Soils and Water

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Introduction

Wildlands play a special role in providing a reliable supply of high-quality water (Dissmeyer 2000), and, in particular, we rely on forest and rangeland soils to ensure clean, abundant water. Soils retain water and make it available to support vegetation, facilitate drainage to soil and ultimately to surface waters (streams and lakes), and recharge aquifers and groundwater. Soils also help regulate water quality by filtering out pollutants and regulating sediments. In this chapter, we explore the links between soil and water and evaluate some potential threats to the ability of forest and rangeland soils to provide clean, abundant water. We also identify information gaps and research needs.

Forest and rangeland soils provide important ecosystem services which can be difficult to quantify or describe in terms of their economic value. However, there are examples of the value of sound soil management for protecting water quality. Almost two-thirds of drinking water in the United States comes from forested watersheds and their soils, and many towns and cities depend on water supplies from national forest watersheds (Dissmeyer 2000; Gartner et al. 2014; NRC 2008).

One example of the value of forest, grassland, and other wildland soils and the water they produce comes from New York, NY. The water management bureau chose to ensure drinking water quality for the millions of New York City residents by protecting the upper Catskills watershed. The bureau plans to maintain the watershed in forest land and purchase conservation easements rather than build a filtration plant at an estimated cost of $10 billion, plus $100 million per year in operating costs (Hu 2018). Protecting forested watersheds is a sound economic choice for many water utilities (Gartner et al. 2014).

The economic value of soil can also be described in terms of the impacts of fires and postburn erosion. More than 765,000 m^3 of sediment entered Denver, CO's Strontia Springs Reservoir following the 2002 Hayman Fire, which burned 56,000 ha in Colorado (Robichaud et al. 2003). Denver Water spent $27 million removing debris and sediment from the reservoir. Similarly, the Los Angeles (CA) County Public Works estimated it would spend $190 million dredging four reservoirs impacted by sediment from the 2009 Station Fire (Bland 2017). Keeping soil in place protects water quality and saves money.

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Soils and the Water Cycle

Forest and rangeland soils regulate many important processes within the water cycle (Fig. 3.1). Not only does the soil strongly affect the vegetation, which intercepts, evaporates, and absorbs precipitation and transpires water back into the atmosphere, but the soil also affects surface and subsurface movement of water through infiltration and percolation. All of these processes can influence both the quantity and quality of water from forests and rangelands. Soil also serves as an essential water reservoir and can affect not only vegetation cover and type but also local and regional climate (Bonan 2008). Perturbations that affect soil through the important processes of infiltration, evaporation, surface runoff, and percolation are likely to affect watershed outputs.

To describe the amount of water coming from a watershed, processes and pools are captured in the water balance equation:

$$Q = P + ET + \Delta S,$$

where Q is the runoff or water yield from a watershed, P is the precipitation amount, ET is evapotranspiration, and ΔS is the change in soil storage. The water balance describes the water cycle for a watershed and is usually simplified to an annual basis.

The USDA Forest Service has conducted much long-term watershed research and has evaluated the water cycle in many forest ecosystems, by using paired watersheds and the water balance approach (Hornbeck et al. 1993; Lisle et al. 2010; Neary et al. 2012a; Swank and Crossley 1988; Verry 1997). In forests, streamflow generally increases with amount of precipitation over a year, although the timing and form of precipitation (rain vs. snow) affects that relationship. Interception of rain or snow by the canopy ranges from 2% to 13% in eastern hardwood catchments (Coweeta Hydrologic Laboratory in North Carolina) to 25% in second-growth hardwood forests (Caspar Creek Experimental Watershed in Northern California) to around 30% of precipitation (snow) lost to sublimated interception (Fraser Experimental Forest in Colorado) (Lisle et al. 2010). Water that reaches the ground ultimately infiltrates into the soil or runs off. Water that infiltrates is available for plant uptake and transpiration, and soil and groundwater recharge. Transpiration ranges from 25% of annual precipitation (Caspar Creek Experimental Watershed in California) to 51% (Fernow Experimental Forest in West Virginia) (Lisle et al. 2010). Water draining downward through soil or fractured bedrock ends up in streamflow or groundwater. Streamflow averages around 37–50% of annual precipitation at most long-term instrumented watersheds (Lisle et al. 2010). Therefore, by using the water balance equation, the estimated annual storage within forest watersheds, including soil storage, is about 25% of annual precipitation. Soil storage can vary significantly depending on season of the year, precipitation, type of vegetation, and soil depth.

The water balance of rangelands is perhaps not as well quantified as for forests. Streamflow that drains from rangeland can be difficult to quantify as these areas experience

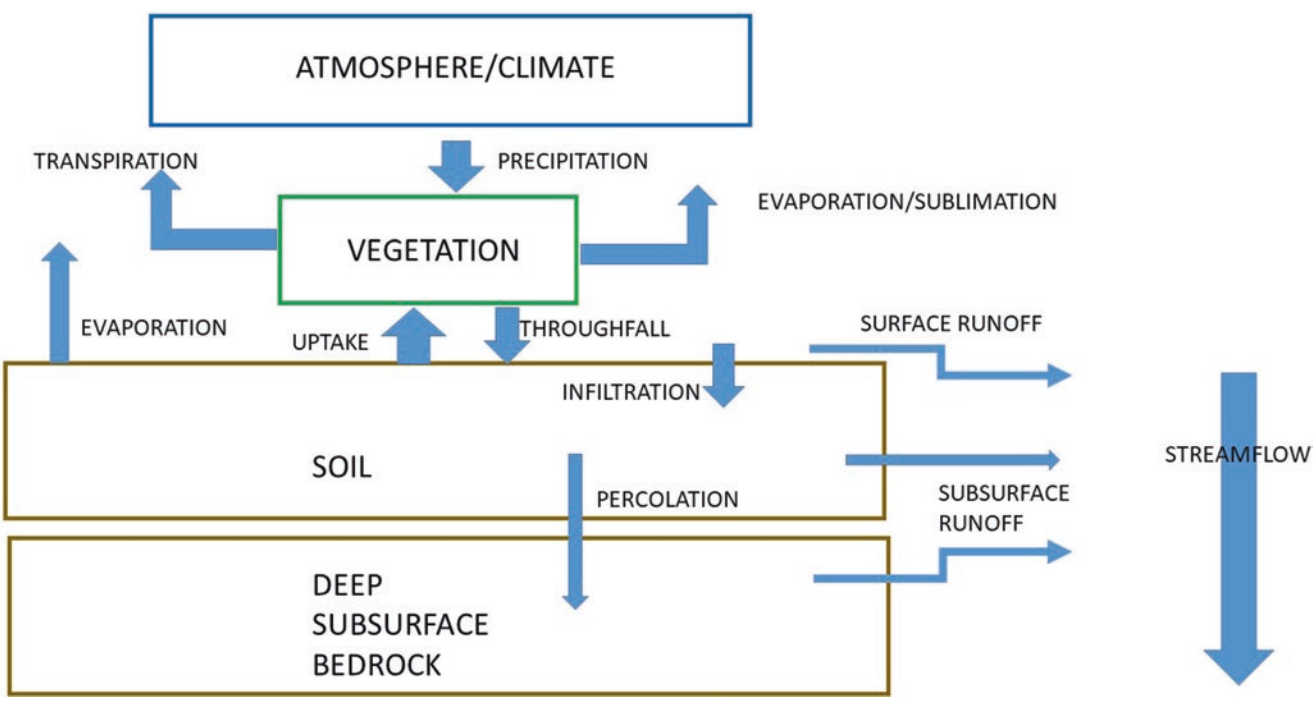

Fig. 3.1 Interactions among soil and water, as quantified in the water balance equation

wide swings in annual precipitation with seasonal pulses of rainfall from winter and summer monsoons. Furthermore, streamwater flowing from rangelands typically has high proportions of groundwater and may emanate from small parts of the watershed. However, evapotranspiration can account for 80–95% of water loss in semiarid rangelands, due to the evaporative draw from high temperatures (Renard 1970). At one research site at the Edwards Plateau in Texas, the annual interception losses ranged from 11% to 18% of precipitation for herbaceous vegetation to as high as 80% (45–80%) for woody vegetation (Wu et al. 2001), although these differences resulted in only subtle changes to overall evapotranspiration. However, soil water status varied with annual precipitation. At this Texas site, grasslands with deep soils sustained higher levels of evapotranspiration than nearby shallow soils with woody vegetation only during drought years, whereas differences were minimal during wet years (Heilman et al. 2014).

In these high-energy ecosystems, water status may relate also to the heterogeneity of the site, that is, the mixture of vegetation life-forms. As Breshears (2006) explained, the network of woody species patches and openings can have patch-scale differences in the amount of water available to plants. Shaded canopies can reduce evaporative losses while also providing organic matter for increasing soil water holding capacity. In degraded rangelands, the effects of that soil heterogeneity can be stark. For example, surface soil in degraded rangelands with honey mesquite (*Prosopis glandulosa*) had much higher water holding capacity and percolation rates under canopies compared to intercanopy spaces subject to wind abrasion and water erosion (Ravi et al. 2010).

Thus, forest and rangeland soils can interact substantially with the water balance of a watershed, mainly through the processes of infiltration, percolation, and soil storage and through relationships with vegetation amount and type. In forests, an intact forest floor (also known as duff, the O horizon, or organic horizon of the soil) is the most important factor affecting infiltration (Aust and Blinn 2004; Kochenderfer et al. 1997). When the forest floor is damaged or removed, infiltration decreases and surface runoff, which is rare in most forests, increases, with concomitant increases in erosion and transport of sediment. Changes in infiltration at the watershed scale will affect other parts of the water cycle, particularly surface runoff (overland flow).

Modeling Soils and the Water Cycle

Models have been developed that explain the basic hydrology of forests (e.g., Liang et al. 1994; Maneta and Silverman 2013; Tague and Band 2004), but they may not always incorporate soil processes well. These models may use simplified representations of soil properties and may not account for lateral redistribution of water through hydrologic flow paths. Furthermore, few models include the dynamics of soil-vegetation interactions (Maneta and Silverman 2013). In particular, issues of scale hinder our understanding of the hydrologic system (Blöschl and Sivapalan 1995). Questions surrounding the sustainability of water (supply and quality) need to be addressed at landscape scales, yet models often are limited by computational demands, data limitations, and the ability to adequately represent processes at that scale (Wood et al. 2002). Recent advances in data science, computing infrastructure, and data availability, as well as new monitoring tools, are providing the opportunity to build a new hydrologic modeling framework. With careful calibration, ecohydrology models can reproduce the dynamics of soil moisture and stream discharge and their interactions with vegetation (Kuppel et al. 2018; Nijzink et al. 2016). Results from these models can contribute to an integrated understanding of water dynamics and biogeochemical cycles.

Although most hydrologic models treat climate and topography as the primary drivers in watershed hydrology, an appreciable body of research has shown that redistribution of water across the landscape is strongly influenced by soil layering and relationship with bedrock (Bathke and Cassel 1991; Kienzler and Naef 2008; McDaniel and Falen 1994; McDaniel et al. 2001; Swarowsky et al. 2011, 2012; Tromp-Van Meerveld and McDonnell 2006a, b). The topography of the soil-bedrock interface is more important than surface topography in describing subsurface flow (Freer et al. 2002). In the same way, McNamara and others (2005) showed that subsurface lateral flow becomes an important hydrologic flowpath when hydrologic connectivity is established. The connectivity of saturated conditions (perched water tables), as controlled by subsurface claypans, was shown to influence streamflow characteristics in headwater catchments (Buttle and McDonald 2002; Detty and McGuire 2010; Newman et al. 1998; O'Geen et al. 2010; Swarowsky et al. 2011, 2012). Though the spatial characteristics of soils influence hydrology, documenting soil variability remains a challenge in terms of both adequately describing it and parameterizing models with these data.

Soil surveys are the standard tool for assessing soil-landscape relationships. However, the scale at which soil surveys are produced in rangelands and forests and the architecture of the data are not always applicable to hydrologic models or fully understood by hydrologic modelers (Gatzke et al. 2011; Terribile et al. 2011). The high degree of variability associated with soil prohibits the mapping at a 1:1 scale; thus, map units are used to depict patterns of soil distribution and hydrologic properties. Ultimately, most landscape- and watershed-scale decisions require soil property data at finer scales than what currently is available. Modelers have noted the need for basic statistical measures (e.g., mean, variance, confidence intervals) on reported soil properties as expressed in soil surveys and the Soil Survey Geographic (SSURGO) database (Brown and Huddleston 1991; Soil Survey Staff 2017), along with some measure of spatial variability (Brubaker and Hallmark 1991).

As with soil surveys, issues of scale pose challenges for hydrologic models (Blöschl and Sivapalan 1995). Process-based models that require a high degree of parameterization can be overwhelmed by data and processing time, limiting their applicability to the hillslope or small catchment scale. In contrast, large-scale (e.g., regional or continental) simulations performed with coarse-resolution data can reproduce discharge from very large basins with reasonable accuracy but are typically not able to resolve local hillslope-scale dynamics (Wenger et al. 2010). Watershed-scale models must aggregate and simplify soil-landscape relationships. Therefore, broad-scale hydrologic models minimize the complexity of soil and rely on primary drivers of the water budget such as evapotranspiration and precipitation. Important characteristics of watersheds such as soil storage, processes that give rise to lateral flow redistribution, and deep percolation are left poorly parameterized, if included at all. As a result of these challenges, no model yet exists to document soil water dynamics at watershed scales.

Threats to the Important Soil Function of Providing Clean, Abundant Water

Forest Harvesting

Forest harvesting effects on the water cycle have been extensively studied on experimental watersheds, many of them mountainous headwater catchments (Neary et al. 2012a). Results generally show that no effect on annual water yield is detectable until about 25% of a watershed is harvested (Hornbeck et al. 1993; Lisle et al. 2010). Above that level, the change in water yield (generally an increase) is related to the amount of the watershed that is harvested, although location of harvest area also matters (Boggs et al. 2016, Kochenderfer et al. 1997). The greatest change in annual streamflow from harvesting was observed in catchments with the highest annual precipitation (Bosche and Hewlett 1982). Recovery of annual water yield generally occurs within a relatively short time in wetter watersheds (eastern hardwoods and coastal Pacific Northwest) but takes longer in more arid ecosystems (Rocky Mountains and Southwest). Other parameters such as high flows can be affected for decades (Kelly et al. 2016).

Harvesting results in (generally) short-term decreases in transpiration and interception, due to lack of or decreased vegetative cover. In general, clear-cutting initially increases soil temperature and soil moisture after harvesting. The length of recovery to preharvesting state is related to the amount of time the soil is bare and the length of time to reach a "full" canopy (leaf area). This ranges from less than a decade in humid eastern deciduous forests to 50–60 years in some arid or Mediterranean western forests (Hornbeck et al. 1993; Lisle et al. 2010).

Changes in infiltration generally are minimal if the forest floor is mostly undisturbed and if care is taken to minimize soil compaction (Aust and Blinn 2004). The increased soil disturbance and water availability caused by timber harvesting can result in slight, but measurable, increases in stream sediment and nutrients. Generally, geologic (background) erosion losses are estimated at less than 500–1000 kg ha^{-1} year^{-1}. In most instances, the increased erosion rates associated with forest harvesting in the Eastern United States are comparable to or less than geologic erosion rates and well below the 2200–11,000 kg ha^{-1} year^{-1} erosion rates that are deemed acceptable and sustainable from agricultural lands (Aust and Blinn 2004). However, these numbers vary widely with forest type, climate, topography, and geology. For example, in coastal Oregon forests, the average sediment yield from unharvested reference watersheds was 180–250 kg ha^{-1} year^{-1} before harvesting (Grant and Wolff 1991). During the 30 years postharvest, however, sediment export increased by 4–12 times, with very large increases due to debris slides and flows associated with a single storm. The pattern of long-term sediment production reflected not just timber harvest but also mass movement history.

When site preparation increases the exposure of bare soil and removes more vegetation, it accentuates water quality problems, especially where sufficient slopes exist. Most water quality problems associated with forest harvesting do not arise from the loss of tree cover; instead, they are the result of poorly designed and constructed roads and skid trails, inadequate closure of roads and skid trails, stream crossings, excessive exposure of bare soil, or lack of adequate streamside management zones (Aust and Blinn 2004; Cristan et al. 2016; Neary 2014).

Grazing of Forests and Rangelands

Grazing most obviously impacts the vegetation biomass and exposes bare soil (Grudzinski et al. 2016; Kutt and Woinarski 2007; Teague et al. 2010) and the surface and near-surface soil. Thus, grazing mostly decreases the processes of interception, infiltration, runoff, and evapotranspiration and increases stream sediment concentrations (Bartley et al. 2010; Grudzinski et al. 2016; Olley and Wasson 2003; Vidon et al. 2008). Diminished riparian vegetation and access to streams by cattle (*Bos taurus*) can result in localized soil erosion from the streambanks greater than twofold that of vegetated streambanks (Beeson and Doyle 1995; Grudzinski et al. 2016; Zaimes et al. 2004). The amount of exposed bare soil and its spatial pattern in the landscape depend on the type of animal grazing. For example, bison (*Bison bison*) create more exposed bare soil at the watershed scale, and cattle create more exposed bare soil in riparian areas (Grudzinski et al. 2016). Excluding all livestock (e.g., cattle and sheep [*Ovis aries*]) and deer (*Odocoileus* spp.) from riparian areas has been shown to reduce soil erosion and decrease suspended sediment in streams (Line et al. 2016; Pilon et al. 2017). Grazed pastureland soils have higher bulk densities and lower infiltration rates and water holding capacities, compared to forest soils, all of which are attributable to compaction rather than differences in particle size distribution (Abdalla et al. 2018; Price et al. 2010).

Grazing may also have large legacy impacts on soil properties, but these impacts are poorly described. Areas that have been overgrazed in the past and are heavily eroded are likely to have altered infiltration over longer periods of time and larger scales (Renard 1970). Indeed, previous agricultural land uses have had long-term effects on surface hydrology in the southern Appalachian Mountains (Leigh 2010; Price et al. 2010), shifting the hydrologic response from soil infiltration and drainage input into streams toward overland flow (Schwartz et al. 2003; Trimble 1985). Nonmanaged systems that are grazed by herbivores may be more dynamic and resilient. Recent research on the global convergence of grassland and pasture structure has suggested that an interplay between bioturbation from roots and soil fauna and biocompaction from grazers creates a stable system of alternating mosaic patches (Howison et al. 2017). Researchers hypothesize that bare soil patches with low infiltration provide water via overland flow to nearby ungrazed patches and may subsequently recover due to bioturbation.

Active management may also ameliorate negative effects of grazing on soil properties. Grazing management treatments increased infiltration on average by about 60%, and the grazing management effect may be slightly larger in more humid environments (DeLonge and Basche 2018). Grazing can also alter infiltration in desert grasslands by damaging biological soil crusts (Belnap 2003). Finally, in semiarid grassland systems, excessive grazing can trigger wind erosion, which results in significant nutrient loss (Neff et al. 2005). The spatial variability of grazing effects points to the need for more research on the scaling of grazing impacts.

Fire and Related Activities

Projected climatic changes may increase drought and fire frequency, particularly in the western United States, and lead to greater stress for forest and rangeland watersheds (Vose et al. 2016). These projected increases in the frequency, size, and severity of wildfire could double the rates of sedimentation in one-third of 471 large watersheds in the western United States by about 2040 (Neary et al. 2005), with sizable effects on stream channel characteristics, water quality for humans and wildlife, and management of extreme flows (Sankey et al. 2017). Fire can alter soil properties that influence infiltration, surface runoff, and erosion and that increase soil water repellency. Fire also affects nutrient cycling and the biological makeup of soils, which mediate nutrient cycling and water quality (DeBano et al. 1998; Neary et al. 2005). These effects on soils translate not only to changes in the amount of water coming from forest and rangeland soils but also to changes in the quality of water. When we think of wildfire effects on soils and water, the effects of wildfire come to mind most often, but activities associated with fire management, fire suppression, and prescribed fires can also have important impacts.

Wildfires usually are more severe than prescribed fires (DeBano et al. 1998), so they are more likely to produce significant effects on water. Fire severity is a qualitative term describing the amount of fuel consumed; fire intensity is a quantitative measure of the rate of heat released. Prescribed fires are designed to be less severe and are expected to have less effect on soils and water. The degree of fire severity is also related to the vegetation type. For example, in grasslands, the differences between prescribed fire and wildfire are small. In forests, the magnitude of the effects of fire on erosion and water quality will be much lower after a prescribed fire than after a wildfire because of the larger amount of fuel consumed in a wildfire. Canopy-consuming wildfires are the greatest concern to managers because of the loss of canopy coupled with the destruction of soil properties and function. These losses present the worst-case scenario for water quality. The differences between the effects of wildfires and prescribed fire in shrublands are intermediate between those in grass and forest environments.

The principal water quality concerns associated with wildland fires are (1) the introduction of sediment, (2) the potential of increasing nitrates in surface water and groundwater, (3) the possible introduction of heavy metals from soils and geologic sources within the burned area, and (4) the introduction of fire-retardant chemicals into streams that can reach levels toxic to aquatic organisms. The magnitude of the effects of fire on water quality is primarily driven by fire severity, rather than fire intensity. In other words, the more severe the fire, the greater the amount of fuel consumed and nutrients released and the more susceptible the site is to erosion of soil and nutrients into the stream, where they could potentially affect water quality.

Fire can produce significant changes in soil physical properties that in turn affect plants and other ecosystem components (Whelan 1995). The effect of fire on soil physical properties depends on the inherent stability of the soil property affected and the temperatures to which a soil is heated during a fire. At or near the soil surface, the sand, silt, and clay textural components have high temperature thresholds and are not usually affected by fire, unless they are subjected to extremely high temperatures (Fig. 3.2) (Lide 2001). Clay undergoes hydration and its lattice structure begins to collapse at 400 °C; internal clay structure is destroyed at 700–800 °C. Sand and silt, which are primarily quartz, melt and fuse at 1414 °C (Lide 2001, p. 81). When fusion occurs, soil texture becomes coarser and the soil more erodible. Temperatures deeper than 2–5 cm below the mineral soil surface are rarely high enough to alter clays, unless heated by smoldering roots. Fires can also affect other soil minerals, such as calcite. Calcite formation occurs at 300–500 °C, but the temperature threshold for formation is not consistent across soil or vegetation types (Iglesias et al. 1997).

Soil structure in the upper horizons can be dominated by organic matter and thus be readily affected by fire if it is directly exposed to heating during the combustion of aboveground fuels. The threshold value for irreversible changes in organic matter is low: 50–60 °C for living organisms and 200–400 °C for nonliving organic matter (DeBano 1990). Soil structure is related to productivity and water relations in wildland soils (DeBano et al. 1998). When soil heating destroys soil structure, both total porosity and pore size distribution are also affected (DeBano et al. 1998). Loss of macropores reduces infiltration rates and produces overland flow. Alteration of organic matter can also lead to hydrophobicity (water repellent soil), further decreasing infiltration rates (see following section).

Soil chemical and nutrient changes during fires can also be dominated by organic matter and may be especially important in coarse-textured soils that have little remaining exchange capacity to capture the highly mobile cations released during the fire. Excessive leaching and loss can result, potentially reducing site fertility, particularly on nutrient-limited sandy soil.

Nitrogen (N) loss by volatilization during fires is of particular concern on low-fertility sites because N is replaced primarily by N-fixing organisms rather than by N mineralization (Hendricks and Boring 1999; Hiers et al. 2003; Knoepp and Swank 1993, 1994, 1998; White 1996). Forest disturbance in general frequently increases both soil inorganic N concentrations (due to reduced uptake by vegetation) and rates of potential N mineralization and nitrification (due to increases in soil moisture and temperature); however, total soil N typically declines with fire. The magnitude of decrease is related to fire severity. Total N, organic matter, and forest floor mass decrease as fire severity increases, whereas concentrations of N in the form of ammonium and exchangeable cations (calcium, magnesium, and potassium) in the soil increase with fire severity (Fig. 3.3). Soil pH generally increases following the loss of organic matter and its associated organic acids, which are replaced with an abundance of basic cations in the ash. In some systems, combustion of monoterpenes in the soil, which inhibit N mineralization, can also serve to increase soil inorganic N

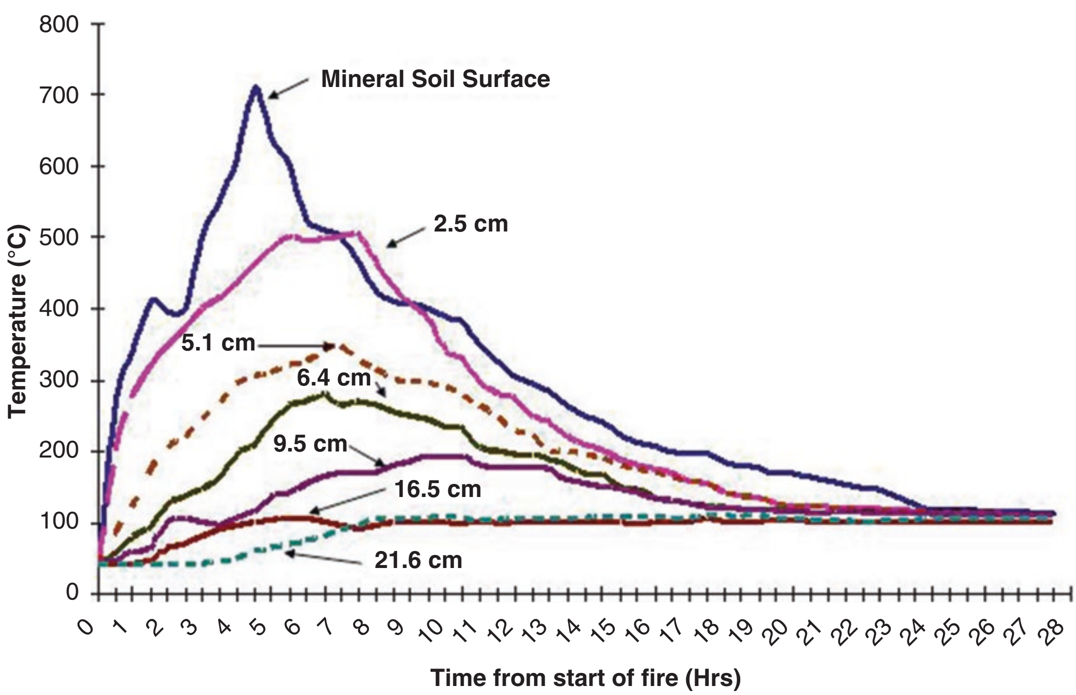

Fig. 3.2 Progression of soil temperature over time, by depth under windrowed logs. (Adapted from Neary et al. 2005; Roberts 1965)

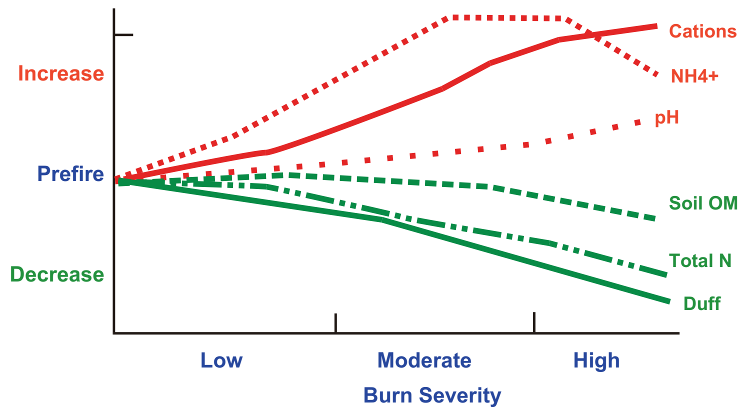

Fig. 3.3 Generalized patterns of decreases in forest floor (duff) mass, total nitrogen (N), and organic matter (OM) and increases in soil pH, exchangeable cations, and N in the form of ammonium (NH_4^+) associated with increasing levels of fire severity. (USDA Forest Service, National Advanced Fire and Resource Institute; adapted from Neary et al. 2005)

following fire (White 1991). The N increases resulting from a combination of changes in soil moisture and temperature, and the decreased plant uptake of N can make more N available for microbial populations in the soil.

Nitrogen is also an important concern relative to water quality. If soils are close to N saturation, it is possible to exceed maximum allowable contamination levels of N in the form of nitrate (10 mg L^{-1}) after a severe fire. On such areas, follow-up application of N-containing fertilizer is not recommended. Further, fire retardants typically contain large amounts of N, and they can cause water quality problems when dropped close to streams (Neary et al. 2005).

Soil microorganisms are complex (Borchers and Perry 1990), abundant, and most likely to experience fire (as they do not leave or go deeper in the soil). How microorganisms respond to fire will depend on numerous factors, including fire intensity and severity, site characteristics, and preburn community composition. In general, however, the effects of fire on microorganisms are greatest in the forest floor and decline rapidly with mineral soil depth, and mortality of microorganisms is greater in moist soil than in dry soil at high temperatures (DeBano et al. 1998; Neary et al. 2005). Most research has documented strong resilience by microbial communities to fire. However, while microorganisms are skilled at recolonizing disturbed forest soils, recovery of microbial and other biotic populations in the forest floor may not occur or may be slowed, particularly in dry systems with slow reaccumulation of organic material. Therefore, minimizing the loss of forest floor is important in prescribed fire. Finally, repeated burning of the forest floor may be detrimental to microbial biomass and activity, although the effects of repeated burning are not well documented.

Soil Water Repellency

Soil water repellency is a soil characteristic that prevents precipitation from wetting or infiltrating the soil. It has been documented in a wide range of vegetation types and climates (DeBano et al. 1998; Dekker and Ritsema 1994; Doerr et al. 2000, 2009). Water repellent conditions are of considerable interest to soil scientists, hydrologists, engineers, and land managers because of the implications for increasing runoff and erosion. Much of the research on soil water repellency has focused on the effects after wildfires, as soil water repellency caused by high burn severity is one of the primary factors in reduced infiltration rates postfire (Lewis et al. 2005). This reduction in infiltration is a primary cause of increased postfire runoff and erosion (DeBano 2000; Shakesby and Doerr 2006).

Burning induces or enhances natural soil water repellency by volatilizing the hydrophobic organic compounds in the litter and uppermost soil layers (Huffmann et al. 2001). Most of the compounds are lost to the atmosphere, but some are translocated downward in the soil profile by the thermal gradient (Fig. 3.4). The decline in soil temperature with depth means that these compounds will condense onto cooler soil particles below the soil surface (DeBano et al. 1976). Laboratory studies show that soil water repellency is intensified at soil temperatures of 175–270 °C but is destroyed at temperatures above 270–400 °C. The duration of heating can also affect the degree of soil water repellency; longer heating times influence the temperature at which these changes occur (e.g., DeBano et al. 1976; Doerr et al. 2004). The effect of wildfires depends primarily on the amount and type of organic matter consumed and the duration and amount of soil heating (DeBano and Krammes 1966; DeBano et al. 1998; Doerr et al. 2004, 2009; Robichaud and Hungerford 2000).

Several studies indicate a positive and significant relationship between soil water repellency and the amount of runoff from rainfall simulations (Benavides-Solorio and MacDonald 2001, 2005; Robichaud 2000), but few studies have rigorously isolated the effect of soil water repellency on infiltration and runoff (Leighton-Boyce et al. 2007). Only recently have these measurements been used to predict infiltration rates. Larson-Nash and others (2018) found significant correlations between minidisc infiltrometer measurements made

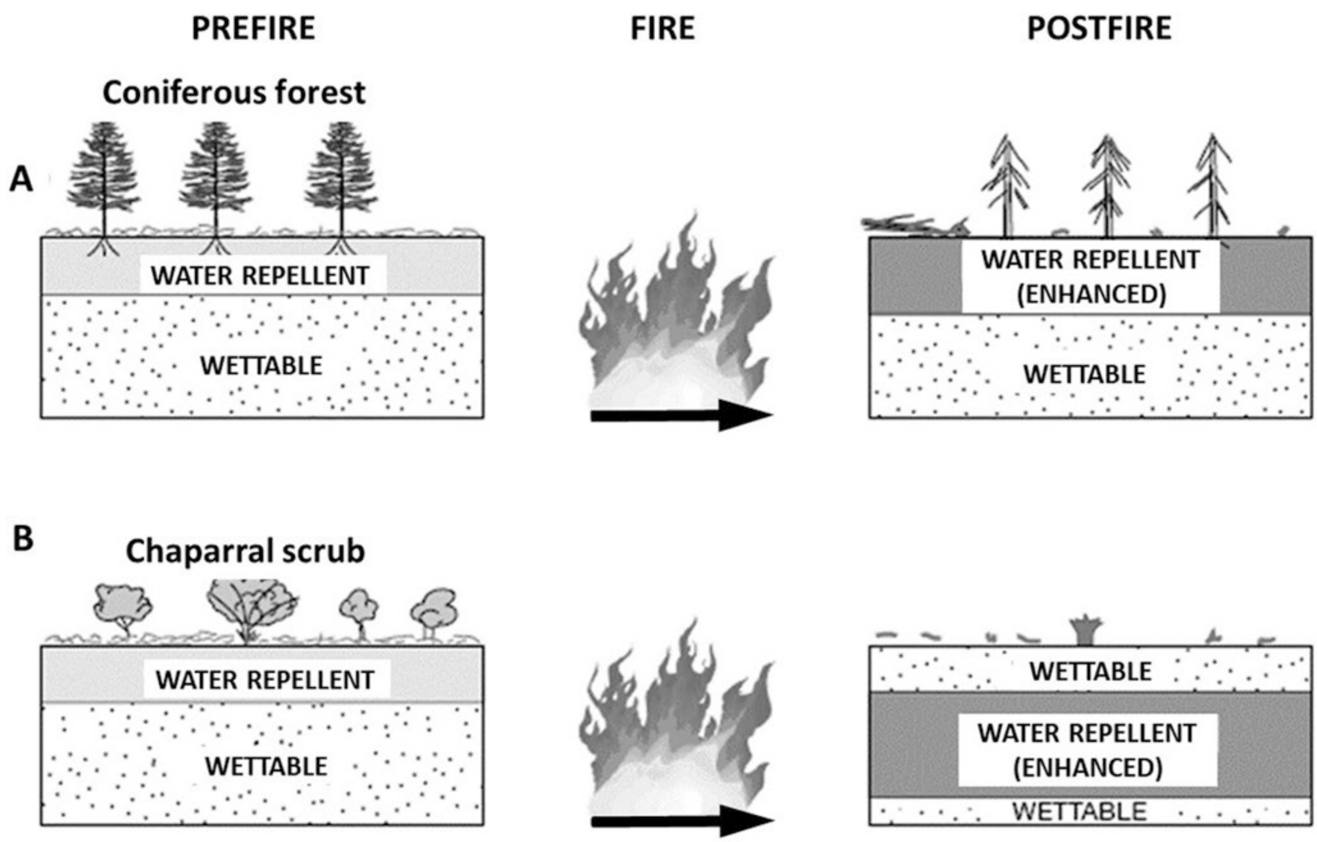

Fig. 3.4 Changes in soil water repellency following fire of moderate or high severity for (**a**) coniferous forest and (**b**) chaparral. Darker shading represents more severe repellency. (Adapted from Doerr et al. 2009)

at three depths to nonsteady-state total infiltration values taken at the 3-, 5-, and 10-min marks within 1-h rainfall simulations. Infiltration rates were reduced by 50–70% immediately after the fire and returned close to preburn conditions 5 years postfire (Larson-Nash et al. 2018).

Soil water repellency changes rapidly in response to changes in soil moisture. Both burned and unburned soils become less repellent or completely lose their water repellency as soil moisture increases. A water repellent soil can resist wetting for days or even months (e.g., Dekker and Ritsema 1994; King 1981), but the presence of macropores or other preferential flow paths means that water will eventually enter the soil. Over a period of months after the fire, soil water repellency decays toward prefire conditions. As a water repellent soil dries out, the soil water repellency is often reestablished. A series of wetting and drying cycles may eventually eliminate the soil water repellency induced by burning.

Temporal and spatial variability of soil water repellency is large, thus making it difficult to determine the single effect of soil water repellency on runoff rates at the watershed scale as compared to the point and plot scales (Woods et al. 2007). However, the greatest influence of soil water repellency on erosion is its potential for increasing overland flow. As the amount of overland flow increases, so do its depth and velocity and hence the ability of the water to scour and transport particles by sheetwash and interrill erosion (Robichaud et al. 2016). The concentration of overland flow can initiate rill erosion (Benavides-Solorio and MacDonald 2005), and the topographic convergence of water at larger scales can result in gully, bank, and channel erosion (e.g., Martin and Moody 2001; Neary et al. 2012b).

Natural Gas Development

Recent advances in drilling technologies have dramatically increased the exploration for and extraction of natural gas in many areas of the United States. However, relatively little is known about the effects of natural gas development on the soils and water cycle processes, in either the eastern or western United States, due to a paucity of research. Other forms of energy development are also increasing but have been studied to some extent. Many of the processes involved in natural gas development—clearing vegetation and site preparation for well pad and pipeline construction—are similar to forest harvesting and land conversion; thus, some conclusions from that research may apply to disturbances during natural gas development. As noted earlier, forest harvesting research has shown that changes in streamflow often are not detectable until around 25% of a watershed is harvested (Hornbeck et al. 1993; Lisle et al. 2010). Thus, individual gas well pads may have little impact on hydrologic processes at larger watershed scales, depending on their location within a watershed. The cumulative effects of many well pads and roads in an area of intensive development (Fig. 3.5) may nevertheless be significant. More research is needed to document conditions that lead to impacts on the quantity of water in a variety of vegetation types and at various scales.

Further, relatively little research has been conducted on the effects of gas well development on surface water quality, although it is believed that surface water quality impacts are likely to be local and result from handling of the large amounts of water required for the hydrofracturing (much of which comes back up out of the well) and from accidental spills. Land application of fracking fluids is seldom permitted, and most fracking fluids are now recycled and reused rather than disposed of on land surfaces or down injection wells. However, localized effects on soils from accidental spills can be significant and may have implications for water quality and potentially water quantity. For example, surface soil concentrations of sodium and chloride increased 50-fold as a result of the land application of hydrofracturing fluids to a hardwood forest in West Virginia and declined over about 2 years to background levels (Adams 2011). The event resulted in major vegetation mortality through direct contact and uptake of soil solution by the trees. Significant impacts on soil chemistry persisted after removal of the flowback storage pond; vegetation failed to reestablish because the high levels of soil salts attracted deer (Adams et al. 2011). The effects on surface water were not evaluated, and such studies are lacking in the literature. Major effects on groundwater quality are not predicted from natural gas development, although concerns still exist, particularly relative to methane (Osborn et al. 2011). The concerns will vary with the type of well, its depth, and cementing technologies used.

Erosion from construction activities associated with the development of the well pad and associated infrastructure can be high and is related to length of time during which the ground-disturbing activities take place, precipitation, slope, best management practices (BMPs) used, and the rate of revegetation (Adams et al. 2011). Williams and others (2008) reported substantial sediment runoff from natural gas development in rangelands, which diminished as natural revegetation occurred. The estimated annual sediment loading from the natural gas field was almost 50 times greater than from typical undisturbed rangelands. Similarly, sediment concentrations and yields from a newly constructed natural gas pipeline were highest initially after completion, averaging about 1660 mg L^{-1} and 340 kg ha^{-1}, respectively, during the

Fig. 3.5 Natural gas development (well pads and roads) in Northwestern Pennsylvania. (Source: Google Earth)

first 3 months following completion of corridor reclamation. As revegetation of the right of way progressed, sediment concentrations and yields declined (Edwards et al. 2017). Early runoff and sediment movement occurred despite heavy straw mulch, suggesting that more than the usual BMPs may be necessary in forests and rangelands with steep slopes. Erosion from a pipeline installed in an existing skid road resulted in higher erosion rates, due to the negative effects of soil compaction on vegetation succession (Edwards et al. 2014). Therefore, during and after construction of gas well pads, reducing compaction to encourage infiltration and successful vegetation establishment is essential for controlling sediment losses.

Development for Recreational Activities

Recreational development and use have increased dramatically in recent decades in wildland areas (Hammett et al. 2015). Recreational use has intensified from primarily foot traffic to a wider variety of recreational activities including riding all-terrain vehicles and snowmobiles, mountain biking, snowboarding, and other activities.

The process of trampling by foot traffic has been recognized from the earliest studies as a major cause of adverse impacts on soil resources in recreation areas (Bates 1935), with implications for water quality and quantity (Fig. 3.6) (Manning 1979). These impacts can result from foot traffic by humans and horses; use of sports equipment such as mountain bikes, all-terrain vehicles, and skis; and management activities, such as grooming ski trails.

Soil erosion is the most widespread trail impact, because trails typically rely on, or result in, a bared soil surface. Sediment generation from trails will increase proportionally with recreation traffic since the shear stress frees soil particles available for transport, which may result in movement of soil from the surface to other areas (Olive and Marion 2009). In general, motorized and equestrian travel will displace trail soil surfaces more than human foot traffic (Cessford 1995; Newsome et al. 2004; Pickering et al. 2010). Careful trail planning can mitigate trail erosion by siting trails on side slopes where water bars and drainage-control features can limit erosion (Marion 2006). However, erosion along unauthorized routes can persist as travel on such routes may expose soil and damage or destroy vegetation that does not readily recover. Surface compaction from recreation activities on unauthorized routes can impede infiltration into soil. This compaction dries out soil along these tracks and paths (Settergren and Cole 1970), which can limit vegetative growth and further limit infiltration and percolation. Trail building helps to protect natural soil and vegetation communities by concentrating traffic and ensuring adequate storm drainage. Most research has evaluated localized impacts to water quality, because cumulative impacts from recreational activities are difficult to detect at larger watershed scales. Recreational activities affect water quality primarily via sediment delivery at trail crossings with streams and along the edges of water bodies (Hammett et al. 2015).

Winter sports resorts also affect soil and water resources, whether from creating and maintaining ski runs, grooming natural snow, or supplementing natural snowfall with artificial snow. The process of creating ski and snowboarding trails can alter temperatures at the soil surface, through compaction of snow (Rixen et al. 2003), and can lead to severe soil frost, with resultant impacts on vegetation, and a potential for increased erosion. At the highest altitudes, the processes of leveling to create ski trails can accelerate the thawing of permafrost (Haeberli 1992). Because of the

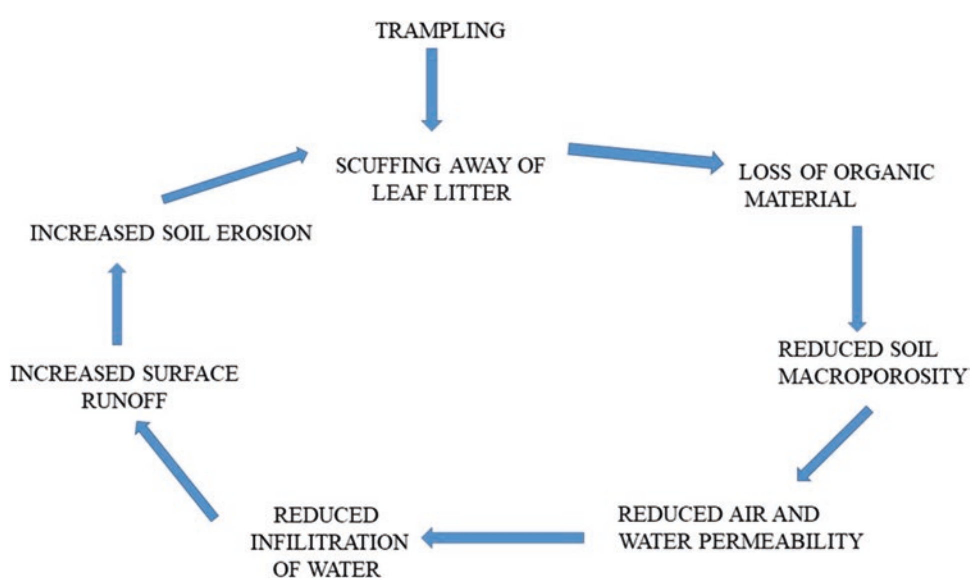

Fig. 3.6 An example of how trampling during recreational activity affects soil and water resources. (Source: Redrawn from Manning 1979)

demand for longer ski seasons and increases in both the number and size of resorts to meet this demand, many ski resorts in the United States rely on snowmaking: In 2001, almost 90% of US resorts supplemented natural snowfall (Rixen et al. 2003). With increasing temperatures and climate change, this percentage is likely to rise. Making of artificial snow requires large amounts of water and creates a more granular snow. This artificial snow has a different chemical makeup than natural snowfall, which has implications for soil and water quality. Physically, adding artificial snow increases the depth of snow. Thus, the insulating value of snow is enhanced, decreasing the incidence of soil frost and possibly the erosion potential. However, the additional mass of artificial snow takes longer to melt, leading to prolonged snow cover in the spring, which also has implications for vegetative development (Rixen et al. 2003). The additional water (possibly up to five times the amount from natural snow) may increase water yield and increase the erosion potential (Wemple et al. 2007). Diverting water from local streams and lakes to manufacture snow also raises the likelihood of drying streams in the summer. The consequences of adding snow, whether using additives or only natural, locally available water, on water quality have not been widely studied.

Soil Pollution

Soil pollution is a global matter of concern, particularly as it relates to issues of food security and safety. It often is more a focus in developing nations than in more developed countries. Soil pollution is also often considered to be more of a problem in urban soils (see Chap. 7). However, soil pollution can be a local problem, as well as a regional, national, and international problem, with implications for wildland forest and rangeland soils and their ability to provide clean, abundant water. The pollutants of most concern are metals (such as lead, cadmium, and arsenic) and organic compounds (such as pesticide residues and by-products) (O'Connor et al. 2018). Though some of these substances occur naturally in the soil, they also originate from vehicle exhaust, waste disposal, untreated sewage disposal, industrial emissions, and, in the case of pesticides, direct application.

Important, but understudied, are soil contaminants carried in dust, including microbial organisms. Over the last two decades, dust emissions have increased by up to 400% in wildlands in the western United States (Brahney et al. 2013). Measurements of dust chemistry have shown that dust can transport appreciable amounts of phosphorus and carbonates to remote wildland areas (Lawrence and Neff 2009) and that heavy metals are also often transported in dust. The distribution of pollutants in soil is highly spatially variable, although generally contaminants are more concentrated in surface soil. Concentrations of contaminants tend to be greater at the surface because of deposition to the surface in wet or dry form and because organic matter content, which often binds metals and organic pollutants, is greater in surface horizons (Kaste et al. 2006). However, when the pollutant source is removed, the pollutant may "disappear" from the surface soil, only to be found deeper in the soil profile or ultimately in water bodies (Smith 1976). The pollutant may also be broken down or transformed into other compounds without moving through the soil (Kaste et al. 2006).

Soil contaminants can be taken up in vegetation and pose a health risk if the vegetation is consumed by humans or livestock. However, contaminants may also have effects on growth of plants and can affect water quantity and quality if the effects are sufficiently severe. For example, soil pollution from a smelter in Copper Hill, TN, which operated between 1843 and 1959, continues to suppress vegetative recovery (Raven et al. 2015). Research shows that soil acidification can increase the availability of heavy metals (Maiz et al. 2000), which has implications for soils still subject to the legacy effects of decades of acid deposition. As deposition of air pollutants decreases, the recovery of soil from acid precipitation is poorly understood and is a research need.

Pollutants can be moved from a site via surface runoff, groundwater flow transport (Mandal and Suzuki 2002), dust transport (Prospero 1999), and weathering (Nriagu 1989). Recent research has evaluated techniques for ameliorating soil pollution, and much of the focus has been on organic matter amendments, including the use of biochar (Ahmad et al. 2014). Only occasionally has biochar been evaluated in settings other than agricultural fields and developed beyond the research phase to operational application. Bioremediation, using plants or other organisms to remove pollutants from the soil, is another promising method and needs development beyond the research phase.

Priority Information Gaps

Linked Soil Climate Information

Soil moisture is a critical component in evaluating water budgets and assessing drought, yet monitoring of soil moisture on a national scale has its limitations. Assessment of soil moisture and soil temperature focuses on three sources of information: in situ sensors, remote sensing, and modeling. In situ measurements typically have sensors at various depths between the land surface and 1 m. These stations are valuable sources of information, such as direct measurements of soil moisture and temperature, but spatial distribution varies among states and regions; many areas have sparse or no coverage. Because each in situ network is installed and managed by different federal, state, local, tribal, university, and private industry groups, there is a lack of consistency in sensor depth, sensor type, transmission of data, data format, data availability (see Cosh et al. 2016 and the following discussion for details), and funding and network coordination. In addition, data are severely lacking in forested environments, where the tree canopy can limit data transmission and where root zones often exceed the 1 m depth typically used.

Remote sensing, both from fixed-wing flights and satellites, provides large spatial coverage of soil moisture but usually at a coarse grid resolution, and does not perform equally well everywhere. For example, the National Aeronautics and Space Administration's Soil Moisture Active Passive (SMAP) satellite provides soil moisture to 5 cm, but does not work well in forests because of the tree canopy (Panciera et al. 2014). In addition, there are temporal limitations due to flight schedules and satellite passes, with gaps of days to weeks as well as time needed to process data. Models, such as the North American Data Assimilation System (NLDAS-2) (Mitchell et al. 2004), offer spatial coverage across the country and use various sources of data, but need to be closely calibrated to in situ and satellite data, and generally have a coarse resolution.

A collaborative effort is underway to develop a National Soil Moisture Network (http://nationalsoilmoisture.com) that includes Federal, state, local, tribal, university, and private industry, as well as citizen science, data collection activities to develop a single dataset and map product. The goal is to combine in situ measurement, remote sensing, and modeling results into a single dataset that is used to develop a gridded map product of soil moisture with daily updates. The utility and application of such a product would benefit the United States Drought Monitor (https://droughtmonitor.unl.edu). This product would also serve as a tool for assessing hydrologic conditions related to agriculture, flooding, fire potential, and other applications at a sufficiently fine spatial resolution across the country.

Expanded Soil Moisture Monitoring

There are many areas where regular, direct measurements of soil moisture are sparse to nonexistent. On a national scale, the USDA operates the Soil Climate Analysis Network (SCAN), which consists of 218 stations in 40 states, Puerto Rico, and the United States Virgin Islands (https://www.wcc.nrcs.usda.gov/scan/). The National Oceanic and Atmospheric Administration operates the Climate Reference Network (CRN), which has 137 stations across the country (https://www.ncdc.noaa.gov/crn/). In addition, the USDA Snow Telemetry (SNOTEL) network in the Western United States has 446 stations that have soil moisture sensors as part of the data collection program (https://www.wcc.nrcs.usda.gov/snow/). These stations are especially important because they are in higher elevations, which are typically data-poor areas for soil moisture, and generally in forested locations, which are also underrepresented in databases. The North American Soil Moisture Database consists of over 1800 in situ soil moisture stations in the United States, Canada, and Mexico (Quiring et al. 2016). These stations generally are concentrated in certain states or regions, so there are many areas with sparse data coverage. In addition, the Remote Automated Weather Stations (RAWS) network (https://raws.nifc.gov), an interagency effort of various wildland fire agencies that provides data to the National Interagency Fire Center, has about 2200 stations, some of which have soil moisture sensors (https://famit.nwcg.gov).

Although these national networks, in conjunction with regional, state, and local networks, provide soil moisture data from across the country, the data lack adequate density of spatial distribution, especially in forested regions. Moreover, many of the networks have a short period of record, typically less than 10 years, which makes it difficult to evaluate trends. In addition, comparing data among networks is hampered by different sensor types, depths, temporal data collection and transmission variations, data format, data type (volumetric water content vs. percentage), data access, and many other factors.

To improve on data collection efforts, a systematic approach to standards and specifications is needed. A more consistent approach would help ensure similar data quality and accessibility, adequate spatial coverage of data collection, and a single source of data storage and product generation to enable users to easily and effectively access and use data, tools, and products for assessing soil moisture.

Continued Support for Hydrologic Monitoring Networks

In the twentieth century, experimental forest catchment studies played a key role in understanding the processes contributing to high water quality (Neary 2016; Neary et al. 2012a). The hydrologic processes investigated on these catchments provided the science base for examining water quality responses to natural disturbances such as wildfire, insect outbreaks, and extreme hydrologic events and human-induced disturbances such as timber harvesting, site preparation, prescribed fires, fertilizer applications, pesticide usage, acidic deposition, and mining.

Another approach is the broad-scale landscape monitoring approach. The United States Geological Survey uses a landscape monitoring approach to acquire data on water resources from over 7200 gauging stations to report on the status and trends of water resources in the country (Neary 2016). It also uses data from cooperators to assemble information on 1.5 million sites in the United States. Landscape-level monitoring is important for discerning trends in national water resources.

Both methods need to be maintained in the light of a changing climate, and both approaches need to link more closely with available soils data and models. Some of the well-understood water-soil relationships could be altered by a more dynamic atmosphere and changing weather phenomena. There will need to be continued solid commitments from scientific organizations, government agencies, and private organizations and enterprises to achieve this goal.

Key Findings

- The key soil process with relevance to water quantity and quality from forests and rangelands is infiltration.
- Erosion and sedimentation remain major water quality concerns in forests and rangelands.
- Soil moisture is largely unexplored in most investigations and is often poorly quantified, particularly at depth.
- Current hydrologic models do not document or model soil water dynamics at watershed scales.
- Though some new threats to soil and water (e.g., natural gas development) can be understood in the context of traditional watershed research (e.g., harvesting impacts), this concept is less useful when processes other than infiltration are affected or when the spatial complexity is high.
- The legacy of twentieth-century paired catchment studies provides a solid framework for evaluating and predicting hydrologic and soil changes in the twenty-first century and beyond.

Key Information Needs

- Documenting soil variability remains a challenge in terms of adequately describing it and parameterizing models with these data.
- New tools should be explored to accurately quantify storage capacity and water dynamics at a variety of scales. The predictive capacity of terrain-based digital soil modeling needs to be further explored as a way to downscale soil surveys.
- Tools are needed to document the characteristics, storage capacity, and water utilization of deep soils and the rock materials overlying bedrock.
- Few studies have rigorously isolated the effect of soil water repellency on infiltration and runoff. The large temporal and spatial variability of soil water repellency makes it difficult to determine the single effect of soil water repellency on runoff rates at the watershed (catchment) scale, as compared to the point and plot scales.
- The effects of natural gas development and other energy and resource development on soils and the links with surface water quantity and quality in forest and rangeland soils need to be evaluated at the local and cumulative scales.
- Few studies have evaluated the cumulative impacts of recreational and trail development, particularly relative to water quality. Most studies have evaluated impacts of such development on a local scale.
- The consequences of adding snow, whether using additives or only locally available water, on water quality are not well described.
- More research on the scaling of grazing impacts on water movement and quality is needed.
- Soil pollution risks in forests and rangelands should be mapped and evaluated.
- The impacts of repeated fire and its effect on soil properties, including soil biota, should be evaluated in a spatially explicit way.
- Recent research to evaluate techniques for ameliorating soil pollution has focused on organic matter amendments. This research needs to be expanded and developed beyond the research phase to operational application. Consistent monitoring protocols and coordination among agencies are also needed.

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