 2000; von Wilpert and Schäffer 2006; Wang et al. 2005). Depending on soil texture, the depth of compaction, and site resiliency, soil compaction can also alter plant successional pathways and overall productivity. Soil compaction is greatest in roads, trails, and landings but can occur in general harvest areas, depending on the weight of equipment, number of passes, soil wetness, and other factors. The upper few centimeters of organic soil can recover quickly from light to moderate compaction (Adams 1991; Burger et al. 1985; Hatchell and Ralston 1971; Kozlowski 1999). The weight of logging equipment used in harvesting and site preparation activities increase soil bulk density by compressing soil macropores. Typically, about three passes of heavy equipment are needed to cause a significant increase in soil compaction (Williamson and Neilsen 2000). The change in pore space diminishes root access to gas exchange, may result in increasingly anaerobic conditions in the soil, limits moisture infiltration and internal drainage, and can lead to increased soil erosion, water runoff, and reduced rooting volume; these changes result in detrimental impacts on seedling establishment and tree growth (Elliot et al. 1998; Greacen and Sands 1980; Williamson and Neilson 2000).

Compaction in mineral soil is not readily ameliorated, and effects can persist for several decades, depending on the severity of compaction and local conditions (Froehlich and McNabb 1984; Greacen and Sands 1980; Landsberg et al. 2003; NCASI 2004; Page-Dumroese et al. 2006). Degree and duration of compaction and effects on tree growth are dependent on climate, moisture regime, soil texture, structure, and organic matter content (Heninger et al. 1997). While new harvest-related equipment and technologies have helped to reduce the effects of compaction and organic matter removal, season of harvest, number of equipment passes, soil texture, and surface organic matter depth all influence the amount of compaction that occurs during harvesting (Page-Dumroese et al. 2009). A number of methods can be used to decrease significant amounts of compaction, including leaving slash in skid trails, increasing equipment operator skill, being aware of soil conditions and properties, or applying biochar and other soil organic amendments (Han et al. 2009; Heninger et al. 2002; Senyk and Craigdallie 1997). Additional actions to mitigate compaction include tilling or ripping and revegetating compacted areas such as haul roads and landings; this is sometimes done on roads that are being abandoned (Luce 1997; NCASI 2004; Sosa-Pérez and MacDonald 2017).

Landscape topography also plays a large role in the amount of disturbance found in forested landscapes. Steep units had less off-trail compaction than flat units because the equipment is usually confined to trails on steeper slopes. Landings, trails, and corridors are usually the locations for soil compaction, and topsoil displacement usually occurs adjacent to many trails (Page-Dumroese et al. 2009). The presence of roads and landings, especially in steep areas, can lead to erosion and other impacts (Neher et al. 2017; Switalski et al. 2004). Occasionally, logging can trigger slope failures (e.g., landslides, mudflows, debris flows) (Guthrie 2002). Indirect disturbances following forestry operations can include changes in microclimate that affect rates of decomposition and nutrient cycling and alter hydrology (Finér et al. 2016; Sun et al. 2017).

The LTSP study found that leaving the forest floor intact after harvesting is critical on many sites to maintain nutrient cycling and C inputs (Powers et al. 2005). Organic matter in woody debris, forest floor detritus, and mineral soil is essential for maintaining ecosystem function by supporting soil C cycling, N availability, gas exchange, water availability, and biological diversity (Jurgensen et al. 1997; Page-Dumroese and Jurgensen 2006). In addition, the buildup of the forest floor may slow the rate of N mineralization, but using fire as a management tool assists in keeping forest floor C/N ratios within ranges more conducive to N mineralization (DeLuca and Sala 2006).

Surface residues often do not increase soil organic matter content (Spears et al. 2003), but using biochar can increase soil C, which can lead to increased soil aggregates and water holding capacity (Page-Dumroese et al. 2017b). Although slash piling is an economical method to dispose of harvest residues and reduce the volume of unmarketable material, pile burning can have short- and long-term impacts that can alter chemical, physical, and biological soil properties and degrade the productive capacity of soils (Rhoades and Fornwalt 2015). These impacts are variable and depend on soil texture, fuel type and loading, weather conditions, and soil moisture (Dyrness and Youngberg 1957; Frandsen and Ryan 1986; Hardy 1996; Rhodes and Fornwalt 2015; Rhodes et al. 2015). In areas where slash piles are plentiful and burned during the fall when conditions are conducive to large heat pulses into the soil, the need for restoration of severely burned soils is key (Page-Dumroese et al. 2017a). Yet, altering the size of slash piles from large (>10 m diameter) to small (<5 m diameter) and adding woodchip mulch have the potential to reduce the need for rehabilitation of burn scars (Rhoades et al. 2015). Guidelines for leaving slash and organic matter in the Pacific Northwest (Forest Guild 2013) and Rocky Mountain forests (Schnepf et al. 2009) are available.

Repeated harvesting without retaining or replacing sufficient amounts of soil nutrients and organic matter leads to continued concerns for loss of fertility and changes in soil biology, particularly on landforms that are weathered from nutrient-poor geological substrates such as granite or quartzite (Bockheim and Crowley 2002; Doran and Zeiss 2000; Federer et al. 1989; Garrison-Johnston et al. 2003; Grigal 2000; Grigal and Vance 2000; Kimsey et al. 2011). Forest fertilization is used in some areas of the United States, typically at the time of planting and during mid-rotation on plantation sites lacking in soil nutrients (Jokela et al. 2010). Fertilizer applications can be cost-effective depending on site conditions, but economic benefits are difficult to predict over the time period involved in growing a stand of timber (Cornejo-Oviedo et al. 2017; Fox et al. 2007; Miller et al. 2016).

WildFire and Prescribed Fire

Wildfires are a keystone process of many forest and rangeland systems, especially in the western United States. Wildfires, whether human-caused or natural, impact the litter layer and associated C and N, alter the environment for soil organisms, and can change the trajectory of forest composition. Fire kills trees and decreases canopy cover, partially or completely burns ground cover, and may form water repellant (hydrophobic) layers in soils, depending on burn severity (DeBano 1981; Madsen et al. 2011). Soil water storage, interception, and evapotranspiration are reduced when vegetation is removed or killed by fire and when organic matter on the soil surface is consumed by fire (Cerda and Robichaud 2009; DeBano et al. 1998; Neary et al. 2005). Fire consumption of ground vegetation and the development of hydrophobic soils increase overland flow erosion and can increase postfire sediment yield (Neary et al. 2005). Some potential secondary effects of severe fires are accelerated erosion and nutrient leaching. However, wildfire can also be beneficial because it reduces hazardous fuel in fire-dominated ecosystems, provides regeneration sites for certain tree and understory species, can increase available water to surviving vegetation, and improves nutrient cycling (Keane and Karau 2010).

As the length of the wildfire season increases due to climate change, the anticipated result is that larger wildfires will occur across the landscape (Abatzoglou and Williams 2016; Jolly et al. 2015). Wildfire concerns have led to an emphasis on accelerated fuel treatments. Busse and others (2014) have summarized the effects of fuel treatments and prescribed fire, noting that in addition to the loss of organic matter, prescribed fire and mechanical thinning operations alter the physical, biological, and chemical properties of the soil. The report encourages land managers to consider the impacts to the soil when planning fuel reduction treatments while acknowledging that these treatments do not make the land completely resistant to wildfire. Similar to the dry western forests and elsewhere, management plans should consider the ecological effects of fuel treatments within a restoration framework to avoid further ecosystem damage.

Many forest stands across the western United States are being thinned to remove fire fuels and reduce the risk of wildfire. Particularly in ponderosa pine (*Pinus ponderosa*) forests, there have been major changes in ecological structure, composition, and processes because of livestock grazing, fire suppression, logging, road construction, and exotic species introductions (Covington and Moore 1994b; Swetnam et al. 1999). These forests are now more susceptible to large, destructive fires that threaten human and ecological communities (Allen et al. 2002). Restoration in these forests requires the need to balance the heterogeneity of ponderosa pine ecosystems and climate fluctuations while also removing large numbers of trees to make the forests more fire resilient and within the natural range of aboveground and belowground C levels (Jurgensen et al. 1997; Rieman and Clayton 1997).

Concerns over large-scale crown fires can be mitigated with hazardous fuel reductions, but these fuel treatments must use ecological principles to limit or prevent further damage (Fulé et al. 2001). Usually, fuel reduction harvesting activities involve cutting and removing small trees with little marketable value (Brown et al. 2004). Residues may be removed and transported to a bioenergy facility (if one is available within a feasible hauling distance), dispersed across a harvest unit by mastication or grinding, or piled and burned (Creech et al. 2012; Jones et al. 2010). Although the impacts of intensive harvests on long-term soil productivity (Powers 2006) are generally known, there is much less known about the impacts of widespread thinning for fire risk reduction. The pattern of disturbance is thought to be different from typical clear-cut and thinning operations (McIver et al. 2003; Miller and Anderson 2002). For example, Landsberg and others (2003) found that the severity of soil compaction on areas thinned for fire risk reduction is dependent on slope.

Wildfire effects and restoration strategies are summarized in Cerda and Robichaud (2009). In the western United States, postfire sediment production may be highly variable, but it can have catastrophic impacts on downstream communities (Moody and Martin 2009). Short-term rehabilitation of burned landscapes tends to be focused on establishing ground cover through mulching or seeding while preventing accelerated erosion. Because of the increased severity and frequency of large wildfires, humans are intervening to assist in postfire ecological recovery efforts (Robichaud et al. 2009). The Forest Service's Burned Area Emergency Response (BAER) program focuses on mitigating unacceptable risks to life, safety, infrastructure, and critical natural and cultural resources on national forests and grasslands. BAER treatments can include erosion control, such as large-scale mulching, to protect municipal watersheds and soil productivity (Beyers 2004; Kruse et al. 2004). The BAER program has been very effective at making timely decisions to protect critical values from short-term damage after fires (e.g., the Hayman Fire in Colorado [Robichaud et al. 2009]). In addition to BAER, long-term postfire restoration focuses on restoring ecosystem function and structure while recovering a level of fire resiliency (Vallejo et al. 2009). Restoration is encouraged in areas where fire is uncommon or fire frequency and severity are outside of the fire regime for the area. Restoration efforts focus on seeding and planting appropriate plant species for the site. In some areas, such as systems managed as wilderness or managed to preserve natural features and ecosystem processes unfettered by humans, it may be desirable to forego restoration efforts after wildland fire. Seeding efforts may inhibit natural regeneration, and soil control measures may lessen the contribution of sediments and associated nutrients in recharging the fertility of streams and lakes located downstream from the burn sites (Christensen et al. 1989).

Restoring fire as an ecosystem process in fire-adapted systems can have beneficial effects (Collins et al. 2009). Over the last century, fire suppression and other management activities have altered the structure and function of forests and rangelands across much of the western United States (Belsky and Blumenthal 1997; Dwire and Kauffman 2003; Hessburg et al. 2005). Forest structure and composition has been most significantly altered due to the lack of fire disturbance. The disruption of the natural fire intervals of the past has resulted in higher stand densities, multilayered stands of mostly one species in some places, and the encroachment of conifers into meadows and grasslands. Dramatically higher stand densities and the development of ladder fuels have increased the risk of uncharacteristically severe wildfire, bark beetle infestations, and in some areas, successional replacement by shade-tolerant competitors. These changes across the landscape increase the probability for disturbances to affect large contiguous areas in uncharacteristic ways. By restoring fire into these ecosystems, generally as part of a management system that includes mechanical thinning and fuel reduction, the forests are restored to lower density stands with higher resiliency to large wildfires and other natural disturbances (Covington and Moore 1994a; Johnstone et al. 2016). Sites that were traditionally savanna ecosystems with widely-spaced trees and grassy understories need more frequent fire to maintain the forest structure (Peterson and Reich 2001). While soil resources are impacted by these management actions, the impacts tend to be less than the results of uncharacteristic large-scale fire disturbances (Hessburg et al. 2005; Johnstone et al. 2016).

Mine Reclamation

After agriculture and infrastructure development, mining is a major driver of deforestation and land degradation in the Americas (FAO 2016; Hosonuma et al. 2012). Millions of hectares of NFS lands are leased for oil, gas, coal, and geothermal operations. In fiscal year (FY) 2015, $1.6 billion worth of products were produced by large mines on NFS lands (USDA FS 2017). As many as 39,000 abandoned mines may be located on NFS lands (USDA FS 2017), and the Forest Service works to minimize or eliminate threats to human health and the environment from these mine sites. Under the BLM, the Abandoned Mine Lands (AML) program aims to restore degraded water quality, clean up mine waste and heavy metal, remediate other environmental issues affecting public lands, and mitigate safety concerns on mine sites abandoned prior to January 1, 1981; currently there are roughly 53,000 abandoned mine sites on BLM lands (BLM 2017). In 2011, the US Government Office of Accountability estimated that the cost of reclaiming 161,000 abandoned mines on public lands was in the range of $10–21 billion. These unproductive abandoned areas are often surrounded by productive forests. They are also usually located in rural areas with rugged terrain and limited access. Eight percent of these abandoned lands contain only physical hazards or limitations and no environmental contamination (American Geosciences Institute 2011).

Several federal and state rules and regulations govern the mining process, from approvals, planning, operations, and finally to reclamation and closure. Reclamation is defined as "the process by which derelict or very degraded lands are returned to productivity and by which some measure of biotic function and productivity is restored" (Brown and Lugo 1994). Mine reclamation rules and regulations vary largely by land ownership and type of extraction and can range from weak to stringent. The Forest Service adopted guidance in 2016 to address short- and long-term postmining maintenance and monitoring after reclamation (USDA FS 2017). Surface coal mining, governed by the Surface Mining Control and Reclamation Act of 1977, has very specific guidelines for topsoil salvage and stabilization. State agencies provide further guidance and regulations for reclamation and bond release. For example, shrub standards for reclaimed coal-mined lands in Wyoming require a minimum of one shrub per m^2 on lands if land use includes wildlife habitat (Wyoming Department of Environmental Quality 1996).

Early research in mine reclamation focused on soil and water protection, thus single-factor approaches (i.e., manipulating one physical, chemical, or biological factor) were employed to protect soil from erosion. Topsoil salvage and reestablishing an adequate plant community to prevent erosion were emphasized and met with varying results (Schuman 2002; Schuman et al. 1998). There are millions of hectares in the United States, including hundreds of thousands of hectares of Forest Service lands that are reclaimed but not restored. Soil quality is still often highly degraded on reclaimed sites. More modern approaches, transpired from years of reclamation practice and research, call for integrated and innovative approaches that target abiotic and biotic processes so that systems are functional (refer to Heneghan et al. 2008; Herrick et al. 2006b; Hild et al. 2009; King and Hobbs 2006; Lamb et al. 2015; McDonald et al. 2016; Stanturf et al. 2014).

Integration of soil stability, hydrology, nutrient cycling, plant functional traits, species turnover and regeneration, and wildlife interactions will not only help unite research with management but can place reclamation within the context of ecosystem function. Commonly, most projects do not have a level of topsoil or subsoil that is reflective of what was on site prior to the disturbance. Overburden generated from open-pit mining can be low in organic matter, soil microorganisms, and plant nutrients such as N and phosphorus and can lack soil structure and texture that are vital to soil fertility and water-holding capacity (Allen 1989; Feagley 1985). Soil organic matter additions are valuable both for their C and the microbes contained in the material and provide substantial benefits to the affected site. The erosion and sediment control industry has started to address topsoil limitations by providing products that help to compensate for the loss of topsoil and setback from topsoil storage (see Abdul-Kareem and McRae 1984). One way they are doing this is by adding compost and other organic amendments that address several aspects of soil health, including the microbiological aspect, particularly ectomycorrhizae (Harvey et al. 1979). Reclaiming some areas may require building soil over rocks that have been dredged from local streams (Page-Dumroese et al. 2018). There are several options to initiate the soil-building process, but applying a combination of biochar, municipal biosolids, and wood chips offers one way to use local resources to begin to restore site productivity. Operations should weigh the costs and benefits of creating soils or adding soil amendments, as these types of treatments can be expensive.

Soils and Problematic Species

One of the most troubling developments related to the ease and speed of travel is the movement of nonnative, invasive species among continents. Across North America, these "problematic" insects, pathogens, and plants pose serious threats to forest and rangeland ecosystems because, unlike natural, abiotic disturbances like wind and fire, they are efficient at changing species composition by targeting specific species or outcompeting native species. Insects that cause substantial tree mortality, such as the mountain pine beetle (*Dendroctonus ponderosae*) that attacks pines (*Pinus* spp.) (Bentz et al. 2010), the gypsy moth (*Lymantria dispar*) (Potter and Conkling 2017), and emerald ash borer (EAB) (*Agrilus planipennis*) (Knight et al. 2012), are powerful drivers of ecosystem and economic change that causes not only shifts in species composition but also changes soil organic matter production, increases coarse woody debris, alters nutrient and water uptake, and changes understory light and temperature (Lovett et al. 2006). In addition, nonnative plants, such as the invasive shrub European buckthorn (*Rhamnus cathartica*) in the midwestern United States, can alter soil chemical and hydrological properties, leaving legacy effects and management challenges (Heneghan et al. 2006). Similar to postfire landscapes, ecosystems altered by problematic species are subject to flash flooding, soil erosion, and sediment loading.

Management responses to insect and disease impacts are often elusive, but in some cases biocontrols and combinations of treatments can be effective (Havill et al. 2016; Margulies et al. 2017) or sites can be transitioned into other vegetation types (D'Amato et al. 2018). Understanding the feedbacks between plant species and soil may help to combat invasion, as exemplified by current research on soil fungal pathogens and cheatgrass (*Bromus tectorum*) (Meyer et al. 2016). The role of mycorrhizal fungi and other fungal species in combating invasive plant species establishment and spread is also noteworthy (Bellgard et al. 2016; Padamsee et al. 2016). Biochar has been shown to increase the growth of native prairie grasses while decreasing or not affecting invasive perennials (Adams et al. 2013). Some technological advances have had adverse effects on forest soils, and management struggles to adapt. Soil ecological knowledge (SEK) has been applied to prevent or reduce the invasion by exotic species during restoration by adding C to promote microbial immobilization of available and mineralized N in abandoned agricultural land (see Heneghan et al. 2006 and references therein), and this has been shown to reduce nonnative plant species cover and colonization (Baer et al. 2003). Carbon addition is a tool to assist community recovery and assembly (nonnative to native). Because biochar is rich in C, it can improve soil quality and increase vegetation growth. In addition, biochar can limit or reduce the growth of invasive species by limiting N availability (Adams et al. 2013; Page-Dumroese et al. 2017b).

Innovative Approaches

Biochar

Land management stresses such as soil compaction, invasive species, disease or insect outbreaks, and wildfire are being exacerbated by a changing climate (Dale et al. 2001). This, coupled with the need to remove encroaching biomass that has little to no value, is increasing operational expenses (Rummer et al. 2005). However, biochar can be one by-product that brings a high value to traditionally low-value biomass. Biochar is a by-product of pyrolysis of materials such as wood, waste organic materials, and agricultural crop residues at temperatures above 400 °C under complete or partial elimination of oxygen (Beesley et al. 2011; Lehmann 2007). Because of its porous structure, large surface area, and negatively charged surface (Downie et al. 2009; Liang et al. 2006), biochar has the potential to increase water holding capacity and plant-nutrient retention in many soils (Basso et al. 2013; Gaskin et al. 2007; Kammann et al. 2011; Laird et al. 2010) and is used to amend food crop soils (Blackwell et al. 2009). Biochar retains much of the C of the original biomass, which can offset the use of fossil fuels and can reduce greenhouse gas emissions from soil (Jones et al. 2010). When used as a soil amendment, biochar contributes to increase C sequestration, enhances the cation exchange capacity, increases pH, and reduces soil bulk density and resistance to gas and water movement (Mukherjee and Lal 2013). All of these changes have been shown to enhance plant growth (Atkinson et al. 2010).

Biochar may be useful for restoring or revitalizing degraded forest, rangeland, and urban soils. It may also provide a method for increasing soil water holding capacity to improve tree health and reduce the incidence of disease and insect attack (Page-Dumroese et al. 2017b). For example, biochar additions of 25 Mg ha^{-1} resulted in a 10% increase in available water in August on coarse-textured soil in central Montana (Page-Dumroese et al. 2017b). The biggest benefit of biochar, however, may be in facilitating reforestation of degraded or contaminated sites (Page-Dumroese et al. 2017b). Biochar amendments have the potential to reduce leaching and bioavailability of heavy metals such as copper, zinc, lead, and cadmium (Bakshi et al. 2014; Beesley and Marmioli 2011), mainly as a result of changing the soil pH. On mine sites that contain toxic chemicals from decades of activity, establishing vegetation cover to limit erosion and offsite movement of chemicals was successful when biochar was used (Fellet et al. 2011).

Variability in biochar type, application rate, and mode (e.g., top-dressing, tilled, pellets), as well as environmental setting, can play a role in plant response (Barrow 2012; Lehmann 2007; Solaiman et al. 2012; Van Zwieten et al. 2010). Applying biochar to coarse- to medium-textured, unproductive soils at rates less than 100 metric tons ha^{-1} can improve nutrient supply, water holding capacity, and water availability (Chan et al. 2008; Jeffery et al. 2011). When adding biochar to the soil to improve moisture conditions (i.e., water repellency), it is more effective when mixed into the profile rather than surface applied (Page-Dumroese et al. 2015). Biochar's ability to absorb water and adsorb nutrients is also contingent upon its chemical and physical properties, a function of pyrolysis temperature (e.g., pH and surface area increase with temperature to a point) (Downie et al. 2009; Lehmann 2007). In forest soil applications, for example, biochar produced at 550–650 °C was better than other temperatures for absorbing water (Kinney et al. 2012). And in a study of different types, in general biochar enhanced water storage capacity of soils, but it varied with feedstock type and pyrolysis temperature (Novak et al. 2012).

Biochar can be designed with characteristics specific to intended objectives, goals, and environmental settings (Novak and Busscher 2013; Novak et al. 2009). Given enough completed studies and data, decision frameworks could help practitioners decide whether or not to use biochar and to determine what type is appropriate based on initial soil properties and other environmental conditions (Beesley et al. 2011). But production costs may outweigh benefits, and it may not be economically feasible for large-scale production and use. Biochar production can cost \$51–\$3747 per ton, a wide range that depends on feedstock type, pyrolysis reaction time (slow or fast), temperature, heat source, and transportation (Meyer et al. 2011). Prices for biochar worldwide vary substantially between \$80 and \$13,480 US dollars per ton (Jirka and Tomlinson 2013). Until there are verified benefits to using biochar and investments in less expensive technologies to produce biochar, the market prices for biochar will remain uncertain (Campbell et al. 2018). On-site production of biochar is one approach to alleviate high transportation and material acquisition costs, especially in forest systems where a constant supply of wood material left over from harvesting operations is available (Coleman et al. 2010).

Seed Coating Technologies

Direct seeding in the western United States is a common restoration practice, but germination and seedling emergence can be major barriers to successful revegetation (Chambers 2000; James et al. 2011). Seedbed conditions are highly variable for temperature and moisture (Hardegree et al. 2003), and conditions need to occur that allow seeds to germinate. For some species, the range of temperature and moisture needed for emergence and growth is narrow (Fyfield and Gregory 1989). Seed coatings that facilitate germination and initial growth may be especially useful in situations where nutrients and water are limited (Madsen et al. 2012; Taylor and Harman 1990). Seed coating technologies that use biochar may potentially overcome moisture and temperature limitations that affect native plant germination and growth, especially on arid and semiarid lands, but initial studies show mixed results (Williams et al. 2016b). The cost of seed coatings can be high, which will add to the already expensive price of non-coated native seeds. During 2000–2014, the US Federal Government spent more than \$300 million on native plant seeds used for revegetating land disturbances; in 2013 alone, the cost exceeded \$20.7 million (U.S. Government 2014).

Soil Transplants

Many of the aboveground changes in plant species biomass and diversity are linked to the abundance and composition of microbes within the mineral soil (Smith et al. 2003). Management-induced shifts in soil microbial populations that regulate nutrients or decomposition will result in a concomitant change in aboveground production (Bardgett and McAlister 1999). Early examination of the mechanisms in which soil microbes influence plant succession and competition (e.g., Allen 1989; Allen and Allen 1988, 1990) led to research and development of soil transplants and inoculum to steer restoration efforts. In recent times, studies have transplanted soils or soil inoculant to restore late-successional plant communities (Middleton and Bever 2012; Wubs et al. 2016). In related efforts, researchers are also using biological soil crusts (BSC) to expedite restoration of severely stressed sites (Young et al. 2016). Biological soil crusts are communities of organisms (fungi, lichens, bryophytes, cyanobacteria, and algae) that are intimately associated with the mineral soil surface (Bowker 2007; also see Box 5.2). These communities are most often associated with rangeland sites, but they can be found ephemerally, and sometimes abundantly, within most terrestrial ecosystems. Soil crusts can facilitate succession, and therefore, the assisted recovery of these crusts may help speed succession on degraded lands. Plants and BSCs interact to help restore soil quality through soil stability, runoff to infiltration balances, surface albedo, nutrient capture, and available habitat for microbes (Bowker 2007).

Monitoring Restoration Success

A restored ecosystem should have the following attributes: (1) similar diversity and community structure in comparison to a reference site, (2) presence of indigenous species, (3) presence of functional groups required for long-term stability, (4) capacity of the physical environment to sustain reproduction, (5) normal functioning, (6) integration with the landscape, (7) elimination of potential threats, (8) resilience to natural disturbance, and (9) self-sustainability (SER 2004).

Monitoring is a crucial part of the restoration effort, yet, in practice, only a few attributes, such as plant composition and cover or soil stability, are generally monitored, and usually these attributes are only tracked for a short period of time (<5 years) (Ruiz-Jean and Aide 2005). Studies support that short-term plant community composition monitoring is a necessary but insufficient predictor of long-term success. Examples of long-term monitoring in the western United States show that short-term monitoring alone of plant community composition has detected “false” and “true” failures. In one situation, a project was abandoned after only 4 years and was determined a failure, but decades later the plant community recovered. The lag in plant community response was attributed to soil properties that need more time to recover (i.e., infiltration and nutrient cycling associated with soil organic matter accumulation). The lack of soil organic matter limited the short-term recovery of the system, so it was deemed a reclamation failure (Tongway et al. 2001; Walton 2005; Walton et al. 2001 as cited in Herrick et al. 2006b). In contrast, many restoration projects deemed successful do not persist because one or more processes are absent (Herrick et al. 2006a; Rango et al. 2005). Integration of ecological indicators that reflect soil and site stability, hydrologic function, and biotic integrity (Pellant et al. 2005) has the potential to help avoid identifying false or true failures in restoration (Herrick et al. 2006b). To understand success in ecosystem restoration, we must understand the linkages of aboveground and belowground changes to biotic interactions, plant community effects, aboveground consumers, and the influences of changing species (Bardgett and Wardle 2010).

Qualities of good monitoring programs include being well-designed and standardized, and there must be long-term support that allows for continuous monitoring. Effective monitoring programs address clear questions, use consistent and accepted methods to produce high-quality data, include provisions for management and accessibility of samples and data, and integrate monitoring into research programs that foster continued evaluation and utility of data. There are several steps involved in planning what to monitor: (1) define the goals and objectives of the monitoring; (2) compile and summarize the existing information; (3) develop a conceptual model; (4) prioritize and select indicators; (5) develop the sampling design; (6) develop the monitoring protocols; and (7) establish data management, analysis, and reporting procedures (Fancy et al. 2009; Jain et al. 2012). Monitoring can be expensive in terms of personnel, equipment, and time, but relative to the value of resources that restoration activities protect and the policy it informs, monitoring costs very little. Considerable planning before restoration begins will determine monitoring needs and overall success. Herrick and others (2006b) suggest a ten-step iterative approach to monitoring that begins before restoration is initiated, collects short-term data for use in adjusting restoration efforts, and lastly, involves long-term monitoring (see Fig. 8.1). This approach allows for short-term monitoring indicators to also be used for long-term efforts.

Monitoring should be scaled both spatially and temporally to the patterns or processes of the response variable, recognizing that patterns often vary with the scale at which a study is conducted (Levin 1992; Wiens 1989). Because processes and populations vary in time and space, monitoring should be designed and conducted at the scale(s) that encompasses the appropriate variation (Bissonette 1997). Given the variety of factors that might influence a response variable, monitoring should be designed to incorporate as much of the variation resulting from those factors as possible. Spatially, this requires sampling at the appropriate scale to detect a biologically meaningful response should one occur. For example, effects may manifest at the landscape level but be obscured at the stand scale, or vice versa (Bestelmeyer et al. 2006). Determining the spatial scale might be particularly relevant when evaluating treatment effects on population trends of selected species (Ritters et al. 1997) because effects discovered within a restoration project area may not extend to the broader population.

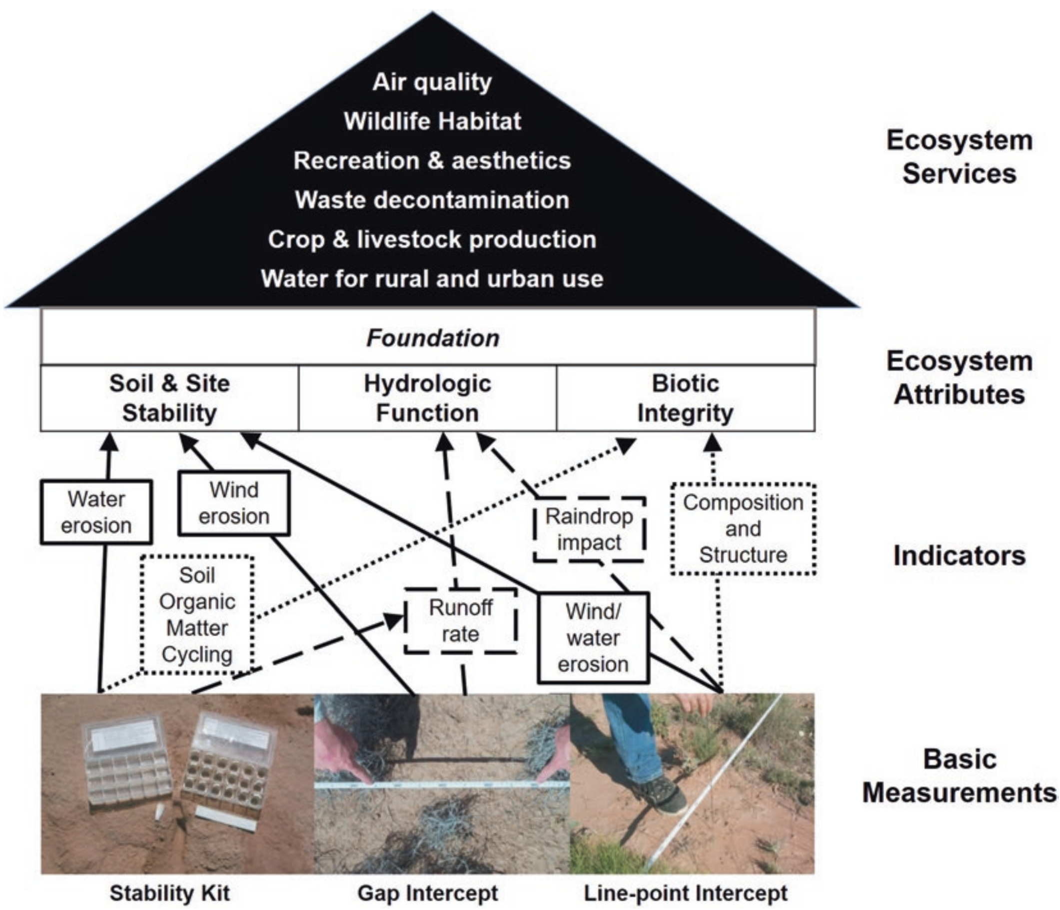

Fig. 8.1 Basic measurements (soil stability, gap intercept, and line-point intercept) used to generate indicators of processes related to ecosystem attributes that serve as the foundation for most ecosystem services and success for restoration projects. (Source: Adapted from Herrick et al. 2006b)

Temporally, the most obvious solution is to conduct studies over a long enough time to detect change if one is occurring (Morrison 1987; Strayer et al. 1986; Wiens 1984). Long-term studies are appropriate when observing slow processes, rare events, subtle processes, and complex phenomena (Strayer et al. 1986). Examples of slow processes include soil organic matter accumulation, plant succession (Bestelmeyer et al. 2006), invasion by exotic species, and long-term population cycles. Rare events include fire, floods, population irruptions of a food item (e.g., insect epidemics resulting in a numerical response by birds), and various environmental crunches (Morrison 1987). Subtle processes are those that may show little change over a short period but whose effects are greater when viewed within a longer timeframe. Complex phenomena are typically the result of multiple interacting factors. This begs the question: How long should monitoring continue? Strayer and others (1986) suggested that "if it continues for as long as the generation time of the dominant organism or long enough to include examples of the important processes that structure the ecosystem under study… the length of study is measured against the dynamic speed of the system being studied." Clearly, ''long-term'' depends greatly on the response variable and system under study. For some restoration projects, a single scale may be appropriate, whereas others might require monitoring to be done at multiple scales. Even then, relationships observed at one scale may differ from those observed at another (Wiens 1986). Timescale is important, but research projects, especially graduate work, may not be long enough to capture any changes.

Conducting long-term studies is important for understanding management and restoration impacts on ecosystem processes (Bienes et al. 2016). In the United States, many government agencies have committed to long-term monitoring to identify ecological insights that inform ecosystem management. The LTSP study has been generating soil and vegetation responses to management activities for the past 25 years (Powers et al. 2005). Other long-term monitoring efforts (e.g., Long-Term Ecological Research, Fire-Fire Surrogate study) underscore management and scientist commitment to generating long-term datasets. While some of these long-term efforts were not meant to evaluate ecosystem restoration, their datasets can inform research and management. For example, the newly developed Land Treatment Digital Library (LTDL), a spatially explicit database of land treatments by the BLM, serves as a clearinghouse for over 9000 land treatments (e.g., seeding, prescribed fire, weed control, vegetation/soil manipulations) in the western United States spanning more than 75 years (Pilliod et al. 2017). The LTDL can be used to study or query, for example, vegetation and soil response (Knutson et al. 2014), successional patterns in relation to management, trends in treatment types across time (Copeland et al. 2018), or ability of treatments to meet habitat requirements for key species (Arkle et al. 2014).

Case Studies

Mower Tract Ecological Restoration: Monongahela National Forest, West Virginia

The Mower Tract Ecological Restoration project is a landscape-scale restoration effort that combines red spruce restoration, watershed development, and creation of early successional habitat to benefit wildlife (Fig. 8.2) (Barton 2014). In the past, timber removal and coal mining left sites devoid of vegetation with severely compacted soils. These sites were subsequently planted with nonnative plant species such as Norway spruce (*Picea abies*) and red pine to control erosion and flooding. These activities left the landscape in a state of "arrested succession," where tree growth was stunted and plant recruitment ceased. A suite of restoration activities is now being used to restore habitat and improve water quality; they include soil decompaction, wetland restoration, woody debris loading, and planting of native trees and shrubs. Heavy machinery breaks up compacted soils, tears up grass sod, and knocks down nonnative trees. The dead wood left on the ground creates habitat for plants and animals while new trees are growing. The downed trees also provide perches for birds to naturally spread native seed and encourage natural regeneration. Organic matter from the decaying wood further improves soil conditions and creates a more suitable environment for growth of red spruce and other native plants. Objectives achieved through this project will help conserve and ensure long-term viability of important plants and animal species associated with unique high-elevation forest and wetland communities.

Long-Term Soil Productivity Study: North America

The North American Long-Term Soil Productivity (LTSP) study is novel as it has been collecting data on soil compaction and organic matter removal after clear-cut harvesting for the last 25 years (Powers 2006). The study spans a wide range of forest and climatic regimes. There are several key points: (1) Soils with an already high bulk density are hard to compact more, and (2) coarser-textured soils recover from compaction faster than fine-textured soils (Page-Dumroese et al. 2006). However, soil density recovery was slow, particularly in soils in the frigid temperature regime. These changes in compaction lead to greater short-term (5 years) tree volume growth on coarse-textured soils and less volume growth on fine-textured soil (Gomez et al. 2002). After

Fig. 8.2 The Mower Tract Ecological Restoration Project. (**a**) The landscape was mined and logged in the 1980s. (**b**) Soil compaction caused stunted tree growth and low recruitment. (**c**) Planted nonnative trees were removed to allow growth of native red spruce. (**d**) Restoration involved ripping the soil, removing nonnative trees, and leaving woody debris on site. (**e**) Aerial view of project (c. 2014). (Photo credit: Chris Barton, USDA Forest Service)

20 years and for a range of soil textures (sandy loams to clay loams), soil compaction resulted in a 15% increase in planted tree biomass on the plot-scale basis. This was attributed to increased seedling survival, along with reduced vegetative competition, and was consistent across all the California study sites (Zhang et al. 2017). This same study points to a

near-complete tolerance of forest biomass growth to compaction and soil organic matter removal; similar to results found in Missouri (Ponder et al. 2012).

Many forest sites are resilient because of their inherent high organic matter levels. However, sites with lower soil organic matter, with deficiencies in one or more soil nutrients, or with fine texture may be at risk for productivity declines as these stands reach crown closure (Zhang et al. 2017). For example, a decade after the complete removal of the surface organic matter, reductions in nutrient availability and soil C concentrations were observed to a depth of 20 cm. Soil C storage was not diminished, likely because of changes in bulk density and the decomposition of residual root systems (Powers et al. 2005).

Soil Matters: Deschutes National Forest, Oregon

Craigg and others (2015) highlight how using soil mapping along with inherent and dynamic soil quality information can help guide forest management for multiple uses. They discuss the importance and value of regarding soils as the foundational resource in forest planning processes. By appreciating the differing inherent capabilities of the soil, land managers can match the appropriate land uses to the soils that will support those uses. This allows the soils information to set the stage for land management by defining the landscape potential and project objectives through understanding how the soils are able to support long-term ecosystem outcomes. Management actions are then planned based on the appropriate soil types in order to have the highest success rate of meeting the objectives. Interdisciplinary teams work together to strategize how to integrate the objectives and the actions. Projects are then designed and implemented with dynamic soil properties that require protection measures. Upon implementation, ecosystem responses can be monitored to determine if anticipated results are achieved. This information then feeds back to the beginning of the cycle to refine land management objectives. Soil types differ widely in their inherent capacity to perform various ecological functions as well as in their dynamic response to and recovery from disturbances. Incorporating these concepts into planning processes can greatly enhance the quality of forest management decisions.

The Sisters Area Fuels Reduction Project (SAFRP) on the Deschutes National Forest of eastern Oregon serves as a case study for the application and potential benefits of this soils-based planning tool. Treatments within the planning area had multiple objectives, including improved forest health and resistance to insect epidemics, drought, and serious wildfires in the wildland-urban interface while also providing quality wildlife habitat and other ecosystem services. The Deschutes National Forest soil resource inventory was used to identify three general soil groups within the SAFRP planning area to help assess and match stand-level tree spatial patterns. Each of these soil groups was then paired with the appropriate management objectives in the project design, and treatments were developed to meet desired resource and habitat goals. Post-treatment monitoring has confirmed expectations of desirable stand patterns and vegetation responses where key soil differences were considered. In addition, the project resulted in a successful fuel reduction treatment which aided fire suppression activities during the 2012 Pole Creek Fire.

Key Findings

- Management and restoration approaches have improved in the last few decades, but innovative developments that inform actions in a timely and cost-effective manner remain priorities.
- Integration of physical, biological, and chemical attributes (a multiple-factor approach) will continue to advance our understanding of soil management and restoration.
- Continuous, targeted, and adaptive monitoring is essential to soil management and restoration, and there is a need for both short- and long-term monitoring.
- Nature is always changing—there is no going back. It takes nearly 500 years to build 2.5 cm of soil. Therefore, protecting and restoring our current soil stocks is critical.

Key Information Needs

- What are the boundaries between healthy, at-risk soils and unhealthy soils? It is important to identify thresholds for soil function and structure of soil types and orders.
- How do we determine the best time to take action and what tools are most appropriate?
- How do aboveground and belowground components of the soil interact? A better understanding of how plants, microbes, organic matter, decomposition, and nutrient cycles interact will help improve soil restoration and monitoring efforts.
- How do we design ecological monitoring efforts to detect fluxes and processes at many spatial and temporal scales? Because ecosystems and soils are in a continued state of flux, we must be able to detect these fluxes to ensure ecosystem services are maintained and land is meeting desired ecological conditions.

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