Chapter 3

Active Forest Management and Community Water: Issues and Interactions

Jeff Behan

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Oregon has some of the most productive forestland in the world. Timber harvesting and associated activities played a key role in Euro-American settlement and development in Oregon, and remain a significant sector in the state's economy. For years, Oregon has led the nation in softwood and plywood production. Oregon's forest industry supports more than 60,000 jobs (OFRI 2017).

Forested watersheds — both managed and unmanaged — also produce higher quality water than any other type of surface water source area and supply drinking water to most of Oregon's community water systems. Forest management practices, including methods for road construction and use, harvesting and site preparation and chemical applications, have markedly improved in recent decades. But forestry can still impair downstream water quality in a number of ways, primarily through the construction, use and maintenance of forest roads, but also silvicultural activities, mainly from when trees are harvested through the first decade or two into the new rotation. Forest roads and active forest management have also been shown to impact the volume and timing of water delivered from watersheds.

This chapter provides an overview of interactions between forestry and water, best management practices to reduce water quality impacts, and some remaining issues and concerns.

Sections 3.1 and 3.2 summarize mechanisms and functions regarding:

- The stability and movement of soil and sediment in forest environments.
- Water collection, retention and production in forested watersheds.
- How forest management can affect these functions.

This summary is drawn primarily from forest hydrology text books (Chang 2012, Amatya et al. 2016 and chapters therein) supplemented with more focused peer-reviewed journal articles reporting on individual studies and literature reviews.

Section 3.3 explores the use of synthetic forest chemicals (herbicides, pesticides and fertilizers), and forest management actions that can modify the production of naturally occurring compounds, such as natural organic matter and nitrates that may affect water quality. Section 3.4 discusses natural organic matter and disinfection byproducts.

Section 3.5 provides a brief history and overview of Best Management Practices for mitigating the effects of forest management on water resources.

The intention with this chapter is to summarize generally established and accepted science knowledge and concepts regarding these topics to complement more detailed discussion focused more specifically on relationships to drinking water in Chapters 4, 5, 6 and 7.

3.1. Forest management and stream sediment

Undisturbed forests have high infiltration rates and little overland flow. Precipitation usually passes through the soil before reaching streams, minimizing erosion and sedimentation and producing high quality water (Stednick and Troendle 2016; Williams 2016). Forestry activities such as road building and timber harvesting disturb soils, which may then be mobilized by water or wind. Consequently, forest operations that increase the erosion, transport and deposition of forest soil into waterways have long been subject to intense focus from stakeholders, researchers, policymakers and practitioners. Wind erosion can be an issue in drier forests (e.g., Whicker et al. 2006), but water erosion is of primary concern, especially in wetter forests, and will be the focus here.

Stream sediments are soil and mineral particles. These particles are usually inorganic but sometimes partly organic, detached from the land by processes that include raindrop impact, surface runoff, streamflow, wind and gravity, often in association with human activity. Sediment inputs that result from human activities frequently impair the physical, chemical and biological properties of streams and degrade beneficial uses. Sediment is a leading cause of stream impairment nationwide and in Oregon (USEPA 2017). Sediment can affect water turbidity, chemical composition, taste and odor. It can also interfere with drinking water treatment. Sediment concentration and yield are widely accepted indicators for the effectiveness of watershed management practices.

3.1.1. The water erosion process

Water erosion is a three-step process consisting of soil detachment, transport and deposition. Raindrops detach soil particles and generally increase in size and terminal velocity with rainfall intensity. Once detached by striking raindrops or overland flow, soil particles are transported by runoff. The distance soil particles travel depends on soil properties, topography, runoff energy and surface conditions. Sediment is deposited when the soil carrying capacity is less than the weight of the soil particles. The ability of soil to resist detachment from raindrop impact and surface flow generally increases with increasing organic matter content and infiltration rate. Depending on conditions and disturbance history of the area, a soil particle can move into a nearby waterway during one rainfall event or over decades or centuries (Chang 2012).

Water erosion in watersheds occurs at a range of scales. Smaller-scale processes include interrill and rill erosion on side slopes and ephemeral gully erosion along shallow drainage ways. These in turn feed downslope into more deeply incised gullies that form when converging eroded drainage ways reach a certain size. Gully erosion generally occurs in well-defined drainage ways and involves soil particle detachment by flowing water and slumping of unstable banks, and transport by flowing water. Sediment loads are often greater downstream due to the additive effect of interrill and rill erosion from adjacent areas and detachment of soil particles upstream in the drainage way. Sediment transport capacity increases downstream along with flow volume.

Gullies are an advanced stage of water erosion and are permanent unless they are actively filled. Without active conservation and mitigation measures, gullies will continue to expand and grow via down-cutting and head-cutting. Down-cutting deepens and widens gully bottoms. Head-cutting extends channels upslope into headwater areas and expands the gully tributary system. Deep gullies may extend to the watershed divide. Poor road layout and construction often accelerate gully development.

In contrast to gully erosion, which occurs in the upper ends of headwater tributaries with water flowing primarily during or immediately after storms, channel erosion occurs in the lower end of headwater tributaries where water flows on a continuous basis. Channel erosion consists of soil erosion on streambanks and sediment transport in the stream channel. Streambank erosion is frequently caused or exacerbated by removal of vegetation.

Slope failures or mass movements, including landslides and debris flows, generally are larger scale processes involving the downhill movement of significant volumes of soil, rocks and organic matter under the direct influence of gravity. Slope failures usually occur in areas with steep slopes and weak geological structures. They are often triggered by some combination of factors or events, including intense and prolonged rainfall, snow buildup and melt, converging overland flows and seepage, earthquakes and forest harvesting. A decrease in slope stability can be caused by increased water content, reduced internal soil cohesion and a higher groundwater table resulting from increased precipitation or deforestation (Chang 2012).

Pore water pressure (or just pore pressure) is the pressure exerted by water held in pore spaces in soil. When a soil is fully saturated, pore-water pressure is said to be positive. Pore pressure rise resulting from rainfall or snowmelt is the most common triggering mechanism for landslides. Positive pore water pressure develops just above a restrictive layer (e.g., bedrock) in rapid response to rainfall infiltration, causing soil shear strength to decrease to the point at which the slope fails. In addition to storm intensity and duration, the extent of pore water buildup is also influenced by soil moisture conditions prior to the storm. Wetter antecedent conditions (e.g., in midwinter) promote more rapid pore pressure response during storms compared to drier conditions such as when fall rains begin. (Sidle and Bogaard 2016.)

3.1.2. The role of forest vegetation in controlling water erosion

Maintaining forest vegetation is an effective, economical and long-lasting approach to mitigating soil erosion and stream sediment loading in forest environments. Plant sizes are taller, canopy density is greater, litter floor is thicker and root systems are deeper in forests than in any other type of vegetation cover. Thus, forests resist erosion and sediment movement much more than other vegetation types (Chang 2012). This resistance is a key reason forests are capable of producing such high-quality water.

Water erosion of soil is initiated when soil particles or soil masses are detached from the soil matrix or underlying surface by some combination of precipitation, runoff energy and gravity. Forest vegetation attenuates soil detachment and transport, at scales ranging from individual soil particles to mass movement of large volumes of soil and rock, via several different mechanisms:

- Interception of rainfall by tree canopies above the ground, which reduces the velocity and energy of raindrops and also the amount of precipitation that reaches the ground.
- The ability of litter, woody debris and ground-level vegetation to reduce raindrop and overland flow energy by shielding the soil and inhibiting runoff movement.
- Root systems and organic matter that increase the cohesive and frictional components of soil shear strength, which contributes to soil stability.
- Transpiration and evaporation (evapotranspiration) of water by trees and other vegetation, which reduces soil moisture content.
- Buttressing or soil-arching action between tree trunks, which counteracts downslope shear forces.

Most rain falling into a forest canopy is intercepted by tree foliage. In smaller storms, nearly all rainfall may evaporate off of the foliage and never reach the ground, especially in dense coastal old-growth forests, but the percentage that evaporates decreases as storm intensity and duration increase (Moore and Wondzell 2005). The degree to which a forest canopy reduces raindrop energy by intercepting drops depends on canopy density, canopy height and tree species. Canopy heights (from ground level to the lowest tree branches) of less than 20 meters significantly decrease raindrop speed and impact energy. Conifer forest canopies intercept more rainfall than deciduous forests. Forest vegetation of any height also helps attenuate wind and increases in raindrop impact energy caused by wind-driven drops striking the soil at an angle. (Williams 2016; Chang 2012.) Canopies that are close to or in ground contact act as shields and essentially eliminate raindrop energy. Litter in the form of leaves, needles, cones and small branches that drop from forest canopies of any height increases ground surface roughness and slows runoff velocity, thereby reducing soil erosion.

Large roots from woody vegetation extend down through the soil surface horizon and anchor the soil mantle to the substrate. In conjunction with these larger taproots and lateral roots, fine roots, fungal mycelia, and decomposed organic matter help form anchored aggregates of surface soils centered around individual trees. The strong binding effects of this dense and interwoven soil–root system stabilize the forest soil mantle.

Any forest management action that reduces canopy coverage and disturbs the forest floor and soil has the potential to generate additional erosion and sediment production. The increased sediment yield resulting from a forest activity depends on the degree of forest and soil disturbance, location and proportion of the watershed affected, watershed characteristics (e.g. slope, soil type, ecological factors), weather patterns and climate.

3.1.3. Forest harvesting, erosion and sediment production

In actively managed forests, logged hillslopes are the largest land surface area subject to potential disturbance. Under modern forest practices, the size of harvested compartments (clearcuts) is restricted to smaller sizes than in the past. These general harvest areas (outside of haul roads and major skid trails) usually have patches of compacted soils interspersed with areas more similar to undisturbed forest floor. Runoff typically builds slowly in general harvest areas, even under heavy rainfall, usually starting on the more disturbed patches of the hillslope. But channelized flow tends not to develop in general harvest areas due to the high spatial variability in soil infiltration capacity, and presence of remaining vegetation and loose material on the soil surface. This patchy nature of runoff generation usually limits the ability of runoff in general harvest areas to mobilize large amounts of sediment. After harvesting, disturbed soils can recover some of their infiltration capacity over time (Croke and Hairsine 2006).

There are exceptions to these general findings regarding general harvest areas, especially when forests are harvested in steeper terrain. Removal of trees has consistently been shown to reduce the stability of steep slopes and increase the risk of landslides and mass movement (Goetz et al. 2015; Imaizumi and Sidle 2012; May 2002; Jakob 2000; Montgomery et al. 2000). More specifically, many studies have shown that from about two to 15–20 years after harvesting on steep slopes, the rate of landsliding is about two to 10 times higher than prior to harvest (Sidle and Bogaard 2016) and that this increase is strongly linked to the loss of root reinforcement and cohesion in forest soils after the trees are removed and as the roots decompose (Sakals and Sidle 2004; Roering et al. 2003; Guthrie 2002; Schmidt et al. 2001). Intact forests also contribute to slope stability by attenuating rainfall and soil moisture (Preti 2013) although Sidle and Bogaard (2016) argue that in temperate forests, root reinforcement is usually more important for slope stability than transpiration or canopy interception.

These findings are particularly relevant in the Oregon Coast Range. In this region, landsliding is a major geomorphic process by which sediment is delivered to headwater streams, "priming" the landscape for subsequent episodic movement of fine-grained sediment downstream during large storms and associated floods (May and Gresswell 2003). Increased landslide risk associated with forest harvesting can be reduced by partial cutting of the stand and retention of understory vegetation (e.g. Dhakal and Sidle 2003; Sakals and Sidle 2004; Turner et al. 2010).

Findings linking forestry activities on steep slopes with increased occurrence of landslides are usually based on landslide inventories comparing logged and unlogged areas. Such inventories are often compiled primarily through air photo interpretation, a method which can be subject to "detection bias" — the difficulty of detecting smaller slides under the canopies of intact forests (Robison et al. 1999). Rigorous studies often attempt to correct for this potential bias by augmenting air photos with subsampling, ground truthing or some type of correction factor (e.g., Turner et al. 2010; Miller and Burnett 2007). The use of Light Detection and Ranging (LiDAR) remote sensing techniques also shows promise for reducing detection bias in landslide delineation and inventory (Guzzetti et al. 2012; Jaboyedoff et al. 2012).

Relationships between active forest management and sediment production have been extensively researched and are discussed in more detail in Chapter 5.

3.1.4. Site preparation and sediment production

Under the Oregon Forest Practices Act, industrial timberlands in the state must be replanted to trees within 24 months after clearcut harvests. Prior to replanting, sites are prepared to reduce vegetation that competes with tree seedlings, habitat for animals that damage seedlings and wildfire risk. Site preparation also creates spots for planting (Fitzgerald 2008). Site preparation can involve the use of herbicides, mechanized equipment, fire or some combination of these methods. In general, any site-preparation activities that contribute to an increase in bare mineral soil, soil compaction or soil mixing have the potential to increase sediment production. See Chapter 5 for a more detailed discussion regarding these interactions.

Industrial timberlands in western Oregon are typically treated with herbicides prior to replanting. Neary et al. (2000) maintain that in general, herbicide use ranks behind both fire and mechanized equipment in severity of impact on sediment production. But herbicide use in western Oregon forestry continues to spark controversy, especially over the potential for it to drift into drinking water sources or populated areas when applied via aerial spraying (e.g. Burns 2019; Perkowski 2018; Swanson 2017). The Forest Practices Act stipulates that herbicides must be prepared for use at least 100 feet from streams that bear fish or are drinking water sources. Aerial applicators must closely monitor weather patterns and only spray when risk of drift will be minimized. They must also spray at least 60 feet from waterways and bodies of standing water larger than a quarter-acre. Any detectable concentrations of herbicides in waterways are usually short-term. Herbicide use in forestry is discussed in more detail in Chapter 6.

3.1.5. Forest roads, erosion and sediment production

Sediment input into streams from forest roads has long been of concern, and forest roads continue to be recognized as the major source of erosion in watersheds (Croke and Hairsine 2006). As Neary et al. (2009) put it, "…the study of nonpoint source pollution from forestry activities has largely been a study of runoff and erosion from bare soil areas created for roads, landings, skid trails, fire breaks, and also bare soils created by site preparation fires. In all forested areas of the United States (except for flat coastal plain areas), roads, landings, and skid trails have been repeatedly implicated as the primary source of sediment from silvicultural operations" (p. 2275).

A watershed-level network of forest roads often contains a mosaic of older and newer roads designed to different standards, sometimes for different purposes, and crossing terrain of differing sensitivities to erosion and mass wasting. The particular pattern and hydrologic connectivity of this mosaic of road segments has implications for how it will interact with the forest watershed, streams and other downstream water uses (Endicott 2008). Impacts of roads range from chronic and long-term contributions of fine sediment into streams to catastrophic mass failures of road cuts and fills during large storms (Beschta 1978; Wemple et al. 2001; Sidle and Ochiai 2006). Megahan and King (2004) concluded that roads affect landslide creation more than any other forest management activity. Problems with drainage and transport of water — especially during heavy rainfall and floods — are primary reasons roads fail.

Roads can also alter channel morphology directly or modify channel flow and extend the drainage network into previously unchanneled parts of the hillslope. The magnitude and longevity of chronic effects of forest roads on suspended sediment in streams depends on many site-specific factors. These factors include traffic, geology, road grade, road connectivity to the stream and sediment availability for transport (Grant and Wolff 1991; Benda and Dunne 1997; Hassan et al. 2005), and also road age, construction practices, maintenance practices, climate and storm history. Volume, weight and the timing of traffic (i.e. during dry or wet weather) also affect the amount of sediment produced.

In recent decades, management practices in road location, design, construction, maintenance and use have markedly improved (Gucinski et al. 2001). Most changes have focused on reducing hydrologic connectivity between roads and waterways. But few studies have quantified improvements in lowering mass erosion rates, and forest roads and their effects on sediment production and water quality remain controversial issues. “Legacy” roads — forest roads that were planned and built before current road-building standards — are also problematic and controversial.

3.1.6. Forestry and sediment production: information gaps

Anderson and Lockaby (2011) found information gaps in science knowledge regarding active forest management and stream sediment. One need is for longer-term studies that can better account for climatic variability and address the effectiveness of current and improved forest practices over time. They note that funding for long-term and paired watershed studies has declined, although the Alsea Watershed Study has been reinitiated on a more limited basis (e.g. Hatten et al. 2018) and the Hinkle Creek Watershed Study was initiated in 2011. They observe that major storms are often a significant driver of sediment movement, and that whether or not one or more major storm occurs during the duration of study can significantly affect results of studies that span only a few years. Knowledge regarding mechanisms of sediment production and the cumulative effects of forest management in larger watersheds is limited. This is due in part to the variability of forestry activities (e.g., roads, harvesting and site preparation) and temporal range of their impacts on stream sediment. Some actions have an immediate effect and others take years to manifest. Research is also needed into how much of the sediment mobilized from silviculture or forest roads is then actually delivered to streams.

Anderson and Lockaby (2011) also note that while forestry Best Management Practices (BMPs) overall have clearly resulted in significant reductions in impacts to water quality, studies that sort out the effectiveness of individual practices are still quite limited. This point is echoed by Edwards et al. (2016). In addition, even when a particular BMP is known to be effective, the exact mechanism for its effectiveness may still be unclear. For example, vegetated buffers along streams have been clearly shown to reduce sediment, but is this due to reducing or intercepting overland flow, reducing bank and channel scouring, or a combination? Similarly, there are significant knowledge gaps regarding the effectiveness of different buffer widths, and the effects of thinning or partial harvest within buffer zones.

3.2. Forest management and water production

Relationships between forest cover and type, forest management, and the quantity and timing of water produced by forested watersheds have been studied for at least 100 years (e.g., Bates and Henry 1928; Griffin 1918). Understanding of these relationships has been greatly enhanced by long-term, paired watershed studies (Stednick 1996, Stednick 2008, Stednick and Troendle 2016). But significant knowledge gaps remain regarding how harvesting may affect water production in larger watersheds and mechanisms that influence the ability of watersheds to store water (McDonnell et al. 2018).

3.2.1. Precipitation, infiltration and watershed storage

Water in stream channels comes from at least one of the following:

- Precipitation intercepted by stream channels.
- Overland flow (surface runoff).

- Interflow (subsurface runoff).
- Baseflow (groundwater runoff).

Precipitation in forests is reduced by canopy and litter interception and to a much lesser degree by wetting of the soil surface. *Effective rainfall* – the amount that reaches the mineral soil — ranges from 70% to 80% of gross rainfall in forested areas. Water enters soil by *infiltration*, a combination of capillary attraction, gravitation, and pressure from water ponding at the surface. The rate of infiltration is initially high and then declines as soil spaces fill with water. The process of water draining to deeper layers is called *percolation*. *Macropores* are voids in the soil through which precipitation percolates, mostly tubular channels created by root mortality or activity by insects, worms or burrowing animals, but also structural cracks or fissures. Macropores are the reason intact forest soils display much higher vertical hydraulic conductivity than those obtained from sieved samples of the same soil (Williams 2016).

Surface conditions, such as vegetation type, land uses, roughness, crusting, cracking, slope, water repellency resulting from fire, and chemicals, have a significant impact on surface ponding, overland flow velocity and the ability of water to infiltrate soil. Below-ground conditions that affect soil water-holding capacity and water movement include soil texture, structure, organic matter content, depth, compaction, water content, groundwater table and root systems. Forested watersheds are generally characterized by deep, loose soils; thick, loosely compacted duff layers on the forest floor; complex root systems; large canopies and high capacities for water infiltration. For a given soil type, water infiltration in a forest can be many times greater than over bare mineral soil. As a result of these factors, infiltration-excess overland flow is rare on undisturbed temperate forests (Neary et al. 2009; Williams 2016), including those in the Pacific Northwest.

Watershed storage is water retained within a watershed after collection from precipitation and before discharge out of the watershed as streamflow. Watershed storage consists primarily of soil moisture, canopy and litter interception, snowpack, ponds and wetlands, shallow aquifers, storage in streambanks, channels and fractured bedrock, and in vegetation during transpiration. Stored water can remain in a watershed for years or even decades (McDonnell et al. 2018). Water storage is the key function of forested watersheds (Black 1997).

3.2.2. Runoff and streamflow

Runoff is precipitation (rain or melted snow) running across the land surface or through the soil to nearby stream channels. It occurs when rainfall or snowmelt is greater than soil infiltration rate, or exceeds soil-infiltration or percolation capacity. The soil surface does not need to be saturated for overland flow to occur. Infiltrated water can become surface runoff again as it flows laterally and downslope or to stream channels as subsurface runoff. Channel rainfall, surface runoff, and subsurface runoff combined are *direct runoff*, a direct response of streamflow to storm precipitation over a relatively short time frame. Water flowing in streams during periods of no rainfall (base flow) comes from groundwater. The sum of direct runoff and base flow is total streamflow.

A *hydrograph* graphically illustrates streamflow discharge or stage over a particular time period, such as a single storm event or a water year. A hydrograph for a storm event typically shows an upward sloping, then level, and then downward sloping line as discharge increases and then declines back to base flow. The *rising limb*, which shows increasing watershed discharge, begins sometime after precipitation starts, and varies with watershed characteristics and storm duration, intensity and distribution. Due to watershed storage, lag time is longer in forests than other watershed cover types. Large watersheds may take days to respond to precipitation. The hydrograph *crest* — the highest concentration of storm runoff, also termed *peak flow* — spans from where the rising limb levels off to where the line begins to decline. The end of the crest indicates the end of direct runoff to the stream. The *recession limb*, showing the draining-off process, represents the contribution of water from watershed storage and is independent of storm characteristics (Chang 2012).

Streamflow discharge varies greatly with watershed. Smaller watersheds tend to be more sensitive to precipitation events, with quicker responses, and sharper rises and declines in their hydrographs than larger watersheds. In general, higher elevation watersheds are cooler and have less evapotranspiration, more precipitation, steeper slopes, and shallower soils. This results in more runoff than in lower-elevation watersheds. Soil infiltration tends to be lower and overland runoff greater and faster in watersheds with steep slopes. Watersheds with shallower slopes typically store more water than those with steep slopes (Chang 2012). However, studies in the California and Oregon coast ranges (Montgomery and Dietrich 2002; Sayama et al. 2011) showed that steeper watersheds in those sites can store as much or more water than — and release it at similar rates to — watersheds with shallower slopes. These findings are attributed primarily to water storage in fractured, permeable bedrock just below the soil layer.

Oregon experiences a Mediterranean climate, resulting in distinct seasonal delivery of precipitation that can be categorized as wet and dry seasons. About 80% of annual precipitation in western Oregon falls between October and March, especially from December to March when there is ample streamflow and virtually no agricultural demand for irrigation water. At higher elevations, much of this precipitation falls as snow, which accumulates through winter then melts during spring. The timing of snowmelt thus plays a major role in shaping annual hydrographs in Oregon.

Hydrographs for most Oregon streams peak in winter and spring, but demand for most water uses peaks during the late summer dry season when flows are lowest (Mucken and Bateman 2017). For water providers trying to meeting late-summer demand, this misalignment poses persistent challenges. These challenges are expected to intensify as a warming climate reduces the proportion of annual precipitation falling as snow and stored as snowpack, increasing winter rainfall that runs off without being stored (Clifton et al. 2018; Mote et al. 2018; Siler et al. 2018).

Streamflow fluctuation is important to water supply and floodplain management and can be used as an indicator for the effectiveness of watershed management conditions.

3.2.3. Forest harvesting and water yield

Forestry activities can affect water production by altering total *annual flow;* and also shorter-term *peak flows* (e.g. during and after storms) and seasonal *low flows* which in Oregon typically occur toward the end of the summer dry season. These subtopics are introduced here and discussed in greater detail in Chapter 4. Many studies in wetter forests have found that forest harvesting increases watershed-level water yield (e.g., Jones and Post 2004). Paired watershed studies indicate that a minimum of 450–500 millimeters of annual precipitation is usually necessary for increases to be apparent. In drier forests, harvest often simply increases soil evaporation or water use by other vegetation (Stednick and Troendle 2016).

Watershed-level water yield increases usually are greatest the year after cutting. Yield decreases as vegetation regrows, eventually returning to preharvest levels.

Water yield increases are attributed to increases in soil moisture due to reductions in evapotranspiration and canopy interception of rain and snow after trees are removed, and vary with harvest intensity, species, amount and timing of precipitation, and soil topographic conditions (e.g. Reid and Lewis 2007). Deep and fine-textured soils can hold more water than shallow and coarse-textured soils and thus have more potential for water yield increase. In soils less than about 1 meter deep, water yield increases are minimal after forest harvest. Harvesting on upper slopes increases water yield less than harvesting on lower slopes or close to stream channels (Chang 2012). Harvesting 20% of the watershed is commonly cited as the minimum necessary to detect an increase in water yield; for 12 studies in the Pacific Northwest this figure averaged 25% (Stednick 1996).

Results of studies on which these generalizations are based vary widely, with some watersheds showing large increases in water yield after harvest and others showing little to none at all. Further complicating this picture are studies indicating that watersheds covered with young, vigorously growing plantations of Douglas-fir significantly reduce summer low flows compared to adjacent unharvested watersheds where cover remains in old-growth forest (Moore et al. 2004; Perry et al. 2017; Segura et al. 2020). Few studies have addressed this issue; effects appear to take around two decades or more after harvesting and replanting to become apparent (Gronsdahl et al. 2019), and relevant long-term data is limited. However, the potential for decreased summer low flows associated with timber plantations is likely to spur additional research, given the critical nature of water supplies during this time of year and the potential for climate change to exacerbate such challenges.

In addition to changes in annual water yield, forest harvesting can also affect the timing of water production from a watershed. Comparing two small, snow-dominated watersheds on the Okanagan Plateau of British Columbia in Canada, Winkler et al. (2017) found only a 5% increase in overall yield after clearcutting of 47% of the logged watershed, but dramatic changes in the timing and magnitude of April-June streamflow, which they said could increase the risk of channel destabilization during the snowmelt season and water shortages early in the irrigation season.

Difficulties in consistently predicting the effects of forest harvest and regeneration on water yield have prompted suggestions that this approach is overly simplistic. As a result, some have called for an expanded research agenda that also encompasses relationships between forest harvesting and processes that affect watershed storage in order to maintain this key ecosystem service (McDonnell et al. 2018; see also McNamara et al. 2011, Sayama et al. 2011). Further, Chang (2012) notes that most studies on harvesting and water yield take place in the upper parts of watersheds, so effects on water quantity changes for downstream water users also warrant further research.

3.2.4. Forest harvesting and peak flows

The effects of forest cover on peak flow frequency, magnitude and timing have been debated since at least the early 20th century, when these issues served as a key rationale for creation of the U.S. National Forest system. Peak flows and flooding can affect drinking water treatment by raising turbidity levels and introducing other pollutants mobilized by floodwaters.

While not definitive, evidence supports the notion that forestry activities can increase peak flows (e.g. Winkler et al. 2017; Zhang and Wei 2014; Schnorbus and Alila 2013; Kuras et al. 2012; Lin and Wei 2008; Moore and Wondzell 2005; Jones 2000; Burton 1997; Jones and Grant 1996). One challenge with this type of research is the difficulty of distinguishing the effects of harvesting from those of roads. Another persistent challenge is that as peak-flow size increases, frequency of occurrence decreases, so the number of observations and resulting statistical power regarding the largest events are usually limited. Also, reviews and summaries (e.g., Stednick and Troendle 2016; NRC 2008; Grant et al. 2008) often find that results are mixed across studies from different areas and with different methods. Some studies show significant increases in peak flows and others show no effects or decreases.

Smaller peak flows generally increase after harvesting, but Sidle and Gomi (2017) caution that "much controversy still persists around the effects of forest removal on large floods," (e.g., Alila et al. 2009; Beschta et al. 2000; Jones and Grant 1996; Kuras et al. 2012; Thomas and Megahan 1998) and that based on physical evidence, attributing increases in major peak flows to forest harvest only is "difficult to justify" (Sidle and Gomi 2017, p.101). Scientific thinking on this complicated topic continues to evolve, and research results can vary widely, depending on topography, climate, site conditions, land use history, scale, study design, analysis methods and other variables. As the National Research Council (2008) notes, scientists are confident regarding general hydrological responses to forest harvesting, but precise prediction of effects in areas that have not received intensive study can be problematic.

Most Oregon streams and rivers peak during winter and spring, and decline to their lowest levels in late summer. Overlain on this seasonal pattern are numerous hydrograph peaks from individual precipitation events. Rain-on-snow events in particular can produce large spikes in hydrographs (Marks et al. 1998). Thus, peak flows are assessed by looking at both their frequency and magnitude. Chang (2012) provides broad conclusions from research on the effects of forestry on peak flows and flooding, including:

1. Forestry activities such as road construction and site preparation that cause soil compaction are more likely to affect flood generation than forest harvesting.
2. Forests can attenuate peak flows for storms of short duration and lower intensity, but cannot prevent floods produced from storms of high intensity and long duration over a large area.

The consensus has been that impacts of forest harvest on peak flows are more noticeable for smaller, shorter storms. When precipitation amount, intensity and duration increase, the relative influence of human activities on runoff volume declines. Soils become saturated, and flows overwhelm any incremental increase attributable to forest harvesting. Human-caused increases in flow volume and peak are less evident downstream due to cumulative effects from other tributaries, decreasing percentages of treated areas as watershed area increases downstream, and attenuating effects of channel storage (Chang 2012, Buttle 2011).

Grant et al. (2008) reviewed the effects of harvesting on peak flows in western Oregon and Washington. They found wide variability in research results, and that assessing the effects of modern forest practices was problematic. They note that peak flows are also affected by overall basin condition; age and pattern of forest stands; the location, age and extent of road networks; and the extent of riparian buffers. The review was complicated by challenges in distinguishing the effects of harvesting from effects of forest roads. General conclusions from the review include:

1. The largest peak flow increases reported were for small storms with recurrence interval much less than 1 year.
2. Increase in peak flow generally decreases with time after harvest.

3. The largest increases occur in clearcut areas.
4. Watersheds in rain-dominated elevations are less sensitive to peak flow changes than those in the transient snow zone. (A lack of sufficient data precluded assessment of harvesting in the snow zone).

The review found studies in larger basins to be limited and complicated by other land uses and factors that affect peak flows. However, reviewers concluded that the magnitude of any peak flow increase in response to forest management diminishes with increasing basin area. Grant et al. (2008) list, in order of potential likelihood of increasing peak flows:

1. High road density.
2. High road connectivity.
3. Fast watershed drainage efficiency.
4. Large harvest patch size.
5. Lack of riparian buffers.

The magnitude of peak flow increases generally increase with percentage of the basin harvested (Buttle 2011). Stednick and Troendle (2016) indicate that peak flow increases seem to occur less frequently under contemporary forest practices, which they surmise is due to generally smaller harvest patch sizes and proportion of the watershed harvested, reduced road lengths and the presence of streamside vegetation buffers.

Research on peak flows in transient snow and snow zones has increased over the past decade and is relevant for Oregon, where many forests receive a large percentage of their annual precipitation as snow. Studies indicate that forest harvest may increase peak flows in these zones to a greater degree than in rain-dominated zones, especially during rain-on-snow events (e.g. Marks et al. 1998; Jones and Perkins 2010). Mechanisms for these increases may include greater snow accumulation and higher wind and advective rain energy available to melt snow in open areas than under forest canopies.

More recently, researchers have debated experimental methods and relevant parameters for assessing the effects of forest harvest on flooding in snow-dominated systems. Green and Alila (2012) argue for a “paradigm shift” from generally accepted methods of comparing floods by equal meteorology or storm input (chronological pairing) to a flood frequency distribution framework (frequency pairing). They maintain that chronological pairing approaches in paired watersheds have yielded inaccurate results that underestimate forestry effects on large flood frequency, and that frequency pairing approaches are more appropriate. Green and Alila (2012) and related work (Kuraś et al. 2012, Schnorbus and Alila 2012) in a low elevation, snow-dominated system in British Columbia found that forest harvesting has substantially increased the frequency of the largest floods. These findings are attributed to increased net radiation associated with conversion from longwave-dominated (infrared) snowmelt beneath the canopy to shortwave-dominated (visible and ultraviolet light) snowmelt in harvested areas, amplified or mitigated by basin characteristics such as aspect distribution, elevation range, slope gradient, amount of alpine area, canopy closure and drainage density.

Alila and his colleagues acknowledge that their results run counter to prevailing wisdom in hydrological science – i.e., that the effect of forest harvesting must always decrease with an increase in flood event size, which is still being taught in textbooks today, including Chang (2012) cited above. Their work spurred debate regarding the use of chronological pairing and frequency pairing approaches (Alila and Green 2014a; Alila and

Green 2014b; Bathhurst 2014; Birkinshaw 2014). Despite various critiques regarding the most appropriate research questions and methods, commenters generally suggested that both approaches provide meaningful information.

The effect of roads on peak streamflow is generally assumed to be strongly related to watershed size, road density, and the roads' degree of hydrologic connectivity. Roads on steep hillsides not only contribute overland runoff from compacted areas, but also intercept subsurface flow along cutslopes, especially where cutslopes intersect with bedrock (Sidle and Gomi 2017). Forest roads can contribute significantly to increases in peak flows, sometimes at levels equal to increases attributed to forest harvest (La Marche and Lettenmaier 2001). In large watersheds, roads usually constitute a smaller proportion of the land area and have relatively little effect on peak flow (Gucinski et al. 2001) but this would depend on road density in the watershed.

At a study site in the H.J. Andrews Experimental Forest, Burt et al. (2015) found that large contrasts between El Nino and La Nina climate patterns were a stronger driver of variability in streamflow response than differences in forest cover. Safeeq et al. (2015) concluded that over time, snowpack changes related to climate warming are likely to result in large increases in peak flow magnitudes in areas such as the Cascades and Blue Mountains. These and similar findings suggest that any effects that forestry activities have on peak flows may intertwine with climate in increasingly complex ways. At the same time if, as expected, the frequency and magnitude of floods in Oregon increase under climate change, public and agency interest in mitigating anthropogenic factors that contribute to peak flows may intensify.

3.2.5. Forest management and low flows

Active forest management also has the potential to affect late summer low flows in streams and rivers. This is particularly relevant in Oregon, where there is essentially no precipitation for about three summer months each year. Thus, late summer can be a challenging time for water providers.

Definitions of low flows vary, from point-in-time flow rates, to number of days below a certain threshold, to recurrence intervals such as 7-, 10- or 30-day average low flow. Defining low flow as a percentage of change could be misleading, because a small change in low flow volume could be expressed as a large percentage (Stednick and Troendle 2016).

Baseline low flows in unmodified landscapes are controlled by natural factors such as geology, soils and topography (Tague and Grant 2004). Flows may be modified by changes in land use and climate. Most studies on forest management and low flows have focused on effects from just after harvest through re-establishment of the new stand. There is consensus that low flows usually increase in the first years after forest harvesting (Buttle 2011). Most studies show that removal of forest vegetation increases low flows as a result of reduced evapotranspiration, which increases soil moisture content (e.g. Stednick 2008; Surfleet and Skaugset 2013). Flows generally decline toward preharvest levels within a few years as transpiration rises in the regenerating stand. But a number of studies have also reported no significant change in low flows after harvest (e.g. Lin and Wei 2008).

In the snow zone, low flows typically occur from late summer through the winter until spring snowmelt. Low flows are a normal part of the yearly water cycle. Low flows are maintained in the dry season through the release of water from groundwater storage and surface water discharge from lakes, wetlands and flow from channel banks (Pike and Sherer 2003). Stednick and Troendle (2016) maintain that because current forest practices exclude many riparian areas from harvest, flow increases may not be as common today, and any such increases appear to return to preharvest conditions within a few years.

There is evidence that forest practices may decrease low flows under some conditions. In a study in coastal Oregon, Harr et al. (1982) found reduced low flows after harvesting and hypothesized that reduced fog interception and canopy drip could explain these results. Jones (2000) found similar results and suggested the same causal mechanism in 2 out of 10 basins examined. Hicks et al. (1991) identified decreases in low flows that could be attributed to changes in riparian vegetation from conifer to deciduous species; with the latter transpiring relatively more water per unit of leaf area.

Some recent studies have focused on how regenerating forests affect summer low flows later into the rotation, after the new stand is fully re-established. In the Oregon Coast Range, Segura et al. (2020) compared responses of daily streamflow in a harvested mature/old forest in 1966; 43- to 53-year-old and 48- to 58-year-old industrial plantation forests in 2006–2009; and the same plantation forests in 2010 and 2014 after harvesting. They found that daily streamflow from a 40- to 53-year-old Douglas-fir plantation was 25% lower on average, and 50% lower during summer, relative to the mature/old forest, and that these deficits lasted at least six months of each year. Similarly, in the western Oregon Cascades, Perry and Jones (2017) showed that summer low flows were 50% lower in basins with 34- to 43-year-old plantations of Douglas-fir than in basins with 150- to 500-year-old mixed conifer forests. In three small watersheds in southern interior British Columbia, Gronsdahl et al. (2019) found that summer flows were reduced starting about 20 years after the onset of forest harvesting. These investigators all attributed persistent streamflow deficits after logging to high evapotranspiration from rapidly regenerating vegetation, including planted commercial timber species.

In light of these findings, water providers would benefit from a better understanding of how intensively managed forests and expected warmer, drier conditions in the future may influence summer low flows. In addition to potential water supply issues, summer low flows in streams are also associated with reduced turnover and mixing in the water column, and with increased potential for harmful algal blooms in receiving lakes and reservoirs. This issue is discussed in more detail below.

3.3. Forest chemicals, nutrients and water quality

A variety of chemicals are used in forestry. Fertilizers are often applied in timber plantations to enhance tree growth. Pesticides are used to control unwanted organisms, including fungi, rodents, insects and plants. Herbicides are widely used after harvest to discourage colonization of clearcuts by deciduous species until newly planted conifers are established. Herbicides may also be applied near forest roads to control weeds or vegetation encroachment. Fungicides and insecticides may be used locally to control for fungi or insects that attack trees.

Some of these chemicals may pose a human health hazard if drinking water sources are contaminated during or after chemical applications. During application, chemicals may drift into waterways or other nontarget areas. After application, chemicals or chemical residues may enter surface water or groundwaters through runoff and leaching (USDA-FS 2012). Plant nutrients, minerals, organic chemicals, fertilizer, and pesticides can attach to soil particles and be carried into streams with sediment (Chang 2012). Chemicals applied to roads can also enter streams by various pathways. The effects of these chemicals on water quality depend on how much chemical is applied, the distance of the road from a stream, and characteristics of weather and runoff events that move chemicals and sediments (Gucinski et al. 2001). Forest harvesting machinery requires petroleum fuel and lubricants, which can leak or spill and wind up in waterways.

While the use of chemicals in forestry is usually far lower than in other forms of agriculture, the risks of contamination of water bodies by silvicultural chemicals are well-recognized. Research indicates these risks are usually low, provided that the chemical is carefully applied according to manufacturer directions (by properly licensed professional applicators in some cases) and that modern best management practices are followed. However, the risks that forestry chemicals pose to human health are a persistent concern. Moreover, there are knowledge gaps regarding the persistence and long-term fate of chemicals after they are applied, and a lack of consensus in some quarters regarding the toxicity of certain chemicals (e.g., glyphosate) and long-term health effects in humans.

Aerial spraying of herbicides is a common practice in western Oregon industrial forests, and can be particularly contentious. (See, e.g., Bernstein et al. 2013; Glucklich 2018). Potential impacts on drinking water have led to efforts to eliminate aerial spraying through county-level ballot initiatives. This was successful in Lincoln County. The risks that such activities could degrade water quality in small, non-fish-bearing streams, and potential impacts on drinking water, were among the factors cited in NOAA-EPA's disapproval of Oregon's Coastal Non-Point Pollution Control Program (NOAA 2015).

Forest chemicals are covered in greater detail in Chapter 6.

3.3.1. Nitrogen and other forest nutrients in drinking water

Nitrogen (N) is essential for all living things and is a key nutrient for trees and other plants. But excess N can also impair water quality and aquatic ecosystems, and is the most common water pollutant in the U.S. Nitrogen occurs naturally in soil in organic forms from decaying plant and animal residues, and also in inorganic forms derived from minerals. In the soil, bacteria convert N to nitrate, which is desirable because most N used by plants is absorbed as nitrate. But nitrate is also highly leachable and easily carried by water through the soil profile. In wet climates, dissolved nitrate often percolates below the plant root zone and travels into surface waters and groundwater.

Because of its importance as both a plant nutrient and pollutant, N dynamics after forest harvest (and forest soil N processes in general) have received extensive study. With some exceptions (e.g. Binkley et al. 2004; Binkley et al. 1999) research regarding N dynamics in forests tends to focus on management effects such as harvesting, site preparation and fertilization on the productivity and sustainability of forest soils, rather than potential effects on drinking water. Temperate conifer forests usually conserve N and other nutrients. Soil N and N leaching often increase (usually temporarily) after timber harvesting as a result of reduced uptake from vegetation, or when N is released from decomposing slash or other plant material (e.g., Mupepele and Dormann 2016). Nitrogen export also often increases after wildfires (e.g., Rhoades et al. 2011; Smith et al. 2011) or prescribed fires. Nitrate is often a major portion of the total N exported from forests to surface waters. Processes (e.g., denitrification) in riparian and wetland areas and in streams can remove nitrate, but the significance of these processes in regulating nitrate flux varies widely. This variation suggests that some watersheds with increased N inputs (e.g., fertilization) will show increased nitrate-nitrogen outputs, while others have buffering capacity within soils, riparian areas and stream channels to mitigate such a response (Stednick 2008).

To track its various sources and fates, total dissolved nitrogen in water is often broken out into total organic nitrogen and total inorganic nitrogen. Undisturbed, mature stands may have large stores of organic N in the soil, forest floor litter layers, and old trees, and may utilize less N than vigorously growing younger stands that have lower ecosystem N stores after removal of slash from prior harvest. Forests and tree plantations in the Oregon Cascades and Coast Ranges established after a previous harvest and site preparation are often N-limited, with trees and other vegetation taking up all available N.

Vitousek and Reiners' (1975) model of N dynamics after forest harvest suggested that there is usually an initial flush of N export (because N uptake by vegetation is interrupted) that declines a few years after a new stand is initiated, and then N often becomes limiting again as the young trees grow. Leaching of N after harvest is often observed in temperate conifer forests (Mupepele and Dormann 2016; Jerabkova et al. 2011; Stednick 2008; Binkley et al. 2004; Antos et al. 2003; Feller et al. 2000; Martin and Harr 1989; Brown et al. 1973) with most studies finding that nitrate export declines to preharvest levels within 5–7 years or less, but confounding factors and exceptions are fairly common (Binkley et al. 2004). Variables that can affect the results of different studies include soil conditions (especially initial N availability) and land use history prior to harvest, site preparation methods and length of time after harvest, sampling strategy, weather and climate, topography, hydrology and other factors.

Recent research illustrates the complexity of this topic. At their sites in southwestern Canada, Grand et al. (2014) found overall moderate increases in N, but a dramatic increase in N variability after harvest, with some sites showing extreme inorganic N values. Consistent with studies of local drainage water chemistry, Grand et al. (2014) concluded that conifer forests export significant N after harvesting, but that leaching would likely vary significantly from plot to plot. They suggest that this small- to medium-scale heterogeneity in N export has implications for nutrient leaching potential as well as researchers' ability to detect and predict harvest-induced changes.

In Oregon's west central Cascades, Cairns and Lajtha (2005) found that younger watersheds with stands 10 years or more in age still lost significantly more N than watersheds with older forests. However, building on this work, Cairns et al. (2009) found that higher N concentrations in streams draining younger stands did not correlate well with N concentrations in soil solutions from those stands that were tested by lysimeter. They surmised that the differences identified in their 2005 study may have been a result of in-stream processing (nitrification) of N, in combination with processes in the dynamic riparian vegetation zone near the streams, and also perhaps the presence of minor amounts of N-fixing red alder, which has been shown to be a significant contributor to N exports in many western Oregon watersheds (Greathouse et al. 2014; Wise and Johnson 2011; Compton et al. 2003). If alder increases after harvest, this adds to the pool of N available for export, especially if alder is a component of the riparian vegetation (Pike et al. 2010; Stednick 2008).

Nitrogen export can increase seasonally with the onset of wet weather in the fall (e.g. Vanderbilt et al. 2003) or during periods of snowmelt. Swank (2000) indicates that knowledge gaps remain regarding nutrient concentration changes associated with storm runoff events, and that such information is most important where drinking water supplies are derived from forested headwaters with rapid streamflow responses to precipitation, e.g., watersheds with shallow soils, steep slopes, intense rainfall and rapid snowmelt.

While Oregon forestlands are some of the most productive in the world, additions of N can often promote even more vigorous tree growth. Also, intensive forest management can reduce N stores in forest soil. For these reasons, N fertilizer is commonly applied on

PNW commercial timberlands, usually by helicopter (Hanley et al. 2006). Although the amount applied is a fraction of that used in conventional agriculture, some 125,000 acres of Oregon timberland are fertilized annually. Nitrogen from forest fertilization can be a significant contributor to elevated N levels in some stream reaches in Oregon's western Cascades and Coast Ranges (Anderson 2002).

Phosphorus is less frequently applied to commercial timberlands, usually as a smaller component of N-based fertilizer blends. While the focus has primarily been on N, there is evidence that phosphorus may be limiting in a significant acreage of PNW Douglas fir forests. This suggests that adding it to these stands may be beneficial from a timber management perspective (Mainwaring et al. 2014). Phosphorus in drinking water is generally not considered to pose direct human health risks (Scatena 2000) but excess phosphorus in water bodies can contribute to harmful algal blooms, discussed below.

Excess nitrates in drinking water can pose human health risks, primarily for infants and nursing mothers, and are regulated by the EPA. Available evidence indicates that nitrate accumulations attributable to forestry rarely approach drinking water standards and that when they do are usually short-term (Bisson et al. 1992; Binkley and Brown 1993; Anderson 2002; Binkley et al. 2004). Binkley et al. (1999) cite the effects of repeated fertilization in short-rotation plantations as a major knowledge gap.

If properly implemented, Best Management Practices to minimize nutrient flushing after forestry activities and the potential for fertilizers to get into waterways are considered to be effective (Cristan et al. 2016; Stednick 2008). Any potential for dissolved nutrients in runoff from actively managed forests to impact drinking water may be higher where the source watershed is relatively small, steep or close to the municipal water intake, contains significant amounts of commercial timberland, where tree plantations in the watershed are fertilized multiple times or just prior to significant storm events.

3.3.2. Forest nutrients and harmful algal blooms

Certain environmental conditions in freshwater bodies (usually involving excessive nutrients) can cause algae and similar microorganisms to grow explosively, causing algal blooms. Blooms that can harm human health or aquatic ecosystems are termed *harmful algal blooms*. Phosphorus and nitrogen both contribute to harmful algal blooms in freshwater systems. In these systems, naturally occurring cyanobacteria (photosynthetic bacteria formerly called blue-green algae) typically cause the most frequent and severe harmful algal blooms.

Some cyanobacterial harmful algal blooms (termed cyanoHABs by the EPA) can produce potent toxins called cyanotoxins. These cyanotoxins can cause sickness and death in humans, pets and livestock who drink the water or otherwise come in contact with it. CyanoHABs can also create hypoxic (low oxygen) conditions in water bodies that can kill fish and other wildlife. CyanoHABs are a growing concern in the United States and worldwide as a result of their potential to broadly impact aquatic ecosystems, drinking water supplies, property and other economic values, and water-based recreational activities (USEPA 2019a).

A range of environmental factors can contribute to cyanoHABs. CyanoHABs are usually initiated by an excess of nutrients (especially phosphorus and nitrogen), compounded by warm, stagnant water, plentiful sunlight and sometimes invasive fish species. Sources of nutrient pollution include wastewater treatment plants, septic systems, fertilizers, agricultural runoff, urban and forestry runoff, and soil erosion. The exact combination of these factors that result in an individual bloom depends on conditions at that particular waterbody. Identifying the specific causes of a cyanoHAB usually requires detailed environmental analysis (Oregon DEQ 2019).

There have been cyanoHABs in a number of Oregon lakes, reservoirs and rivers, usually in late summer when inflows, water levels, and vertical mixing in the water column are lowest. Depending on local conditions the cyanoHABs vary in appearance from green, blue-green to reddish brown colored in the form of mats, foam, slicks or scum. If cyanotoxins over the USEPA national 10-day Health Advisory levels occur in tap water, people are at risk of health impacts including upset stomach, vomiting and diarrhea, and liver and kidney damage. Oregon has several documented cases of dogs dying and humans becoming ill from exposure to cyanotoxins from cyanoHABs. Conventional water treatment can usually remove cyanobacterial cells and low levels of cyanotoxins. However, providing safe drinking water can challenge providers during a severe bloom event, when drinking water sources contain high levels of these pollutants.

Conditions that cause cyanobacteria to produce cyanotoxins are complex and not fully understood. Some species that can produce toxins may not do so under all conditions. Both toxic and nontoxic varieties of most of the common toxin-producing cyanobacteria exist, and it is not possible to determine toxicity by how the bacteria look. Even when toxin-producing cyanobacteria are present, they may not always produce toxins. To further complicate matters, some species can produce multiple types and variants of cyanotoxins. Molecular testing can establish if the cyanobacteria carry the toxin-producing gene but quantitative cyanotoxin analysis is necessary to determine if they are actually producing the toxin (USEPA 2019a).

Conditions that favor longer and more severe cyanoHABs, such as warmer temperatures and increased nutrient inputs into waterways, are increasing. Reducing excess nitrogen and phosphorus in drinking water sources is important for long-term mitigation of the risks from cyanoHABs. As of June 2019, there were no federal regulatory guidelines for cyanobacteria or their toxins in drinking water or recreational waters. However, the EPA published drinking water health advisories with recommended 10-day limits for children and adults for the toxins microcystins and cylindrospermopsin in June 2015. In 2016, the cyanotoxins anatoxin-a, cylindrospermpsin, microcystins and saxitoxin were listed on the EPA Contaminant Candidate List, requiring monitoring for them between 2018 and 2020 using analytical methods developed by EPA and consensus organizations (USEPA 2019a, b).

The Oregon Health Authority (OHA) is responsible for posting warnings and educating the public about cyanoHABs. Once a bloom is identified, the Oregon Department of Environmental Quality (DEQ) is responsible for investigating the causes, identifying pollution sources and producing a pollution reduction plan. The DEQ and the OHA coordinate the handling and analysis of harmful algal bloom water samples (Oregon DEQ 2019). The DEQ also focuses on addressing nutrient, sediment and other HAB-related load allocations via its Total Maximum Daily Load process and both the Oregon Agricultural Water Quality Management Area Plans and the Oregon Forest Practices Act; which the Oregon Department of Agriculture and the Oregon Department of Forestry use to meet water quality standards. (Schaedel 2011.)

At the statewide level, forestry-related nutrient runoff that contributes to cyanoHABs in Oregon probably ranks well below agricultural and urban runoff in significance. But contributions from forestry activities could be important or even dominant for particular blooms at the local level. CyanoHABs are expected to increase as climate change progresses. With concern about cyanoHABs growing and increased scrutiny from agencies charged with oversight of drinking water, science knowledge will also expand.

This may trigger additional regulatory and agency action to monitor and control harmful algal bloom-related nutrient runoff from all sources, including forestry.

3.4. Natural organic matter and disinfection byproducts

3.4.1. Natural organic matter

Natural organic matter is ubiquitous in drinking water source waters. Defined as nonliving organic molecules found in the environment in soil, sediments and water, natural organic matter is a product of plant and animal tissue decay and plays a pivotal role in the carbon cycle (Nebbioso and Piccolo 2013). Living matter is mostly composed of well-defined molecules such as proteins, nucleic acids, lipids, sugars and cellulose. In contrast, due to interactions with soil and rocks that alter its plant and animal-derived precursors, natural organic matter is mostly composed of molecules of unknown structure. Nevertheless natural organic matter has been extensively researched because of its ecological and geochemical importance and influences on pollutant fate and transport in the environment. Natural organic matter in water includes *particulate organic matter* (POM) and *dissolved organic matter* (DOM), each defined by isolation using filtration, with POM being the fraction caught in the filter and DOM the fraction passing through with the water.

Prior to the early 1970's, treatment of natural organic matter in raw water focused on aesthetic issues such as color. Then, research demonstrated that natural organic matter is a precursor constituent in the formation of hazardous disinfection byproducts. Today, natural organic matter is the raw water constituent that most often influences the design, operation, and performance of water treatment systems. In addition to its role in the formation of disinfection byproducts, natural organic matter can overwhelm activated carbon beds used in water treatment and reduce their ability to remove organic micropollutants. Natural organic matter also contributes significantly to the fouling of membranes in all membrane technologies used in water treatment, and can promote microbial fouling and regrowth in water distribution systems. Expanded understanding of linkages between natural organic matter and disinfection byproducts continues to spur changes in drinking water treatment and regulation (O'Melia 2006).

3.4.2. Disinfection byproducts

Disinfection byproducts are an unintended outcome of using chemical disinfectants to kill harmful pathogens (e.g., cryptosporidium) in drinking source water. Disinfection byproducts form when disinfectants react with natural organic matter (usually decaying plant matter), or with bromide, iodide, or various pollutants. People ingest disinfection byproducts primarily through drinking water, but also via inhalation and skin exposure while bathing and swimming. Documented health risks include bladder cancer, miscarriage, birth defects, liver and kidney damage and respiratory problems. Based on existing research, disinfection byproducts such as trihalomethanes and haloacetic acids are regulated by the EPA and in other countries. Research combining toxicology and chemistry has identified other emerging disinfection byproducts of concern. Disinfection byproducts are produced by four major disinfectants used by water providers (chlorine, chloramines, ozone and chlorine dioxide) and also by UV treatment with postchlorination. Each disinfectant can produce its own suite of disinfection byproducts (Richardson and Postigo 2012).

A key consideration for drinking water providers is identifying sources of and reducing the quantity of natural organic matter that arrives at their raw water intakes. Natural organic matter from forest detritus is a major precursor to disinfection byproducts in drinking water sources (Bhardwaj 2006, Majidzadeh et al. 2019). Thus, forest management activities that influence the quantity and mobility of this source of natural organic matter in source waters can influence the potential for disinfection byproducts to form during water treatment. Natural organic matter and disinfection byproducts are discussed in greater detail in Chapter 7.

3.5. Best management practices

3.5.1. Best management practices: history and overview

Recognition that forestry activities can affect soil and water quality emerged by the early 1900s. Organized research programs into the causes and mechanisms of these effects were initiated in the 1950s, as harvesting increased to accommodate the post-war housing boom, giving greater visibility to forestry activities and awareness of their impacts. Passage of the Clean Water Act in 1972, and additional provisions under the 1987 Clean Water Act reauthorization to address nonpoint source pollution prompted further development, implementation and refinement of forestry procedures to minimize soil and water quality impacts. These methods are termed *best management practices*, or BMPs.

A number of different definitions for forestry BMPs appear in scientific and government agency literature. The most detailed definition, and that from which several others have been derived, may be this one from the U.S. Forest Service *Soil and Water Conservation Handbook*, first published in 1988:

A practice or a combination of practices, that is determined by a State (or designated area-wide planning agency) after problem assessment, examination of alternative practices and appropriate public participation to be the most effective, practical (including technological, economic, and institutional considerations) means of preventing or reducing the amount of pollution generated by nonpoint sources to a level compatible with water quality goals (USDA-FS 1988).

The most common definition is probably this one from the Clean Water Act (40 CFR 130.2[Q]; Clean Water Act: Definitions), used for many years by the EPA and currently found in some archived EPA documents, and still in use by many state forestry agencies:

A practice or combination of practices considered by a State [or authorized Tribe] to be the most effective means (including technological, economic and institutional considerations) of preventing or reducing the amount of pollution by nonpoint sources to a level compatible with water quality goals.

As of 2019, this is the definition used in the Clean Water Act and by EPA, and appears to have been in use since at least 2011:

Methods, measures or practices selected by an agency to meet its nonpoint source control needs. BMPs include but are not limited to structural and nonstructural controls and operation and maintenance procedures. BMPs can be applied before, during and after pollution-producing activities to reduce or eliminate the introduction of pollutants into receiving waters.

Other definitions include:

Practical control measures (including technological, economic and institutional considerations) that have been demonstrated to effectively minimize water quality impacts (Ice 2004).

Proactive and often voluntary practical methods or practices used during forest management to achieve goals related to water quality, silviculture, wildlife and biodiversity, aesthetics and/or recreation (Smallidge and Goff 1998).

For BMPs to be successful, they need to be effective and consistently implemented. By most accounts, adoption and refinement of forestry BMPs over time have been effective in reducing (although not eliminating) water quality impacts resulting from timber harvesting, forest road building and use, and other forest management activities, as compared to these activities without the use of BMPs (Ice et al. 2010; Cristan et al. 2016). Reviews also suggest that implementation rates are generally high (Cristan et al. 2018). But the term “effective” is open to different interpretations, and there is still debate regarding differences in focus between implementation monitoring and effectiveness monitoring, the role of voluntary measures, and assessment of watershed-scale and cumulative impacts (e.g., MacDonald and Coe 2014). For Oregon DEQ’s purposes, “effective” BMPs are those that ensure water quality standards are met and beneficial uses of water are protected and maintained.

Another set of issues involve “lag time” — the time elapsed between when a particular BMP is implemented and the first measurable improvement in water quality in the target water body occurs. If lag time is not accounted for, assessments and monitoring may underestimate BMP effectiveness (Meals et al. 2010). Conversely, lag time can also apply to the time elapsed between when forest management activities take place and detection of any resulting impacts, e.g., residence times for eroded sediment in hill slopes or stream channels, or chemicals in forest soils, before they are detected lower in the watershed.

The concepts that underlie most BMPs emerged from the experiences of working foresters combined with results from scientific studies conducted in the 1950s through the 1970s, mainly at U.S. Forest Service experimental watersheds (Jackson 2014). BMPs are generally understood to be dynamic and always subject to improvement and development (USDA-FS 2012); development of effective BMPs and protection of water quality at the watershed scale has been an iterative process (MacDonald and Coe 2014). Evolution of BMPs continues to this day as understanding of environmental impacts and the effectiveness of control measures advances, resulting for the most part in ongoing refinement of previously developed practices to further enhance effectiveness (Ice 2004, Cristan et al. 2016).

Cristan et al. (2016) reviewed the effectiveness of forestry BMPs, breaking out their summary by region. They compiled results from 31 studies conducted in the West Coast region, mostly the Pacific Northwest and including five studies from western Oregon. Cristan et al. (2016) note that BMPs differ by state and by region, but typically include similar operational categories:

- Forest road construction and maintenance
- Log landings (decks)
- Skid trails
- Streamside management zones
- Stream crossings
- Wetland protection and management
- Timber harvesting

- Site preparation.
- Reforestation

Cristan et al. (2016) concluded:

- BMPs can minimize erosion and sedimentation.
- Implementation rates and quality are critical to BMP effectiveness for reduction of erosion and sediment yield.
- BMP implementation can be enhanced with pre-operation planning and the involvement of a registered professional forester.
- Increased logger training and landowner knowledge of forestry BMPs can help improve implementation.

Cristan et al. (2016) also submitted specific BMP guidelines:

- Forested streamside management zones trap sediment and reduce stream total suspended solids concentrations.
- Critically important BMP practices for forest roads include proper drainage structures, surfacing, erosion control of cut and fill slopes, traffic control and closure.
- Sediment control structures applied to stream crossing approaches can significantly reduce runoff and sediment delivery.
- BMPs need to be applied during forest operations, not merely as a closure measure.
- Effective skid trail closure practices can include installing waterbars and applying slash, mulch, or a combination of mulching and seeding.
- Improved stream crossings such as portable skidder bridges and temporary culverts can decrease total suspended solids concentrations and turbidity compared to unimproved stream crossing structures.

National-level guidance for forestry BMPs is provided in the EPA document *National Management Measures to Control Nonpoint Source Pollution from Forestry* (USEPA 2005). This document summarizes causes of nonpoint source pollution from forestry activities and approaches to reducing the effect of such pollutants. The manual also discusses the application of management measures in a watershed context and nonpoint source monitoring and tracking techniques. Since the document is national in scope and does not address all BMPs specific to regional soils, climate, or forest types it encourages states to utilize research and guidance developed under local harvesting circumstances and to implement the national BMPs within the context of state laws and programs wherever possible.

Oregon’s BMP program is primarily regulatory (the Oregon Forest Practices Act, discussed in more detail below) buttressed by some voluntary measures. The agencies responsible for BMP policy development in Oregon are the Oregon departments of forestry, state lands, agriculture and environmental quality. Some examples of specific BMPs for timber harvesting, forest roads and forest chemicals are discussed below.

3.5.2. Best management practices: timber harvesting

BMPs for timber harvesting related to water quality focus on harvest activities near streams, wetlands or other water bodies. The basic approach is to designate buffer strips along waterways where some or all forest vegetation is left in place to retain mobilized sediment or forest chemicals, to provide shade to maintain or lower stream water temperatures, and to serve as a source for woody debris to maintain certain stream functions. These areas are variously referred to as Streamside Management Zones (SMZ), Streamside Management Areas (SMA), Riparian Management Areas (RMA) and similar terms. Buffer widths may vary by landownership and management strategy, with the Federal government under the Northwest Forest Plan having the widest buffers, and with state forests management plans generally requiring the buffer widths exceeding those of private lands under the Forest Practices Act (Boisjolie et al. 2017).

Typically, the width of stream buffers and the extent of forestry activities allowed within them vary according to the size of the stream, whether the stream contains fish species of concern, beneficial uses of the stream (including drinking water) and other factors. Smaller streams that do not support populations of salmonids and are not specifically designated as sources for drinking water often have no buffers. Harvesting can be precluded in streamside management zones, but in other cases allowances may be made for some limited harvesting activities, e.g., trees of a certain size class, or a certain percentage of trees. These variables have been and continue to be researched extensively, and BMPs are updated and refined based on findings. It has been suggested that adopting a more flexible approach to buffer widths would allow site-specific tailoring to account for local conditions and management goals, but such an approach would be more complicated to administer and monitor for compliance (Richardson et al. 2012).

3.5.3. Best management practices: forest roads

Research shows that unpaved forest roads are a primary source of sediment entering streams and estuaries in forested watersheds (e.g., Reid and Dunne 1984; Amaranthus et al. 1985; Bilby 1985; Ketcheson and Megahan 1996; Luce and Black 1999; Carson and Younie 2003; Endicott 2008). Any forest road, no matter how carefully constructed, may contribute to soil erosion and potential stream sedimentation. Thus, a key tenet of road BMPs is minimizing road number and extent through careful planning (Daniels et al. 2004).

Forest road BMPs continue to be the subject of research. Over time BMPs have been developed and refined for forest road design, placement, construction practices, maintenance, temporary decommissioning, and complete decommissioning and reclamation (NCASI 2009). Three examples of significant areas of improvement are:

1. Actively routing runoff away from existing streams (as opposed routing it into existing channels, as was the previous practice).
2. Improving stream crossings by installation of bridges and/or culverts to keep road traffic from directly crossing stream channels, to minimize disturbance of the stream channel and maintain the integrity of stream structure and function.
3. Upsizing culvert diameters to increase their flow capacity and reduce the likelihood that they will plug during storms, diverting water down roadways and/or causing fill failures.

Other key tenets of forest road BMPs include maximizing the distance between roads and water bodies and minimizing stream crossings, the total area of roads and road grades (Megahan and King 2004).

Sugden (2018) provides a list of current BMPs for forest roads:

- Minimize the road density and area of road prism.
- Locate roads away from streams; i.e., outside streamside management zones unless stream crossings are required.
- Install road drainage features at regular intervals to reduce erosion and divert overland flow from roads onto undisturbed hillslopes to promote water infiltration.
- Ensure road runoff is disconnected from streams toward filtration areas.
- Revegetation and ground cover establishment on disturbed areas near streams (cutslopes, fillslopes and road ditches).
- Gravel surfacing on highly erodible soils or when wet weather use is required.
- Install supplemental filtration for suspended sediments where needed to prevent direct sediment delivery to streams. This includes slash windrows, silt fences and straw bales.
- Install appropriately sized stream crossing structures that allow passage of flood flows, sediment and wood, and minimize disruptions to aquatic species movement.
- Manage or restrict seasonal road access to vehicles as needed to prevent rutting, and perform any necessary maintenance such as grading through time.
- Consider road closure or decommissioning of unneeded roads.

Edwards et al. (2016) synthesized information from almost 800 studies pertaining to BMPs for forest roads. They conclude that forest road BMPs are *generally* effective when “effectiveness” is simply defined as producing less sediment compared to not using the BMP. They also warn that despite the widespread assumption that road BMPs are well-supported by scientific research, rigorous quantitative studies of road BMP effectiveness under different climatic, geologic and topographic conditions are limited. Sources cited as evidence of effectiveness include paired watershed studies with limited pretreatment data and where BMPs are assessed together, making it difficult to assess which particular BMPs were most or least effective. They note that sediment measured at the mouth of a watershed does not account for hill-slope and in-channel storage of eroded sediment, and associated lag times for this sediment to reach the measurement point.

Edwards et al. (2016) also criticize statements that BMPs “minimize” sediment or pollution as misleading. They note that studies on effectiveness often find that some practices are more effective than others, or more effective in some situations than others in reducing sediment. Thus, all practices cannot be effective at “minimizing” sediment, the authors argue, so this term should be avoided because it gives a false impression about the degree of pollutant generation and transport that can be expected with BMP implementation. They note that BMPs cannot and are not intended to completely eliminate pollutants but rather to control them to levels compatible with environmental goals.

A growing area of active research and knowledge is BMPs for the decommissioning and/or removal of old forest roads. This topic is discussed in more detail in Chapter 5.

3.5.4. Best management practices: forest chemicals

Silvicultural chemical BMPs have been developed by many states for fertilizers used to improve crop tree growth and yield and pesticides used to protect trees from competing vegetation and insect pests. BMPs to protect water quality may include multiple layers of specificity based primarily on stream classification, hillside slope, soils and the presence of anadromous fish. As with sediment, the primary means of protecting streams from silvicultural chemicals is usually designation of a streamside management zone that consists of the stream and an adjacent area of varying width where preparation and use of the chemicals is restricted (Michael 2004).

Other BMPs for silvicultural chemicals can be categorized as follows:

- Following all product label instructions
- Disposal of excess chemical and containers
- How, when and where to apply or not apply the chemical
- Maintenance and service of application equipment
- Prevention of direct application to surface water
- Prevention of contamination by drift
- What to do in case of spills

There is some overlap in these categories, i.e., the first is “follow label instructions,” and most labels have instructions regarding disposal of containers, following recommended application rates and some of the other categories.

3.6. Implementing BMPs in Oregon

3.6.1. The Oregon Forest Practices Act: overview and history

The Oregon Forest Practices Act is the state’s primary regulatory framework for addressing the environmental impacts of forest operations on state and private forest lands. The Forest Practices Act sets standards for all commercial activities involving the establishment, management or harvest of trees in the state. The seven-member Oregon Board of Forestry has primary responsibility for interpreting the Forest Practices Act and setting enforceable forest practice rules. Under ORS 468B.110(2), ORS 527.765, and ORS 527.770, the forestry board establishes BMPs or other control measures by rule that, to the maximum extent practicable, will ensure attainment and maintenance of water quality standards.

The Oregon Environmental Quality Commission is a five-member panel of Oregonians appointed by the governor for four-year terms to serve as Oregon Department of Environmental Quality’s policy and rule-making board. The Environmental Quality Commission has the authority to request rule changes to rules in the Forest Practices Act, including strengthening protections for soil and waterways. If the Environmental Quality Commission does not believe that the Forest Practices Act rules will accomplish this result, it is authorized to petition the forestry board for more protective rules.

When passed in 1971, the Forest Practices Act was the first legislation of its kind in the U.S. The Forest Practices Act’s first rules were implemented in 1972 and emphasized BMPs, which have since been revised repeatedly in response to emerging environmental concerns and science findings. Rules for pesticide use were strengthened in 1977 and again in 1996. In 1983, new rules focused on road and log landing parameters were added in response to heightened concern over road-related landslides in western Oregon. Rules to address landslide risks associated with harvesting in steep areas were more controversial, but were enacted two years later. The issue of linkages between forestry and landslides on steep slopes surfaced again 1996, one of the wettest years on record, when impacts from numerous slides in western Oregon increased public attention on the matter. In 1997, additional restrictions focused on public safety were placed on logging on steep slopes near roads or where people might be present (OFRI 2018a, Langridge 2011). Langridge (2011) describes scientific and policy debates associated with the 1997 rule changes and how the issue was framed primarily in terms of human safety while environmental protection was de-emphasized. As of June 2019, the Forest Practices Act does not have any water quality-related landslide-prone area rules.

Rules associated with riparian vegetation and buffer strips have arguably been the most contentious and have evolved to the greatest degree. Riparian rules were modified in 1987 and again, more significantly, in 1994. Increasingly comprehensive and integrated science reports on topics such as the cumulative effects of forest practices (Beschta et al. 1995) and the status of salmonids and their habitat (Botkin et al. 1995), coupled with federal direction to mitigate dwindling salmon runs kept pressure on the forestry board to further restrict harvesting in riparian and landslide-prone areas. But the studies also demonstrated the inherent complexity of these issues (Hairston-Strang et al. 2008).

In 2003, Forest Practices Act rules were updated to require the use of higher quality rock or the suspension of log hauling during very wet weather, based on findings from an Oregon Department of Forestry monitoring study on wet season use of forest roads (Robben et al. 2003, ODF 2003).

The most recent Forest Practices Act rule changes were in 2016 and 2017, and include 60-foot no-spray buffers for aerial herbicide use around homes and schools; a new salmon-steelhead-bull trout category of stream classification and wider riparian buffer strips that must be left around these streams, and additional protections for bald eagles (OFRI 2018b). The salmon-steelhead-bull trout rules are the first change to Forest Practices Act riparian rules since 1994.

3.6.2. Forest Practices Act administration and compliance monitoring

Oregon Department of Forestry (ODF) stewardship foresters administer Forest Practices Act rules by working with forest landowners and operators to help them comply with Forest Practices Act requirements. The Oregon Forest Resources Institute publishes a detailed manual to assist with planning and execution of timber harvests that comply with the Forest Practices Act (Cloughesy and Woodward 2018). The ODF Forest Practices Monitoring Program reviews the effectiveness of the Forest Practices Act and its rules. This program provides science information for adapting regulatory policies and management practices, delivers education and training on Forest Practices Act rules, assesses whether Forest Practices Act rules and voluntary guidance sufficiently protect natural resources, and evaluates whether Forest Practices Act rules are complied with and if voluntary measures are implemented. If Forest Practices Act violations are identified, ODF starts with education and notices of correction before going into formal enforcement. Citations may be issued requiring cessation of the violating practice until agreement is reached on a mitigation strategy, and a legally binding consent order signed (ODF 2019).

Since 2013, compliance monitoring has been conducted through the ODF Private Forests Monitoring Unit using contractors who audit Forest Practices Act rules for road construction and maintenance, timber harvesting, some riparian management area measures, measures for small wetlands, and rules for operations near waters of the state. Audits through 2016 found 97% overall compliance (ODF 2018).

The Forest Practices Act also requires forest landowners and operators to notify the ODF at least 15 days before they begin forest operations on any nonfederal lands in Oregon. As defined in the Forest Practices Act, forest operations include timber harvesting, road construction and reconstruction, site preparation, slash treatment, woody biomass removal, chemical application, land use changes, and certain noncommercial forest activities. In addition, permits are required for any operation using power-driven machinery or fire. The Notification of Operations and Application for Permit (NO/AP) process is conducted through the ODF Private Forests and Protection from Fire divisions. In 2014 the ODF updated the NO/AP process by implementing its Forest Activity Electronic Notification and Reporting System (FERNS), a web-based, centralized database of all forestry operations subject to ODF oversight. The FERNS application is integrated with the state's GIS system. Any interested person or party can subscribe to FERNS and receive electronic notifications of pending forest operations in their area. Subscribers can also review and submit official comments about the forest operation work plans. Online subscriptions to FERNS are free.

About 60% of Oregon's forestland is owned by the federal government, about 34% is privately owned (of which 22% is held by owners with 5,000 acres or more and 12% with less than 5,000 acres), 3% is owned by the state, 1% by local government, and 2% by tribes (OFRI 2017). Because the Forest Practices Act and its rules apply only to nonfederal forestland in Oregon, and to ensure that consistent minimum standards are met, the ODF, U.S. Forest Service, and U.S. Bureau of Land Management agreed that Oregon's forest practice rules would be met or exceeded on federal land in Oregon (Hairston-Strang et al., Adams and Ice 2008). The Clean Water Act requires federal land managers to ensure that their practices will meet state water quality standards, laws and rules (consistency review). In addition, state forests owned by the Department of State Lands and the forestry board typically exceed Forest Practices Act requirements through their management plans.

3.6.3. Oregon Forest Practices Act rules with particular relevance for drinking water

Arguably, the original Forest Practices Act and most subsequent revisions to it were intended primarily to maintain or improve water quality. But certain sections are more directly related to drinking water than others. Minimizing soil disturbance and erosion potential to protect water quality is fundamental to nearly all Forest Practices Act rules for timber harvesting (Division 630). Other Forest Practices Act sections that are relevant for drinking water include:

- Division 620 — Chemical and other petroleum product rules
- Division 625 — Forest road construction and maintenance, and several divisions of the water protection rules
- Division 635 — Purpose goals, classification and riparian management areas
- Division 642 — Vegetation retention along streams
- Division 645 — Riparian management areas and protection measures for significant wetlands
- Division 650 — Riparian management areas and protection measures for lakes
- Division 655 — Protection measures for "other wetlands," seeps and springs
- Division 660 — Stream channel changes

Provisions relating to riparian management areas, streamside buffers, and stream crossings for forest roads are often focused on maintaining conditions for coldwater fish species, but domestic water use is also explicitly referenced in the Forest Practices Act stream classification system. Protection of water quality to benefit fish and maintaining safe drinking water sources for humans are not mutually exclusive goals — measures targeted toward either goal often produce benefits for the other (Abell et al. 2019).

3.6.4. The Oregon Forest Practices Act stream classification system

The Forest Practices Act protection goal for water quality is to ensure that, to the maximum extent practicable, nonpoint source discharges of pollutants resulting from forest operations do not impair the achievement and maintenance of the water quality standards (ODF 2018, p. 53).

The Forest Practices Act uses a stream classification system to align the physical flow characteristics and beneficial uses of a water body to a set of appropriate protection measures. This classification system, and methods by which streams are classified, have been refined over time to reflect new science knowledge or policy imperatives. A *Type F* stream is any stream used seasonally or year-round by anadromous fish, game fish, or fish listed as threatened or endangered under the federal or state endangered species acts. Type F streams may also serve as community water sources. In July 2017, the salmon, steelhead and bull trout (*Type SSBT*) category was added along with modified stream buffer rules to better protect the cooler water quality temperatures needed by these fish. (Groom et al. 2018.) A *Type D* stream is any stream which does not contain fish (as defined above) and is located within a specified distance upstream of any domestic water intake for which an Oregon Water Resources Department permit has been issued. All other streams are classified as *Type N*.

The distance upstream from an intake that Type D (domestic water use) classification applies varies according to whether the intake meets Oregon's definition for a community water supply: has 15 or more service connections used by year-round residents, or which regularly serves 25 or more year-round residents. If the intake meets one of these criteria, Type D classification initially applies to the length of stream that was designated Class I under the classification system in effect on April 22, 1994 (as shown on district water classification maps). If the intake is not for a community water supply (as defined above) Type D classification initially applies for the shortest of 1) the distance from the intake upstream to the farthest upstream point of summer surface flow, 2) half the distance from the intake to the drainage boundary, or 3) 3000 feet upstream from the intake. Type D classification also applies to tributaries off the main channel as long as the above conditions hold.

Streams are further classified by size:

- *Small* — average annual flow of 2 cubic feet per second (cfs) or less
- *Medium* — average annual flow greater than 2 but less than 10 cfs
- *Large* — average annual flow of 10 cfs or greater.

Criteria for establishing average annual flows are explained in Forest Practices Technical Note Number 1 (ODF 1994). Actual measurements of average annual flow may substitute for the calculated flows described in the technical note. Any stream with a drainage area less than 200 acres is assigned to the small stream category regardless of the flow calculated.

3.7. Forestry and drinking water source protection: controversial or unresolved issues

3.7.1. Forest roads and sediment input into streams

Among forestry-related sources of sediment inputs to streams, forest roads are a primary, if not *the* primary contributor. For this reason, runoff from forest roads continues to be a contentious issue relevant to forestry and drinking water source protection. This section summarizes a long-running legal dispute regarding forest roads in Oregon that eventually reached the U.S. Supreme Court.

In 2006, the Northwest Environmental Defense Center (NEDC) sued the State of Oregon, the forestry board, and several timber companies, claiming that forest roads are point sources of pollution under the Clean Water Act and thus require a National Pollutant Discharge Elimination System (NPDES) permit. This challenged decades of precedent under the EPA "Silvicultural Rule" which specifies which types of logging-related discharges EPA considers point sources and excludes forest roads (Boston 2012). The case centered on Clean Water Act language stating that *"the term 'point source' means any discernible, confined and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit ... from which pollutants are or may be discharged"* and whether this should include logging roads that convey sediment through ditches and culverts into nearby streams.

Since the Clean Water Act exempts stormwater runoff, except when the runoff is "associated with industrial activity," the legal case also focused on whether transport of sawlogs constituted an "industrial activity." The EPA Industrial Stormwater Rule lists "logging" as an industry, and "transport of raw materials" as an "industrial activity," but also states that "associated with industrial activity" refers to "manufacturing, processing or raw materials storage areas at an industrial plant." Another area of dispute was whether the phrase term "at an industrial plant" referred to just "storage areas" or also "manufacturing" [or] "processing." [40 CFR § 122.26; *Decker* v. *NEDC*, 568 U.S. 597, particularly Justice Scalia's dissent 616].

In 2007, the U.S. District Court for Oregon ruled that forest roads do not require a NPDES permit. The NEDC appealed. In 2011, the U.S. Ninth Circuit Court of Appeals overturned the district court, ruling that despite longstanding policy to the contrary, the EPA had misinterpreted clear language in the Clean Water Act when drafting its regulations and that roadside ditches on forest roads used to transport sawlogs *do* require an NPDES permit. In 2013, the U.S. Supreme Court, in a 7-1 decision, reversed the Ninth Circuit ruling. Deferring to EPA's interpretation of Clean Water Act language when the agency drafted pertinent regulations, the court cited previous case law that an agency's interpretation need not be the "best" one, only that it be "reasonable." Justice Scalia, in a detailed and strongly worded dissent, sided with NEDC on the merits, arguing that deference was not warranted because EPA language in its Silvicultural Rule clearly conflicts with Clean Water Act definitions of "point source" in the statute and with EPA language elsewhere listing "logging" as an "industry" and detailing what is "associated with industrial activity" (Wasson 2016; Carr and Dively 2013; Boston 2012).

Complicating the Supreme Court case, the EPA amended the Industrial Stormwater Rule just before oral argument in 2012, clarifying that the NPDES requirement applied only to logging operations involving rock crushing, gravel washing, log sorting and log storage facilities. In this amendment, EPA said it would evaluate other silvicultural discharges "under section 402(p)(6) of the Clean Water Act because the section allows for a broad range of flexible approaches that may be better suited to address the complexity of forest road ownership, management, and use." (Carr and Dively 2013.)

In January 2014, Congress amended Clean Water Act Section 402(l), effectively prohibiting the requirement of NPDES permits for the discharge of runoff resulting from a range of silviculture activities, including surface drainage or road construction and maintenance. In December 2014, the Environmental Defense Center (EDC) and Natural Resources Defense Council petitioned the Ninth Circuit to compel EPA to respond, within six months, to a question remanded in a 2003 case (*EDC* v *EPA*) in which EDC contended that EPA arbitrarily failed to regulate discharges from forest roads under its 1999 Phase II stormwater rule. In the 2003 case, the court directed EPA to consider, in an appropriate proceeding, whether Clean Water Act Section 402(p)(6) requires it to regulate forest roads, then either accept or reject EDC's arguments using valid reasoning set forth in a way that permitted judicial review. Following a settlement agreement in August 2015, the Ninth Circuit established a schedule requiring EPA to issue a final determination (Wasson 2016). On July 5, 2016, EPA issued a Notice of Decision not to regulate forest road discharges under Section 402(p)(6) of the Clean Water Act. (USEPA 2016.)

In their rationale for this decision, the EPA acknowledged ongoing and significant water quality impacts attributable to forest road runoff, but argued that many states already have programs to address these impacts that are similar to options that would be available under Clean Water Act Section 402(p)(6). The EPA said progress continues to be made in strengthening these programs to reflect new technology and research, specifically tailored for locations in which they are implemented. Pointing to nationwide diversity in topography, climate, soil types, and intensity of timber operations and water quality impacts, the agency concluded that working with states to strengthen existing programs would be more effective than superimposing an additional federal regulatory layer over them. The EPA argued that despite the potential benefits of a more consistent and enforceable approach to mitigating forest road runoff, the complexity, cost and regulatory burden of a nationwide program could outweigh these benefits.

The EPA indicated that while it had decided not to regulate under Clean Water Act Section 402(p)(6), it would facilitate ongoing improvements in approaches to mitigating water quality impacts from forest roads. EPA said it plans to help strengthen existing programs by forming an ongoing dialogue with all relevant stakeholders (e.g., industry, environmental groups, academics and government agencies at federal, state, tribal and local levels) on program improvements, technical and policy issues, research results, state of the art technologies, success stories, and solutions to problem areas. The EPA envisions a forum where stakeholders can exchange information and expertise on problems and solutions to forest roads, such as existing/legacy roads or stream crossings as well as particularly effective forest road programs and best practices. As an example of a state-led effort to adopt new methods for reducing sediment impacts, the EPA cited a stipulation in California's "Road Rules Package" (California Board of Forestry and Fire Protection 2014) that, where feasible, all forest roads must be hydrologically disconnected from streams (USEPA 2016).

As a result of these court decisions and policymaking, the EPA shifted efforts to regulate forest road runoff from a consistent, nationwide framework developed under Clean Water Act Section 402(p)(6) to state-led nonpoint source programs for forestry. But the EPA also noted that it has other tools to address forest road discharges, such as Sections 303, 305 and 319 of the Clean Water Act.

3.7.2. 'Legacy' forest roads

Nationwide, state-level monitoring shows generally high levels of compliance with forestry BMPs (Cristan et al. 2018). Regulatory frameworks for BMPs continue to be updated to reflect new knowledge and increase their effectiveness. But Oregon and other western states have a large number of so-called "legacy" forest roads that were constructed without BMPs to reduce their impacts. These substandard roads were sometimes poorly sited (e.g., along waterways), constructed with steep grades, or have poorly designed stream crossings. In other cases, problems stem primarily from lapsed maintenance.

Some unmaintained legacy road segments gradually revert back to vegetative cover, but others develop gully systems that become chronic sources of sediment. Legacy road segments that have been stabilized by revegetation can become sediment sources again if they are subsequently encompassed in new harvest units. And a significant number of legacy roads remain in use. "Problem" legacy road segments present a major challenge for managers, because they can generate many times the amount of sediment than roads constructed using modern BMPs (Ice and Shilling 2012) and resources are often scarce to repair and decommission them.

Recent interagency discussions in Oregon on the topic of forest roads and sediment resulted in adoption of the following terms for clarity:

- *Legacy road*: Built and abandoned prior to passage of the Oregon Forest Practices Act and has not been used in the post-Forest Practices Act. The Forest Practices Act does not apply to these roads.
- *Old road:* Built using now-obsolete techniques (e.g., built pre-Forest Practices Act or pre-1984 construction standards) but in use post-Forest Practices Act and therefore subject to Forest Practices Act rules for water quality performance and vacating).

State and federal forest agencies are inventorying and decommissioning or repairing legacy roads. From 1997 to 2013, 2,668 miles of logging roads in Oregon public and private forests were closed or decommissioned (OFRI 2017). From 1995 through 2008, ODF installed 63,055 cross drains on logging roads to route runoff away from streams (Mortenson 2011). But the number of such roads greatly exceeds the resources available to address them, and legacy and old forest roads remain an urgent issue.

3.7.3. Nonpoint source pollution management in the coastal zone

Relationships between forest practices and nonpoint pollution have been contentious in Oregon's coastal zone. The National Oceanic and Atmospheric Administration (NOAA) administers the 1972 Coastal Zone Management Act to address population growth and development in coastal areas by focusing on clean water and healthy ecosystems (NOAA 2018). The Coastal Zone Act Reauthorization Amendments of 1990 included a new Section 6217, "Protecting Coastal Waters,", requiring each state with a coastal zone management program approved under section 306 of the Coastal Zone Management Act to develop and implement a Coastal Nonpoint Pollution Control Program (Coastal Nonpoint Program) to prevent and control polluted runoff.

Section 6217 requires coastal states to implement nonpoint source pollution management measures developed by the EPA, which are organized into two tiers. If the first tier does not enable coastal waters to meet water quality standards and protect designated uses, the state must implement a second tier of "additional" management measures targeted more specifically at restoring coastal waters to maintain water quality standards and to protect beneficial water uses designated by the state (NOAA and USEPA 1993). For Oregon's coastal waters, designated beneficial uses include "public domestic water supply" in all streams and rivers inland from the estuary or head of tidewater influence (Oregon Legislative Counsel Committee 2017; Oregon DEQ 2018b).

Section 6217 also requires each coastal state to submit its Coastal Nonpoint Program, which lays out how they intend to implement their pollution management measures, to the NOAA and EPA for approval. Failure to submit an approvable program can result in a state losing a portion of its federal funding under section 306 of the Coastal Zone Management Act and section 319 of the Clean Water Act.

As required, Oregon submitted its Coastal Nonpoint Program in 1995. In 1998, the NOAA and EPA conditionally approved Oregon's program. Full approval was to be granted when the state met specific conditions, which required application of EPA management measures to address impacts stemming from a range of activities. In regards to forestry, the NOAA and EPA found that the following additional management measures were necessary to meet water quality standards and protect beneficial uses:

- Protect riparian areas for medium-sized and small fish-bearing (type "F") streams and non-fish-bearing (type "N") streams.
- Address the impacts of forest roads, particularly legacy roads.
- Protect landslide-prone areas.
- Ensure adequate stream buffers for the application of herbicides, particularly on non-fish bearing (type "N") streams.

Oregon met nearly all conditions laid out in 1998 by modifying its program over time, but faced challenges in meeting conditions related to development, onsite sewage disposal and forestry. In 2009, Northwest Environmental Advocates (NWEA) sued NOAA and EPA, alleging that despite Oregon's failure to submit an approvable program, the federal agencies had not disapproved the program or withheld grant funds as required and that as a consequence, Oregon had not improved its forest practices sufficiently to protect coastal water quality. In 2010, the Oregon U.S. District Court directed NOAA and EPA to either fully approve or disapprove Oregon's nonpoint program (NWEA 2010a,b).

In 2015, the federal agencies found that the state had met conditions for new development and onsite sewage disposal systems, but not for forestry. As a result, the agencies disapproved Oregon's Coastal Nonpoint Program, triggering a 30% holdback of Oregon's Coastal Zone Management Act Section 306 funds and Clean Water Act Section 319 funds. These funds will be withheld until the state's Coastal Nonpoint Program is approved, and represent a loss of about $1.2 million from roughly $4 million in annual federal grant funding that the state had been using to address coastal pollution (NOAA 2015; House 2016). Programs affected are Oregon Department of Environmental Quality's nonpoint source reduction program, and Oregon Coastal Management Program planning assistance grants for local governments in the coastal zone. As of fall 2018, the Oregon Department of Environmental Quality reported the loss of nearly $2.1 million in Clean Water Act section 319 funding for its nonpoint source program since 2015. As of spring 2019, the Oregon Department of Land Conservation and Development calculated the loss of Coastal Zone Management Act Section 306 funds for the Oregon Coastal Management Program at $2.6 million since 2015 (Oregon DLCD 2019).

In April 2017, the Oregon Board of Forestry adopted a new set of rules to increase shade buffers on small and medium salmon, steelhead and bull trout fish-bearing streams, called the SSBT rule. The SSBT rule covers about a third of Oregon's small and medium Type F stream network according to ODF. The 80-foot buffers adopted are narrower than 90 feet that Oregon forestry and environmental quality staff recommended and the 100 feet recommended by the EPA to ensure compliance with the Protecting Cold Water criterion (OAR 340-041-0028). (Oregon DEQ 2018a.) Progress was made in more clearly defining "legacy" and "old" forest roads and how these are treated under the Forest Practices Act. But no action has been taken regarding additional management measures for landslide prone areas, or buffers on non-fish-bearing (Type "N") streams for protection from aerial herbicide application. Oregon has described the strategies in place (mostly voluntary rather than legally binding) to address these remaining additional management measures, and also pointed to Oregon's strong land use planning system, which has been effective in helping keep Oregon forestland in forest rather than other land uses. But to date the EPA and NOAA have not found these measures to be acceptable and have not approved Oregon's Coastal Nonpoint Program.

As result of such factors as steep topography, high rainfall, and the relatively, small size and close proximity of drinking water source watersheds to commercial timberlands, issues associated with forest management and drinking water protection are likely to remain salient and a source of tension among stakeholders in Oregon's coastal zone.

3.7.4. Aerial herbicide spraying

Aerial spraying of herbicides for the control of understory and deciduous vegetation to promote conifer regeneration is common practice in western Oregon commercial forests, and is a perennial concern among some sectors of the public. In 2017, the Board of Forestry amended Forest Practices Act rules to require that operators leave a 60-foot unsprayed strip between aerially sprayed forests and inhabited dwellings or schools. Efforts to pass more restrictive county-level ordinances have continued, including Measure 21-177 that passed in Lincoln County in 2017. The Lincoln County measure was overturned in court on grounds that it is pre-empted by state law, but the issue of aerial spraying is likely to remain active. The use and effects of forestry pesticides, including aerially-sprayed herbicides, is covered more extensively in Chapter 6.

3.7.5. Landslides

The effects of timber harvesting in steeper, landslide-prone areas on landslide risk and impacts on water quality have been contentious issues in Oregon for several decades (Langridge 2011) and remain so today. Oregon's measures in the Forest Practices Act to mitigate landslide risk were one facet of "additional management measures" that NOAA-EPA indicated were insufficiently addressed in the ongoing Coastal Zone Act Reauthorization Amendments of 1990 and Coastal Nonpoint Pollution program dispute. The Forest Practices Act restricts harvesting in areas of landslide risk that could potentially pose a risk to lives and property, but these measures do not address water quality or aquatic habitat. The effects of landslides on sediment production and water quality are discussed in greater detail in Chapter 5.

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