

The white spruce forests in the Black Hills are important habitats for wildlife. Big game species, such as deer and elk, utilize the white spruce forest for summer-fall range and thermal and hiding cover. Nongame animals also use these forests for habitat. Since these forests have limited distribution in the Black Hills, their contribution to the ecosystem is very important (Hoffman and Alexander 1987).

The white spruce/twinflower (*Picea glauca/Linnaea borealis*) habitat type (fig. 14) is found in the higher elevations and wetter areas of the Black Hills. This habitat is usually found on northwest to northeast aspects and is distinguished from other types by the abundance of twinflower (*Linnaea borealis*) and the absence of grouseberry (*Vaccinium scoparium*) in the understory. Stands are typically found at elevations ranging from 5,800 to 6,423 feet (1,768 to 1,958 m). Soil

texture is loamy with pH ranging from 5.4 to 7.3 and organic matter ranging from 2.7 to 10.5 percent. Other important shrubs in the understory include: bearberry, common juniper, prickly rose, russet buffaloberry, and white coralberry. Major grasses include Kentucky bluegrass and rough-leaved ricegrass. Forbs, such as wild strawberry, sweet-scented bedstraw, American sweetvetch (*Hedysarum alpinum*), and longspur violet (*Viola adunca*), are also in abundance. Mosses and lichens are also prevalent (Hoffman and Alexander 1987).

The white spruce/grouseberry (*Picea glauca/Vaccinium scoparium*) habitat type (fig. 15) is found in the cooler, moist, high elevations of the Black Hills. This habitat is distinguished from other types by the presence of grouseberry in the understory, although its coverage can be variable. Stands are typically found at elevations ranging from 5,697 to 6,693 feet (1,737



Figure 14: White spruce/twinflower (*Picea glauca/Linnaea borealis*) habitat type.



Figure 15: White spruce/grouseberry (*Picea glauca/Vaccinium scoparium*) habitat type.

to 2,040 m). Soil texture is loam to silty clay loam with pH ranging from 4.9 to 6.9 and organic matter ranging from 2.3 to 7.7 percent. Other important shrubs in the understory include: bearberry, Oregon grape, common juniper, prickly rose, wild spirea, and white coralberry. Herbaceous species found in the understory include: yarrow, pussytoes, rough-leaved ricegrass, and Kentucky bluegrass (Hoffman and Alexander 1987).

Natural Disturbances

The ponderosa pine ecosystem of the Black Hills has been subjected to several natural disturbances throughout its history, and these disturbances are an integral part of the ecological processes of the system. Disturbances produce heterogeneity in forest structure that affects timber volume, wildlife habitat, biodiversity, nutrient cycling, and other ecological processes. Disturbances identified include wind, hail, lightning, fire, bark beetles, root rot, neighboring tree fall, weak parasites and pathogens, and ice and snow buildup (Lundquist 1995a, b). Two or more disturbance agents will often interact to kill trees (Lundquist 1995a). Understanding these disturbances and how they shape forest structure and influence ecological processes could allow foresters to better manage their forests by integrating disturbance history into informed management decisions.

Insects

Mountain pine beetle (*Dendroctonus ponderosae*)

The mountain pine beetle is a native insect that plays an important role in the natural disturbance regime in the Black Hills (Lessard 1984; Schmid and others 1994). Each year, various stress factors weaken a few trees in a stand and lower their defenses to mountain pine beetle infestation. Mountain pine beetles carry spores of blue-staining fungus (*Ceratocystis montia*) into the sapwood of the tree (Amman and others 1989a). The fungus develops and spreads throughout the sapwood and interrupts the flow of water to the crown and reduces the tree's sap flow. Endemic population levels of mountain pine beetles, defined as tree mortality of less than 1 tree per acre per year, reproduce in these stressed trees (McMillin and Allen 1999). The combination of the fungus and the beetles cause tree death (Amman and others 1989a).

Periodic mountain pine beetle outbreaks influence both forest structure and growth (Berryman 1986). The mountain pine beetle improves growth rates of trees

adjacent to beetle-killed trees by thinning the stand, thereby reducing competition for light, water, and nutrients. In addition, canopy gaps created by mountain pine beetle epicenters allow regeneration to occur. Even-aged stands tend to convert to irregular uneven-aged stands due to mortality of mid-range diameters (Lessard 1982). Furthermore, mountain pine beetles create snag trees and downed logs for wildlife habitat, but contribute to the fuel load for fires. Coarse woody debris not consumed by fire contributes to microfauna diversity and favors nitrogen-fixing mycorrhizal fungi (Graham and others 1994).

Stress agents such as drought can lower the resistance of a stand and increase mountain pine beetle populations to highly destructive epidemic levels (Schmid and others 1991). Epidemics typically have an 11- to 20-year cycle (Lessard 1984; Pasek and Schaupp 1992) with an outbreak lasting from 2 to 14 years (McMillin and Allen 1999; Schmid and Mata 1992a). From 1895 to 1910, the mountain pine beetle was estimated to have killed trees containing up to 2 billion board feet of lumber in the Black Hills (Boldt and Van Deusen 1974). Outbreaks continued to plague the Black Hills throughout the 1930s, 1940s, 1960s, and 1970s (McMillin and Allen 1999). Tree mortality exceeded 250,000 trees per year for several years during the 1960s and 1970s (Thompson 1975). From 1988 to 1990, 50,000 trees on less than 11,000 acres (4,451 ha) were killed by mountain pine beetle (Pasek and Schaupp 1992). By the mid-1990s, tree mortality by mountain pine beetle was light and scattered throughout the Hills (McMillin and Allen 1999), but aerial surveys in the late 1990s indicated a sharp increase in mountain pine beetle populations with over 25,000 trees killed in 1999 (McMillin and Allen 1999).

In an epidemic, both stressed and healthy ponderosa pine trees are attacked and killed (Lessard 1982). Even-aged densely stocked forests are more conducive to widespread outbreaks than the irregularly spaced uneven-aged forests. Typically trees more than 8 inches (>20 cm) d.b.h. are attacked, but in an epidemic, mountain pine beetles may attack saplings less than 8 inches (<20 cm) d.b.h. if they are intermixed with larger trees (Sartwell and Stevens 1975; Schmid and Mata 1992a). This widespread mortality alters the forest ecosystem in several ways. Forested areas can be converted to grasslands and shrub systems, thus changing the distribution of hiding and thermal cover for wildlife (Amman and others 1989a). The increase in tree mortality will increase water yields in streams for several years following an infestation (Amman and others 1989a; Shepperd and others 1992). Timber stocking levels are decreased, and the dead trees become fuel

for high-intensity fires (Amman and others 1989a; Schmid and others 1985). Understory diversity and biomass also increase following a mountain pine beetle epidemic (Kovacic and others 1985; McCambridge and others 1982).

Life cycle—The majority of the 1-year life cycle of the mountain pine beetle occurs under the bark of the ponderosa pine tree (fig. 16) (Amman and others 1989a). Female mountain pine beetles lay eggs in long, straight, vertical egg galleries in the phloem adjoining the cambial interface. These galleries range from 4 to 48 inches (10 to 122 cm) long, but typically average around 10 inches (25 cm). The small, white eggs are located along the sides of the galleries. The eggs are laid in summer and early fall and hatch within 10 to 14 days.

In August, the eggs hatch and the larvae, white with brown heads, feed on phloem. The larval stage may last until the following June. Short-term temperatures

of 0 °F (−18 °C) and below in October or April decrease brood survival more than if they were to occur in December (Schmid and others 1993). In fact, air temperatures less than −20 °F (−29 °C) must occur continuously for at least 7 days in December to decrease the larval population (Schmid and others 1993).

Once the larvae mature, they excavate oval cells and develop into pupae (fig. 16). Formation of pupae is a function of time and temperature (Schmid and others 1993). By July, the pupae change into adults (fig. 16), which feed within the phloem before they emerge to attack nearby trees (Amman and others 1989a). The emergence of the adult mountain pine beetles is also a function of time and temperature (Schmid and others 1993).

Mountain pine beetle emergence in the Black Hills usually occurs in the last days of July, with peak emergence around August 15 (Schmid 1972). The female mountain pine beetle is the first to emerge, from an exit hole about 0.09 inch (2.4 mm) in diameter (Amman and others 1989a). Daily emergence is affected by air temperature. When maximum temperatures are below 55 °F (12.7 °C) beetles will not emerge (Schmid 1972). Optimal temperature range for emergence is between 60 °F (15.5 °C) and 80 °F (26.7 °C) and appears to be influenced by factors such as humidity, cloud cover, and bark temperature (Schmid 1972; Schmid and others 1991).

Mountain pine beetle attacks show a pattern of reduced frequency during midday and increased frequency around 6:00 p.m. (McCambridge 1967). The decrease in attacks at midday corresponds with the period of highest temperatures in the forest understory where bark temperatures can exceed 90 °F (32 °C) (Schmid and others 1991). The increased frequency of attack around 6:00 p.m. coincides with a period when bark temperatures are decreasing but the air temperature is still above the emergence temperature threshold (Schmid and others 1991).

The process of a mass attack of a susceptible tree is a complex sequence of chemical interactions. The study of pheromone chemistry in ponderosa pine ecosystems is limited; therefore, the following discussion on the sequence of chemical interactions was obtained from research performed in lodgepole pine (*Pinus contorta*) ecosystems. Although ponderosa pine and lodgepole pine are different tree species, the same species of mountain pine beetle attacks both types of trees with similar patterns. The pioneer female is attracted to a susceptible tree emitting monoterpenes such as alpha-pinene (Lindgren and Borden 1989), myrcene (Billings and others 1976; Borden and others 1983), and delta-3-Carene (Miller and Borden 2000). Once a

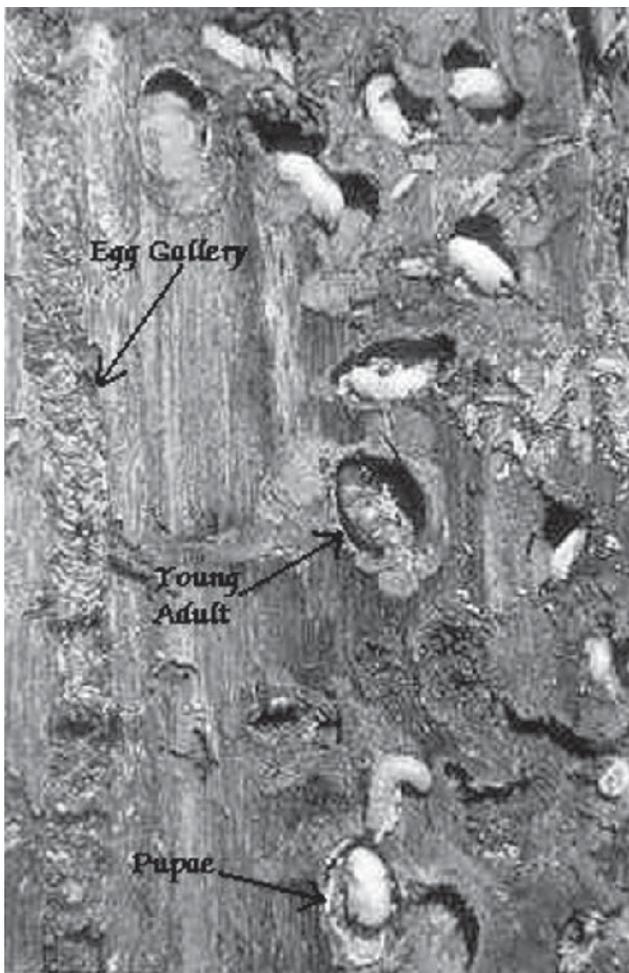


Figure 16: Mountain pine beetle brood at various stages of development in an infested ponderosa pine tree (photo courtesy of Stephen Mata, Rocky Mountain Research Station).

pioneer beetle attacks, this tree becomes a focal point for subsequent attacks from other mountain pine beetles (Eckberg and others 1994) because the unmated female pioneer mountain pine beetle releases an aggregating pheromone, trans-verbenol, which mixes with the tree's monoterpenes to initiate a mass attack by male and female mountain pine beetles (Lindgren and Borden 1989). Males joining the attacking females release low levels of exo-brevicomin (Lindgren and Borden 1989), which attracts mainly females. As the tree becomes occupied by beetles establishing brood galleries, increasing concentrations of exo-brevicomin and frontalin from males and decreasing concentrations of trans-verbenol from females reduce the attractiveness of the tree and act as anti-aggregation pheromones (Lindgren and Borden 1989). Furthermore, autoxidation of cis- and trans- verbenol to verbenone produce a strong anti-aggregation message to prevent overcrowding (Lindgren and Borden 1989). As the focus tree, which is usually a tree under stress, reaches the carrying capacity, subsequent arriving mountain pine beetles attack adjacent healthy trees, which then become focus trees (McCambridge 1967).

Detection—Mountain pine beetle attacks are usually evident on the lower 15 feet (4.4 m) of the tree bole (Amman and others 1989a). In response to an attack by mountain pine beetles, a tree will attempt to produce enough resin to “pitch out” the beetles as they bore into the bole (fig. 17). Unsuccessful attacks will have large pitch tubes (0.75 to 1 inch [19 to 25 mm] in diameter) widely scattered on the trunk. The pitch tubes are often white in color and may have an embedded adult beetle. Successful beetle attacks produce pitch tubes about 1/4 to 1/2 inch (0.64 to 1.27 cm) in diameter with cream to dark-red masses of resin mixed with boring dust. Reddish boring dust at the base and in the



Figure 17: Pitch tubes on a ponderosa pine tree resulting from attack by mountain pine beetle.

bark crevices of the tree is also good indicator of a successful beetle attack. In years of severe drought, pitch tubes may not be evident because trees produce little pitch. A later indication of infestation is the blue staining of the sapwood caused by the fungus carried by the beetle (Boldt and Van Deusen 1974). Several months after the infestation, usually late spring, the tree needles will start to fade from green to yellowish green to sorrel to red and then finally rusty brown. The fading begins in the lower crown and progresses upward (Amman and others 1989a).

Stand susceptibility—Because of the devastating economic impacts that mountain pine beetles can generate, a variety of research studies have been carried out to determine how best to avoid outbreaks and control infestations (Lessard and others 1986). One of the best methods of prevention is to maintain unsusceptible stands. Mountain pine beetles generally infest ponderosa pine trees that are between 8 and 12 inches (20 to 30.5 cm) in diameter (Sartwell and Stevens 1975), although trees larger than 20 inches (51 cm) have been attacked (Obedzinski and others 1999; Olsen and others 1996). Typically, stands with average stem diameters less than 6 inches (15 cm) do not sustain outbreaks because the phloem thickness is inadequate to sustain a large mountain pine beetle population (Sartwell and Stevens 1975).

Stand density is the driving factor of a potential outbreak of mountain pine beetle. Stands with basal areas between 140 and 260 ft² per acre (32 to 60 m² per hectare) are highly susceptible to beetle attack (Olsen and others 1996; Sartwell and Stevens 1975). Schmid and Mata (1992b) suggested stands over 120 ft² per acre (27 m² per hectare) were highly susceptible. In their study, they noted an absence of tree mortality in stands with Growing Stock Levels (GSL²) of 60, 80, and 116, even though the average diameters were in the susceptible-size category of greater than 8 inches (>20 cm). Several studies have attempted to explain this trend with empirical data.

As stand density increases, the amount of competition between trees within the stand increases. This intensive competition in high-density stands lowers a tree's resistance to beetle attack and represents a key feature in mountain pine beetle outbreaks (Sartwell and Stevens 1975). For instance, Sartwell (1971) examined the radial growth of 666 beetle-killed trees and reported that only three had grown over 1 inch in

² Growing Stock Levels (GSL) are defined as the residual square feet of basal area when average diameter is > 10 inches (Alexander and Edminster 1981).

diameter in the previous 10 years, and 435 trees had not grown an inch in the previous 20 years. Even in managed stands with increased spacing and decreased competition, some individual trees will grow slowly and will be less resistant to mountain pine beetle attack (Obedzinski and others 1999).

Larsson and others (1983) suggested that susceptibility to damage was closely related to tree vigor, which has been demonstrated to respond to stocking levels. In their study, the number of trees attacked per plot decreased as average tree vigor increased. Average tree vigor decreased with denser tree spacing and larger leaf area index. They noted a threshold response in which plots with vigor of less than $100 \text{ g m}^{-2} \text{ yr}^{-1}$ (based on amount of stem wood produced per square meter of leaf area) had significantly higher average percentage of trees attacked.

The ability of a tree to resist a mountain pine beetle attack has been linked to the amount of carbohydrates that can be utilized directly for defensive wound reactions (Christiansen and others 1987). Environmental factors that restrict the size of a canopy or its photosynthetic efficiency weaken tree resistance. Drought can affect the carbon balance of a tree by halting photosynthesis, which depletes carbon reserves for defensive compounds and eventually reduces the size of the canopy. Tree vigor also decreases when live crown ratio decreases to 30 percent (Smith and others 1997). For instance, Sartwell (1971) examined nearly 4,000 beetle-killed trees and reported that 91 percent had foliage-bearing branches on less than one-third of the stem height.

Based on several studies performed in the Black Hills, Schmid and others (1994) developed a hazard-rating guide based on mean stand diameter, stand density, and spacing for even-aged ponderosa pine stands (fig. 18). Schmid and Mata (1992b) suggested that the critical threshold for mountain pine beetle infestation in Black Hills ponderosa pine stands might be in a stand of GSL 120, instead of an earlier level of GSL 150 proposed by Sartwell and Stevens (1975). Stands with GSLs between 80 and 120 are considered moderately hazardous, while stands with GSLs less than 80 are considered to be at low risk for infestation (Schmid and others 1994). Growth potential of a partially cut stand should be considered because a GSL 80 stand can grow into a high-hazard stand within 10 to 20 years depending on its growth rate (Schmid 1987). It should be emphasized that the hazard rating was developed for relatively homogeneous even-aged stands, and that the method needs modification for locations where stands are composed of low-density but uniformly spaced even-aged trees intermixed with clumps of high

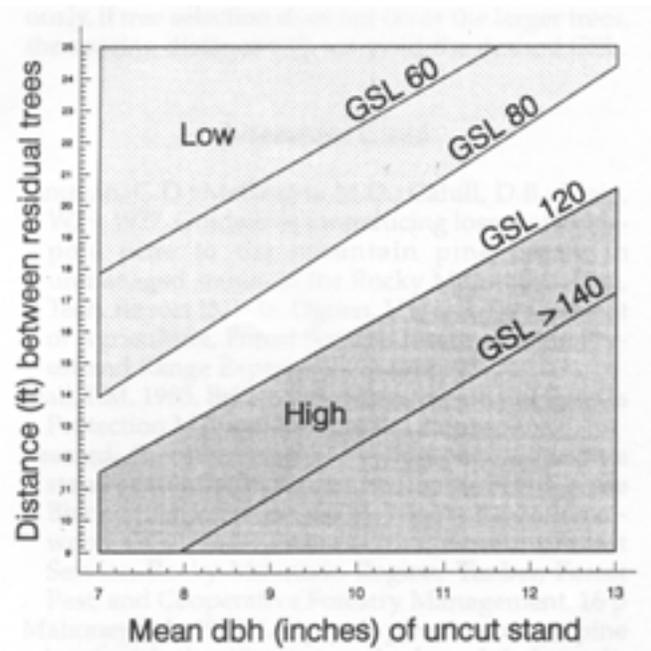


Figure 18: Hazard levels of mountain pine beetle infestation of even-aged ponderosa pine stands in the Black Hills (graph adapted from Schmid and others 1994).

density trees or where stands are uneven aged (Schmid and others 1994).

Semiochemical control—Semiochemicals are naturally occurring compounds that an individual of a species creates to produce a behavioral response in other individuals of the same or different species (Lindgren and Borden 1989). These compounds have shown promise in managing insect populations (Schmid and others 1989). Several methods have been tested with mixed results, including mass trapping, tree baiting, and repellants.

Mass trapping uses funnel traps to suppress the mountain pine beetle population (Amman and Lindgren 1995). The funnel traps are baited with pheromones to attract the mountain pine beetle, but beetles attracted to the traps frequently attack nearby trees. On an operational level, mass trapping probably has little or no potential for mountain pine beetle control (Amman and Lindgren 1995). The technique is commonly used to detect the presence or absence of mountain pine beetle populations, however (Schmid and others 1989).

Tree baiting is an operational procedure adopted by the British Columbia Ministry of Forests to manage mountain pine beetle populations in lodgepole pine forests (Amman and Lindgren 1995). In contrast to mass trapping where the objective is to reduce populations, tree baiting manipulates the beetle into attacking selected trees (Borden and others 1983). For instance, lodgepole pine trees baited with myrcene, trans-verbenol, and exo-brevicomin had higher attack densities than unbaited trees. In the

study, 59.8 percent of all attacked trees and 68.1 percent of mass attacked trees (attack density of 336 per ft²; 31.25 per m²) were either baited trees or trees surrounding the baited trees (Borden and others 1983). In theory, by concentrating the beetle attacks into a defined area, the newly infested trees could easily be located and cut. Although tree baiting does not directly control insect populations, used in conjunction with silvicultural techniques it can be very effective (Amman and Lindgren 1995). However, absence of harvesting or other followup control treatments can intensify the infestation problem since both attack densities and tree mortality are increased where baits are used (Amman and Lindgren 1995).

Tree baiting has been successfully tested in ponderosa pine forests in the Black Hills. Beetle baits of trans-verbenol (Phero Tech) were placed on two uninfested green trees immediately adjacent to 11 infested areas (defined as 1 to 13 infested trees) (Schmid and others 1989). Attack densities were significantly higher on baited trees than on unbaited trees. Furthermore, 82 percent of the newly infested trees developed within a 55-foot (17-m) radius of the baited trees. Because tree baits have a limited range of effectiveness, baited trees should be no more than 300 feet (91 m) and probably only 100 to 150 feet (30 to 46 m) from the source of mountain pine beetles. The effective distance is also influenced by wind direction (Schmid and others 1989). To improve containment, the bait should be placed on a tree in the densest stand adjacent to the existing infestation. Trees should be baited before beetle emergence, preferably in early July, with baits placed on the north side of susceptible trees as high as the applicator can reach (Amman and Lindgren 1995; Schmid and others 1989). This strategy is particularly appropriate in the beginning of an outbreak when small group infestations are evident (Schmid and others 1989). Tree baiting might also be useful in groups of susceptible trees scheduled for removal under group selection management, providing the baited trees can be removed before adults fly the next year.

The use of anti-aggregating pheromones has been shown to reduce attack densities of mountain pine beetles on several pine species (Amman and others 1989b; Lindgren and Borden 1989). Verbenone is produced by the autoxidation of cis- and trans-verbenol, a pheromone produced by mountain pine beetle (Lindgren and Borden 1989). In British Columbia, field tests of commercially produced verbenone in the presence of mountain pine beetle bait inhibited both the attraction of male and female mountain pine beetles to funnel traps (Borden and others 1987). The presence of verbenone baits reduces attacks, but does not

prevent tree mortality (Amman and others 1989b; Lindgren and Borden 1989; Lindgren and others 1989).

In the Black Hills, two studies have investigated the use of verbenone to decrease the number of trees attacked by mountain pine beetles (Bentz and others 1989; Lister and others 1990). Verbenone levels tested were 0, 10, 20, 40, and 68 capsules per acre (0, 25, 49, 100, 169 capsules per hectare) at a release rate of 5 mg per day per capsule (Bentz and others 1989; Lister and others 1990). In both studies, there was a downward trend in attacked trees with increased numbers of verbenone capsules, but due to the high variation and low number of replicates, significant treatment differences could not be detected (Bentz and others 1989; Lister and others 1990). Current studies in the Black Hills are investigating this further. Even though mortality is not eliminated with the use of verbenone, it may be feasible in areas where some mortality can be tolerated (Lindgren and Borden 1993).

The use of anti-aggregation pheromones cannot be recommended on an operational basis due to the inconsistent results. The effect of verbenone on beetle attacks may depend on whether the infestation is endemic or epidemic. Lindgren and Borden (1993) found an increase in the percentage of lightly attacked trees in verbenone treated subplots in a lodgepole pine study. The authors suggest that the verbenone interfered with the ability of pioneer beetles to attract large numbers of additional beetles to initiate the mass aggregation necessary to overcome tree defenses. They expect that verbenone would work best at endemic levels, but at epidemic levels, multiple pioneer beetles attacking the same tree may release enough aggregating pheromone to negate the effect of verbenone.

Chemical control—Insecticides offer temporary management of mountain pine beetles by slowing infestation (Amman and others 1989a). These agents also affect nontarget species, however, and use in forested ecosystems is less desirable (McCambridge 1982). In areas with high-value trees, such as campgrounds and on private property, preventative spraying may be necessary. Spraying individual trees may provide effective protection if applied correctly, but it does not address the cause of the outbreak (McMillin and Allen 1999). Furthermore, it does not guarantee protection and the application must be thorough. To be effective, the insecticide must be applied directly to the boles of individual trees with a high-pressure sprayer, prior to beetle flight, forcing the insecticide into crevices in the bark. The entire bole must be treated to a top diameter of 6 inches (15 cm) or less. Annual treatments are necessary for as long as the outbreak is occurring (McMillin and Allen 1999).

If the use of insecticides is the chosen method, trees must be treated before the beetles emerge (McMillin and Allen 1999). Carbaryl (tradename Sevin) has been shown to be effective in protecting test trees from mass attack (Shea 1989). It is most effective in concentrations ranging from 0.5 to 2.0 percent and with water used for dilution having a pH of 6 (Leatherman 2001a). Carbaryl at concentrations greater than 2 percent did not increase protection success (McCambridge 1982). Carbaryl may be effective for up to 2 years. Permethrin (trade names: Astro or Dragnet), a synthetic form of pyrethrin, was shown to prevent beetle attacks in California, Montana, and the South (Leatherman 2001a), and preliminary results indicate that it provides one year of protection in the Black Hills.

Silvicultural control—Several silvicultural activities can be used to minimize the chance that endemic mountain pine beetle populations may become epidemic. Options include thinning, salvage/sanitation harvesting, and treating infested logs. If the manager takes the stance that beetle-caused mortality and the impacts associated with this mortality are part of the natural process, then the best option would be to do nothing (McMillin and Allen 1999). Some advantages of doing nothing are no mechanical site disturbance, increase in light and enhancement of understory species and regeneration, and enhancement of habitat for some wildlife from the formation of snags and decreased crown closure. A risk of this decision is that beetle populations may spread to adjacent land. If timber productivity is a primary objective, there will be a loss of timber revenues as well as an increase in fire hazard (McMillin and Allen 1999).

Some management practices can help impede mountain pine beetle population explosions. One key to inhibiting epidemic mountain pine beetle populations is to maintain a healthy population of trees that can resist mountain pine beetle attack. Active forest management with traditional thinnings is crucial for reducing competition, sustaining healthy trees with good growth rates, and minimizing mountain pine beetle-caused tree mortality (Olsen and others 1996). Without a thinning program, stands will eventually grow into dense, unhealthy conditions and again be susceptible to mountain pine beetle attack.

For instance, basal area growth rates of 1.3 ft² per acre (0.298 m² per hectare) or more are possible in partially cut stands (Obedzinski and others 1999). If 150 ft² per acre (34.43 m² per hectare) is considered the threshold level for epidemic mountain pine beetle populations (Sartwell and Stevens 1975), stands that increase 1.5 ft² per year (0.14 m² per year) need to be thinned about every 20 years (Obedzinski and others

1999). For GSL greater than 120 growing at 1.5 ft² per year (0.14 m² per year), the threshold would be reached before the 20 years and would need to be thinned earlier. For stands of GSL less than 120 and greater than 80, the threshold may be exceeded between the first and second entry and should be cut (Obedzinski and others 1999). If 120 ft² per acre (27.55 m² per hectare) is considered the threshold level (Schmid and Mata 1992b), then GSLs greater than 100 would reach the susceptible level before 20 years, while reentry in a GSL 80 might be necessary halfway between the first and second reentry (Obedzinski and others 1999).

A problem arises when trying to apply even-aged principles and susceptibility thresholds to uneven-aged forest structures. The susceptibility thresholds in terms of average d.b.h. and basal area are probably different for even-aged and uneven-aged forests (Olsen and others 1996). Mean stand density, a common descriptor of even-aged forest stand structure, does not provide a useful description of tree groups within an uneven-aged forest. In some uneven-aged stands, particularly susceptible groups of trees may exist unintentionally due to management practices (Olsen and others 1996). Furthermore, delaying scheduled thinnings in uneven-aged stands might increase the chance of increased group densities and lack of detection in average stand density estimates. Large groups containing densely spaced trees in susceptible diameter classes are certainly more at risk of attack. Smaller clumps might be less so, due to increased vigor from edge effect, but no empirical data are available. Conversely, managed uneven-aged stands might be at somewhat less risk to mountain pine beetle outbreaks because the portion of total basal area in susceptible d.b.h. classes would be below threshold limits. If uneven-aged stands were managed with clumped or grouped spacings as called for under current Northern Goshawk (*Accipiter gentiles atricapillus*) habitat guidelines (Reynolds and others 1992), individual clumps of trees might be susceptible to mountain pine beetle, but the discontinuity of suitable host trees might slow or prevent large-scale mountain pine beetle outbreaks. Further research is needed to assess this situation and for guidance in defining threshold limits for uneven-aged forest stands. Managers should monitor mountain pine beetle activity in stands managed under irregularly spaced uneven-aged prescriptions to test these hypotheses.

Salvage and sanitation harvesting may be appropriate when it is desirable to reduce mountain pine beetle populations and recover timber resource value. Salvage harvest removes the dead trees, while a sanitation harvest removes infested pines before the beetles mature and leave to infest new trees (McMillin and

Allen 1999). Sanitation harvest can also remove trees that are susceptible to infestation. Sanitation harvesting can be used in conjunction with semiochemical tree baiting. An advantage of salvage and sanitation harvesting is that beetle populations can be reduced in a localized area, especially when tree baits are used to concentrate the population (McMillin and Allen 1999). More advantages include recovered timber value, fuel hazard reduction, and regeneration enhancement by overstory removal and site disturbance. A major disadvantage is the short execution time and the possibility of spreading the infestation to other areas during transportation of the cut wood. The infested areas must be marked, cut, and processed before the beetles emerge. In addition, success on a landscape level will need to incorporate the sanitation of infested stands as well as thinning of susceptible stands to decrease the chance of infestation, since sanitation cuts are only effective on a stand level (McMillin and Allen 1999).

A spinoff of the salvage/sanitation harvesting method is the fell and treat method. Beetle populations are reduced or eliminated from the treated area and site disturbance is typically less than in conventional harvest. As with any overstory removal process, regeneration is enhanced and crown fire hazard is reduced (McMillin and Allen 1999). However, execution time is short and the treatment must be applied after newly infested trees are located and before beetle emergence. Furthermore, treatment only reduces the beetle pressure on a stand level and does not address the stand conditions that lead to the infestation (McMillin and Allen 1999).

Infested trees can be felled and exposed to solar radiation to kill mountain pine beetles. The premise of this technique is that increased bark temperature creates lethal temperatures within the phloem that kill the mountain pine beetle larvae (Negron and others 2001). Furthermore, increased humidity levels encourage mold growth beneath the bark, which minimize normal beetle development (Leatherman 2001b). Beetle mortality of 80 to 90 percent is realistic when maximum bark temperatures exceed 104 °F (40 °C). Treatments should be initiated in late May/early June with a duration of at least 6 weeks. There are two different approaches for this technique: with clear plastic or without plastic (Negron and others 2001).

When using the solar radiation technique with plastic, the main objective is to increase the temperature and humidity of the felled logs to inhibit the development of the mountain pine beetle. Logs should be cut into 4-foot (1.2-m) sections, moistened with water, and covered with clear plastic. When using plastic, logs can be oriented in one or two layers, but stacks more

than two layers high may not increase temperatures enough to reduce mountain pine beetle populations (Negron and others 2001). Logs should be placed in an area that receives direct solar radiation. After the logs are covered with plastic, the edges of the plastic should be covered with surrounding soil. Treat the logs for at least 2 months to kill mountain pine beetles. After the treatment is complete, remove and dispose of the plastic sheeting. The wood can then be safely stored or used without infecting live trees (Leatherman 2001b).

The solar radiation technique without plastic involves using sunlight to heat the felled trees to lethal temperatures. Infested trees should be cut into 4-foot (1.2-m) long logs and delimbed. Logs need to be placed in a sunny location, preferably on southern slopes. Arrange the logs side by side and do not stack them because the bark surfaces need direct sunlight to reach lethal temperatures. Logs need to be rolled at least three times during the treatment period to ensure all sides have been exposed to direct sunlight. If logs are not rotated, then the north aspect of the log should face the sun to ensure a decrease in brood populations (Negron and others 2001). The survival status of the beetles should be checked by the end of June (Leatherman 2001b). If beetles are still alive, logs should be covered with plastic, or stripped of bark to prevent beetle escape.

Pine engraver (*Ips pini*)

The pine engraver is a native beetle in the Black Hills. In general, pine engraver beetles are nonaggressive and breed in windthrown ponderosa pine trees, trees damaged by wind and snow, and logging slash greater than 2 inches (5 cm) in diameter (Boldt and Van Deusen 1974; Kegley and others 1997). However, if populations become larger than the surrounding slash can absorb, the beetle may attack nearby live trees. Pine engravers have the potential to cause damage to surviving trees after a wildfire (Kegley and others 1997). Fire scorched trees that still have suitable phloem are frequently attacked, and there is a chance for green stands surrounding a burned area to be attacked if the pine engraver population reaches high levels.

In the event that pine engravers do attack live trees, risk is highest for trees 2 to 8 inches (5 cm to 20 cm) in diameter (Boldt and Van Deusen 1974). Although not as aggressive as the mountain pine beetle, pine engravers have been reported to kill up to 500 trees in an unthinned young stand (Kegley and others 1997). Large trees that are attacked are often top killed by the pine engravers, while the lower bole is infested by other insects (Kegley and others 1997).

Outbreaks of pine engravers are linked to the amount of precipitation from April through July (Kegley and others 1997). If precipitation is 75 percent of normal or less, moderate to heavy tree mortality can be expected in overstocked, second-growth ponderosa pine stands. Typically, these outbreaks only last one season, but in cases of extreme drought, damage can continue for 2 or 3 years.

Life cycle—The pine engraver life cycle from egg to adult is approximately 40 to 55 days long (Kegley and others 1997). Because of this short life cycle, pine engravers can produce 2 to 3 generations from April through late September (Kegley and others 1997). The flight of the first generation occurs in April or May when the daily maximum temperature reaches 60 to 70 °F (15.5 to 21 °C) (Kegley and others 1997). The attack is pioneered by the male beetle, which is attracted to a stressed tree releasing monoterpenes. The male bores through the outer bark into the phloem and excavates a nuptial chamber. Once the chamber is ready, he releases an aggregation pheromone, Ipsdienol, to attract several females and males (Kegley and others 1997; Raffa and Dahlsten 1995). An attacked tree is indicated by reddish-orange boring dust in small mounds on the bark surface at the point of entry (Kegley and others 1997).

One male will mate with 3 or 4 females, and once mating has occurred, each female constructs a 4- to 7-inch-long (10 to 18 cm) egg gallery in the phloem layer (Kegley and others 1997). Several galleries radiate from the nuptial chamber, often forming a Y or H pattern aligned with the grain of the wood (fig. 19). Once the egg gallery is formed, the female lays 30 to 60 oval, pearly white eggs along the sides. The eggs hatch into larvae within 4 to 14 days.

The larvae are creamy-white, legless grubs with brown heads and about 0.2 inch (0.5 cm) long (Kegley and others 1997). The larvae mine laterally up to 1 to 2 inches (2.5 to 5 cm) from the egg gallery and feed for 2 to 4 weeks. At the end of this time, each larva excavates an oval chamber at the end of the tunnel to pupate. The pupae stage typically lasts for 12 days.

The adult pine engraver beetle is cylindrical, about 0.125 to 0.2 inch (0.3 to 0.5 cm) long, and has four small spines on each side of its lower posterior (Kegley and others 1997). If it is still early in the year, the adult beetle will emerge and reproduce, otherwise, it will spend the winter in the adult stage under the bark of an infested tree, slash, or in the duff and litter on the forest floor.

Control—Several management practices can minimize infestation by the pine engraver beetle. The most important practice is to maintain a vigorous growing



Figure 19: Pine engraver breeding gallery in ponderosa pine (photo source: Kegley and others 1997).

forest. Stands with basal areas of 80 to 100 ft² per acre (18.37 to 22.96 m² per hectare) have been shown to be less susceptible to pine engraver beetle attack (Kegley and others 1997).

Since pine engraver breed in slash, two methods have been proposed to control their population: (1) provide enough slash for endemic populations; or (2) promptly dispose of slash to minimize populations.

If enough slash is provided, the pine engraver beetle may ignore the green trees surrounding the slash piles. This technique calls for providing a continuous supply of fresh slash throughout the flight period of emerging adults (Kegley and others 1997). New slash should be produced at the pupal stage, and a large slash pile should be created prior to the first beetle emergence in the spring. The piles should be large enough (20 feet [6m] wide and 10 feet [3m] deep) to keep the interior pieces moist enough to support beetle reproduction. When the adults emerge, they will go deeper into the pile and stay away from the standing green trees (Kegley and others 1997). There are some potential problems with this method, however. First, to create sizable slash piles, there is need for constant timber operations, which is not always feasible. Second, this method has the potential to increase the pine engraver population

to an epidemic level and increase the probability of attack on green trees.

Promptly disposing of slash is the other method to help minimize a population explosion (Kegley and others 1997). Removal of slash is especially important when slash is created from early winter through late spring because it provides a large amount of breeding material (Gara and others 1999; Kegley and others 1997). If logging does occur during this time, burning the slash can destroy potential brood sites, but workers must be careful not to scorch the residual trees. The optimal time to log is late summer to early winter because this allows time for the phloem in the slash to dry out and become unsuitable pine engraver habitat. If the slash is covered by early snows, however, it may maintain enough phloem moisture to attract beetles in the spring. Therefore, scattering the slash material around the site and exposing it to direct sunlight can minimize possible pine engraver habitat (Kegley and others 1997). Gara and others (1999) reported that drying host material and predation by natural enemies helped regulate endemic pine engraver populations in southeastern Montana.

Several insects and birds play a role in lowering pine engraver populations, although they are not known to prevent outbreaks (Kegley and others 1997). Wasps and nematodes are parasitic, while woodborer larvae (family Cerambycidae) can unintentionally munch on pine engraver broods as they feed indiscriminately through the galleries. Predators such as woodpeckers and several beetle species can consume large numbers of pine engravers. Because the predator beetles blackbellied clerid (*Enoclerus lecontei*), redbellied clerid (*Enoclerus spegeus*), and trogositid beetle (*Temnochila chlorodia*) emerge at different times from April to August, competition among them is low, and predation is constant throughout most of the pine engraver emergence. The predator beetles all respond to pine engraver pheromones (Gara and others 1999).

Semiochemicals—Competition among developing pine engraver broods as well as with other bark beetles and woodborers can reduce pine engraver populations. Mountain pine beetles and pine engravers often coexist in the same tree but in separate areas of the bole (Furniss and Carolin 1977). In lodgepole pine forests in British Columbia, Rankin and Borden (1991) reported that extensive pine engraver attacks before mountain pine beetle could establish gallery systems exerted a major negative effect on mountain pine beetle offspring. However, pine engraver beetles and mountain pine beetles can coexist within the same tree if populations are low due to the mutual exclusion created by the semiochemicals produced by both beetles

(Rankin and Borden 1991). The mutual exclusion occurs because the mountain pine beetles produce verbenone, a repellent to pine engravers (Miller 1991), while the pine engraver produces Ipsdienol, a repellent to mountain pine beetle (Hunt and Borden 1988). Pine engraver attraction to Ipsdienol was significantly reduced in the presence of trans-verbenol and exobrevicomin, two pheromones produced by mountain pine beetles (Hunt and Borden 1988).

The chemical (+)-Ipsenol has been shown to reduce pine engraver attacks of ponderosa pine by 68 percent at release doses of 2 mg per day (Furniss and Livingston 1979). Borden and others (1992) reported a mixture of verbenone and ipsenol dramatically reduced pine engraver attack density and lowered the number of emergent brood on felled and standing lodgepole pine in British Columbia. When ipsenol was applied alone, however, there was not a protective effect (Birch and Light 1977; Borden and others 1992).

Lanierone is another male-produced aggregation pheromone for some populations of pine engravers (Teale and others 1991). The attraction of pine engravers to ipsdienol is synergized by lanierone. However, this response has only been documented among eastern and midwestern populations of pine engravers (Miller and others 1997; Teale and others 1991). Among western populations, Miller and others (1997) showed only a weak significance in Montana and British Columbia, while populations in California showed no response (Miller and others 1997; Seybold and others 1992). Ipsdienol enantiomers have a high degree of geographic variation (Raffa and Dahlsten 1995), which might be the case for lanierone (Miller and others 1997). Miller and others (1997) suggested that specific mixtures of pheromones might be specific to a region due to other species utilizing the same pheromones. For instance, the blackbellied clerid predator beetle can detect the lanierone pheromone. In California, however, pine engravers were found not to respond to lanierone, suggesting that pine engraver beetles that utilized lanierone would have a greater selection pressure toward them.

Utilizing pheromones to control pine engraver populations is still in its infancy. Furthermore, no studies have been performed in the Black Hills to determine if pine engravers will respond to lanierone, or which enantiomers of ipsdienol or ipsenol attract or repel the Black Hills pine engraver population.

Red turpentine beetle (*Dendroctonus valens*)

The red turpentine beetle is a native bark beetle of the Black Hills. In contrast to the mountain pine beetle, the red turpentine beetle is not aggressive. Red turpentine

beetles are not usually a primary cause of ponderosa pine mortality in the Black Hills; instead, they are a secondary cause of mortality. Red turpentine beetles typically attack trees already weakened by drought, fire, insects, lightning (Boldt and Van Deusen 1974) or logging damage (Smith 1971). Occasionally, red turpentine beetles will be the primary attacker of a tree and cause the tree to be attacked by other insects or diseases (Boldt and Van Deusen 1974).

Trees greater than 1 inch (2.5 cm) in diameter are subject to red turpentine beetle attack (Boldt and Van Deusen 1974). Weakened trees are often attacked, but trees close to freshly cut stumps or green logs are also susceptible (Boldt and Van Deusen 1974; Smith 1971). Schmid and Mata (1991) reported that the percentage of red turpentine beetle-attacked trees in a Black Hills ponderosa pine stand was greater in stands that were located near recently logged areas. Furthermore, all red turpentine beetle attacked trees were attacked within the first year after logging. Although these trees were attacked, red turpentine beetles did not appear to be a significant mortality factor in either uncut or partially cut stands.

Red turpentine beetles can increase tree mortality in areas that have been disturbed by fire (Smith 1971). Eckberg and others (1994) reported that red turpentine beetle attacks were evident on fire-scorched trees in the Black Hills. This raises some concerns for the fire-dependent ponderosa pine ecosystem of this region, especially in the aftermath of wildfires. For example, trees that survived but were damaged by the catastrophic Jasper Fire in the southern Black Hills in 2000 might be subjected to red turpentine beetle attack, and the population of red turpentine beetles might build up enough to attack nearby green trees (USDA Forest Service 2001a). Current and future monitoring of the Jasper Fire area will aid in determining mortality caused by red turpentine beetles.

Life cycle—There is typically one generation of red turpentine beetles per year (Smith 1971). Pupae and eggs rarely overwinter (Smith 1971), but adults and larvae do overwinter (Furniss and Carolin 1977). The distinctive large (0.5 to 1 inch wide) longitudinal (vertical) egg galleries of red turpentine beetles differ from other bark beetles (Boldt and Van Deusen 1974). The eggs, which are shiny, opaque white, ovoid cylindrical, and approximately 0.04 inch (1 mm) in length, are laid in an elongated mass along the side of the egg gallery and can number from a few to more than 100 (Smith 1971).

The eggs hatch into larvae within 1 to 3 weeks after being laid (Boldt and Van Deusen 1974; Smith 1971) and look grublike, legless, and white, except for a

brown head capsule and a small brown area at the hind end. As the larvae grow, a row of small, pale-brown tubercles develop along each side of the body (Smith 1971). Larvae are typically 0.39 to 0.47 inch (10 to 12 mm) long when fully grown. The larvae feed on the inner bark tissue between the outer dry bark and the wood. As the larvae grow, they feed constantly and form a fan-shaped cavity with an irregular margin, a characteristic feature of red turpentine beetle infestation (Boldt and Van Deusen 1974; Smith 1971).

Once the larvae are ready to pupate, they create separate cells and transform into adults (Boldt and Van Deusen 1974). The pupae are slightly shorter than the larvae, but are still white (Smith 1971). Close examination of the pupae shows the wings, legs, and antennae being held against the body. An adult beetle, 0.24 to 0.39 inches (6 to 10 mm) in length and quite stout, finally emerges from the pupal cells by boring through the bark (Boldt and Van Deusen 1974) in the spring. A distinguishing feature of a red turpentine beetle adult is that they are reddish brown and large (up to 0.375 inch; 0.95 cm), while the mountain pine beetle adult is black and smaller (0.125 to 0.33 inch; 0.3 to 0.85 cm).

A red turpentine beetle may fly up to several miles to attack a new host (Boldt and Van Deusen 1974). The female red turpentine beetle is the first to attack a weakened tree, and bores into the bark to reach the surface of the wood (Smith 1971). The male follows shortly afterward (Boldt and Van Deusen 1974) and an egg gallery is excavated. Typically, only one pair of beetles is found per gallery (Smith 1971).

Generally, red turpentine beetles concentrate their attack in the lower 6 feet (1.8 m) of the tree, but can reach up to 12 feet (3.7 m) (Boldt and Van Deusen 1974; Smith 1971). Trees attacked by red turpentine beetles usually show pitch tubes on the outer surface of the bark and boring particles (reddish frass) either in bark crevices or on the ground at the base of the tree (fig. 20) (Boldt and Van Deusen 1974; Smith 1971). Pitch tubes are large (up to 1.5 inches [3.8 cm] in diameter) and reddish white in color. Mountain pine beetle pitch tubes are sometimes confused with red turpentine beetle pitch tubes. Successful mountain pine beetle attacks produce pitch tubes that are smaller (0.25 to 0.5 inch [0.64 to 1.27 cm] in diameter) and are cream to dark red in color. Unsuccessful mountain pine beetle attacks produce pitch tubes that are to 1 inch (1.9 to 2.5 cm) in diameter.

Fading needles indicate the infestation was successful and the tree is dying. The needles will fade in a progression from yellowish green to yellow to sorrel and finally red (Smith 1971). In addition, examination of the phloem and sapwood around the galleries might



Figure 20: Pitch tubes on ponderosa pine resulting from attack by red turpentine beetle.

show a blue stain because of a fungus brought in by the beetle (Boldt and Van Deusen 1974).

Control—As with any insect or pathogen, the best defense against attack is to keep the trees growing vigorously. Limiting root or stem injuries will aid in protecting the trees from red turpentine beetle attack (Boldt and Van Deusen 1974). In addition, minimizing logging activities in stands adjacent to recently cut stands, which might harbor red turpentine beetles, can limit the number of red turpentine beetle attacks (Schmid and Mata 1991). Finally, avoid piling green logs or timber near live trees since this might attract red turpentine beetles and other insects to the standing trees (Boldt and Van Deusen 1974).

The use of semiochemicals to control red turpentine beetles is in its infancy. Red turpentine beetles are known to be attracted to three chiral monoterpenes, (S)-(-)-beta-pinene, (S)-(+)-3-carene, and (R)-(+)-alpha-pinene, emitted by stressed ponderosa pine trees (Hobson and others 1993). To date, no evident aggregation pheromone has been identified for red turpentine

beetles, and it has been suggested that red turpentine beetles rely solely on the tree's monoterpenes (Borden 1982; Hobson and others 1993).

Other modes of protection include chemical and biological agents. The chemicals 4 percent carbaryl (Sevimol 4), and 2 percent carbaryl (Sevin XLR) were shown to be effective in protecting trees from red turpentine beetles (Hall 1984). Chlorpyrifos and permethrin were shown not to be effective in protecting trees from red turpentine beetles (Hall 1984). Biological control of red turpentine beetles occurs via competition with other beetles and from woodpecker predation (Smith 1971).

Pine tip moth (*Rhyacionia* spp.)

Two species of pine tip moth reside in the Black Hills, the western pine tip moth (*Rhyacionia bushnelli*) and the southwestern pine tip moth (*Rhyacionia neomexicana*) (Furniss and Carolin 1977). Both moths attack small trees (less than 10 feet [<3 m] tall) and are not considered serious pests (Boldt and Van Deusen 1974). However, repeated attacks on regeneration can seriously slow down a revegetation project.

Adult western pine tip adult moths have yellowish gray to reddish brown wings that span 0.39 to 0.59 inch (1 to 1.5 cm) and have gray scales covering the head, body, and appendages (Furniss and Carolin 1977). The moths fly in late May and early June to lay eggs on pine needles, buds, and shoots of ponderosa pine seedlings. The larvae of the western pine tip moth, which are yellow with black heads and about 0.35 to 0.47 inch (0.9 to 1.2 cm) long, feed on the tissues during June and July (Furniss and Carolin 1977). The moth pupates over the winter in the litter or soil.

The southwestern pine tip moth is larger than the western pine tip moth, with a wingspan of 0.79 to 1 inch (2.0 to 2.5 cm) (Furniss and Carolin 1977). This moth has front wings that are dark gray with the outer third having a reddish-orange color with two horizontal black lines. The moths fly in late May and early June (Furniss and Carolin 1977).

Peak flight activity occurs at night when males are attracted to females by a sex pheromone (Jennings and Stevens 1982). The southwestern pine tip moth lays eggs on 1-year-old needles in the upper three whorls of the foliage. The eggs are light greenish yellow and as they develop change to yellowish orange and then to bright reddish orange. The larvae have five instar stages and when fully developed are 0.47 to 0.63 inch (1.2 to 1.6 cm) long (Jennings and Stevens 1982). Through the instar stages, the larvae change from orange to yellowish orange to reddish orange. The pupal stage starts in July and early August, and the pupae

overwinter in bark crevices on the base of the tree below the litter (Furniss and Carolin 1977).

Ponderosa pine seedlings less than 6 to 8 feet (less than 1.8 to 2.4 m) tall are most susceptible (Jennings and Stevens 1982). In an Arizona study, ponderosa pine trees greater than 20 years old or taller than 13 feet (4 m) were not attacked by the moths; instead, attacks were highest in the 5- to 7-year age class (Spiegel and Price 1996). Both species of moths bore into new expanding shoots and feed on the tender tissues of the needles (Dix and Jennings 1982; Jennings and Stevens 1982). An early indication of infestation is the shiny pitch “tents” formed by the larvae in May and June (Jennings and Stevens 1982). By midsummer, infested shoots turn a reddish brown and become dry and brittle. Both terminal and lateral shoots are attacked (Jennings and Stevens 1982; Lessard and Jennings 1976), but damage is usually greatest on the terminal bud (Dix and Jennings 1982). New buds form below the damaged terminal bud, resulting in multiple stems, crooks, forks, spiketops, and a bushy appearance (Jennings and Stevens 1982; Lessard and Jennings 1976). Although trees are seldom killed by tip moths, repeated infestations can jeopardize the survival and growth of young planted seedlings (Jennings and Stevens 1982).

Special care should be taken when revegetating a site after a large-scale fire. Lessard and Jennings (1976) reported that small trees in large, open areas seem most vulnerable to persistent infestations. For instance, after a 1956 fire in the Apache-Sitgreaves National Forest, several large areas were naturally reseeded or replanted with seedlings; after 10 years, 38 percent of the seedlings were damaged by tip moths. By 1969, 82 percent of the seedlings were damaged.

Control—Although growth loss is not considered serious enough to justify the cost of direct control, several methods can be used to mitigate pine tip moth damage. First, silvicultural treatments that promote rapid tree growth, such as thinning overstocked “dog-hair” stands, may help reduce pine tip moth damage (Jennings and Stevens 1982). Although not practical for large areas, clipping infested shoots before the larvae leave the shoots can reduce pine tip moth populations. Finally, relying on natural predators such as mites, ants, spiders, lizards, birds, mice, and parasitic wasps can help control the pine tip moth population (Jennings and Stevens 1982). Hand spraying of commercial insecticides labeled for shoot moth control can be effective in landscape plantings of ponderosa pine if applied during shoot elongation and 2 to 3 weeks later.

Diseases

Armillaria root disease (*Armillaria ostoyae*)

Armillaria root disease is a native pathogen in the Black Hills (Boldt and Van Deusen 1974; Holah 1993; Lundquist 1991). Until the late 1970s, it was assumed that Armillaria root disease was caused by a single polymorphic species, *Armillaria mellea* (Vayl. Ex Fr.) Kummer, but now several species are recognized within the genus (Watling and others 1991). In the previous edition of this publication, Boldt and Van Deusen (1974) described the shoestring root rot, which they identified as *Armillaria mellea*. It is now known that only *Armillaria ostoyae* exists in the Black Hills (Kallas 1997; Wu and others 1996).

As a natural component of the forest, the fungus causes mortality, wood decay, and growth reduction. Armillaria root disease infects both healthy and stressed trees, and kills the trees either directly or indirectly by predisposing the trees to attacks by other pathogens or insects (Williams and others 1986). This parasitic fungus attacks the roots and root crown of trees of all ages, killing the cambium and inner bark and causing decay of both sapwood and heartwood (Smith 1993). Direct mortality in ponderosa pine seems to occur by the combination of destruction of living root cambial tissue by the fungus and girdling resinous lesions produced by the tree (Holah 1993).

Armillaria root disease has been reported as a potentially important disease of ponderosa pine in the Black Hills since the 1930s (Boldt and Van Deusen 1974; Holah 1993; Lundquist 1991). Several studies (Hinds and others 1984; Holah 1993; Kallas 1997; Lessard and others 1985; Lundquist 1991) have reported a wide distribution of armillaria root disease throughout the Black Hills. In general, the incidence of armillaria root disease is greater in the northern Black Hills than in the southern Black Hills (Holah 1993; Lessard and others 1985; Lundquist 1991). Several hypotheses have been proposed for this difference in distribution: different soil types, higher frequency of fire in the southern hills, differences in tree physiology, and historically sparser forests in the southern hills prohibiting the spread of the fungus (Holah 1993).

Areas affected by armillaria root disease are characterized by patchy openings containing fallen trees, patches of dead and dying trees that are broken at the root collar, and trees with poor crown conditions (Holah 1993; Williams and others 1986). Armillaria root disease kills trees in a pattern of gradually expanding disease centers (Williams and others 1986). These centers can occur in managed or unmanaged areas and vary in size from several trees up to 1,000 acres (404.7 ha).

Armillaria root disease centers can be detected from aerial photographs by searching for openings in the forest canopy that have dead or nearly dead standing trees on the margins, or by looking for snags and windthrown conifers (Williams and Leaphart 1978). These centers are sometimes large enough to appear as circular openings surrounded by green timber (Roth and others 1980).

The crown condition of a tree may suggest armillaria root disease infection. Typical characteristic symptoms include red or chlorotic needles (Lundquist 2000; Williams and others 1986), stunted internodal growth of recent lateral and terminal shoots, reduced growth of shoot and foliage on older trees (Williams and others 1986), distressed crop of cones (Lundquist 2000), and branch dieback (Williams and others 1986). On large, lightly infected or vigorous trees, crown symptoms typically occur over a number of years (Williams and others 1986). However, on small, extensively infected or low-vigor trees, the crown symptoms develop rapidly and the tree often dies within a year. It should be noted, however, that the crown symptoms reported for armillaria root disease infection are similar to symptoms caused by drought, bark beetles, and other fungi. For this reason, further verification will be needed to correctly diagnosis the problem.

Examination of the lower stem can aid in the correct diagnosis of armillaria root disease. Infected areas of the lower stem are typically enlarged and secrete large amounts of resin (Lundquist 2000; Williams and others 1986). Mycelial mats and rhizomorphs are usually present beneath the bark. Mycelial mats are white, irregular, fanlike striations that decompose the inner bark (fig. 21) and leave resin-impregnated impressions (Williams and others 1986). These impressions can occur for several years after the fans disappear (Morrison and others 1991). Rhizomorphs are flat, black to reddish brown rootlike features that are up to 0.2 inch (0.5 cm) wide (Williams and others 1986).

Diagnosis by examining the soil is more difficult. Rhizomorphs do occur in the soil, but are half as wide as those found in the bark (Williams and others 1986). Mushrooms, the reproductive stage of armillaria root disease, usually occur in clusters around the base of the infected tree or stumps, but are typically short lived. The mushrooms are sporadic in late summer and autumn and are most abundant in moist periods (Williams and others 1986).

The majority of armillaria root disease infection in green trees occurs via rhizomorphs and not spores, thus limiting its ability to infect large areas quickly (Roth and others 1980). Infection typically occurs when

rhizomorphs in the soil encounter uninfected roots, or when uninfected roots are exposed to infected roots (Williams and others 1986). Rhizomorphs can grow up to 10 feet (3 m) through the upper soil layers (Williams and others 1986), but *Armillaria ostoyae* typically forms fewer rhizomorphs and spreads more via root contacts (Redfern and Filip 1991). Root contact infections are quite common in dense stands where root contact is frequent (Williams and others 1986).

Armillaria root disease has a long life span. The inoculum of armillaria root disease can survive for several decades in both broad-leaved and conifer stumps (Redfern and Filip 1991) and coarse woody debris (Williams and others 1986). Shaw (1975) reported that wood cut from 30-year-old ponderosa pine stumps contained viable armillaria root disease. Roth and others (1980) reported the occurrence of armillaria root disease in large, old growth stumps that were over 35 years old.

The ability of armillaria root disease inoculum to reside in stumps for several decades increases the opportunity for infection under silvicultural treatments such as clear felling, selective cutting, and thinning (Morrison and others 2001; Shaw and others 1976). Several studies show an increase in the frequency of armillaria root disease and mortality in managed stands compared to undisturbed stands (Filip 1977; Hagle and Goheen 1988; Shaw and others 1976). Physical disturbance of soil by logging or scarifying may split rhizomorphs, causing them to initiate new growing tips



Figure 21: Characteristic Armillaria white mycelial fan under the bark of an infected ponderosa pine tree (source: Williams and others 1986).

and increase likelihood of new infections (Redfern 1973). Kile and others (1991) noted that the spread of armillaria root disease was more dependent on the frequency of cutting than the intensity of the operation. On the Northern Hills Ranger District (Spearfish/Nemo Ranger District) in the Black Hills, Holah (1993) noted that stands with more recent harvest activity had over twice as much acreage (10 percent versus 5 percent) occupied by armillaria root disease as stands that had not been logged for at least 5 years, although the number of root disease centers was similar. Thus, management activities that create stumps might increase the potential for armillaria root disease to spread, but may not necessarily increase mortality.

A major concern that managers should be aware of is the risk that natural regeneration, which does not have the vigor to resist the infection, might become infected by the residual inoculum. Stumps are food bases for armillaria root disease (Wargo and Harrington 1991) and can limit the ability for a site to revegetate due to reinfection. Shaw and others (1976) reported that mortality of small trees was first noticed around stumps at a dry site 4 to 6 years after harvest.

Trees with reduced vigor are most susceptible to armillaria root disease (Wargo and Harrington 1991). Thinning operations increase the growth and vigor of residual trees, thus lowering the susceptibility to disease. Vigorous trees can limit armillaria root disease in localized lesions and curtail the spread of the disease by secreting resin and forming callus tissues (Williams and others 1986). However, trees are typically stressed upon the initial release and might become infected before the trees can respond to the benefits of the release (Wargo and Harrington 1991). Yet, in central Oregon, Filip and others (1989) reported that an early precommercial thinning on a moderately poor ponderosa pine site did not increase armillaria root disease infection.

Several studies have suggested an association between armillaria root disease and mountain pine beetle (Cobb 1989; Cobb and others 1974; Eckberg and others 1994; Hinds and others 1984; Lessard and others 1985). Hinds and others (1984) surveyed over 115 trees on 40 plots in the northern Black Hills and found 28 percent of trees infested with mountain pine beetle were also infected with armillaria root disease. In addition, 75 percent of the dead trees were infected with armillaria root disease, while none of the vigorous trees showed signs of infection. Holah (1993) reported a high probability (67 to 78 percent) of being in or near an armillaria root disease center when an area of mountain pine beetle was located. Furthermore, the probability of finding a beetle-killed tree or pitch-out tree

when armillaria root disease was present was 43 to 48 percent. Her survey data suggest that many armillaria root disease centers are not associated with mountain pine beetle, but most areas with mountain pine beetle contain root disease (Holah 1993).

Several hypotheses have been suggested for the association of armillaria root disease and mountain pine beetle. First, armillaria root disease stresses a tree and predisposes it to beetle attack, colonization, and death (Cobb 1989; Cobb and others 1974; Hinds and others 1984; Lessard and others 1985). Eckberg and others (1994) suggest that when armillaria root disease is present in stands without an established root disease center, it infects the roots of a small percentage of the trees within the stand. These trees then become primary focus trees for the mountain pine beetle. If the mountain pine beetle population is large enough, trees surrounding the primary focus tree are also attacked and an added stress is placed on the trees without armillaria root disease. Once these trees start combating the mountain pine beetle, defenses against armillaria root disease infection decrease and the probability of infection increases (Eckberg and others 1994).

Holah (1993) made an excellent point when discussing the association of mountain pine beetle and armillaria root disease: "Areas impacted by Armillaria appear to have been frequently confused with areas infested by mountain pine beetle, probably because they often occur in the same area and patches of Armillaria-caused mortality can look like beetle-caused mortality to the untrained eye." However, it is extremely important to recognize beetle frass, pitch tubes, and fungal mycelium or fruiting bodies to distinguish the two because management activities will affect the two agents differently. Furthermore, managers should be aware that Armillaria is a persistent agent that will kill trees over extended periods of time, whereas beetle-caused mortality is generally concentrated over short periods of time.

Control—Eradication of armillaria root disease is not a realistic option, but management that limits the buildup of the disease and reduces its impact is feasible (Williams and others 1986). As with any disease, maintaining vigorous tree growth and minimizing stresses such as wounding the tree bole can limit the development of armillaria root disease.

Another effective technique is to reduce the food source available to armillaria root disease by uprooting infected or susceptible root systems and stumps (Williams and others 1986). However, we do not recommend removal of all stumps because of the risk of reducing soil biota and site productivity (Graham

and others 1994). The health of young trees within a stand is also a useful indicator of armillaria root disease (Roth and others 1980). If there are groups of dead trees of pole size or larger located around old stumps, then armillaria root disease is present in the stand (Roth and others 1980). Removal of old stumps in armillaria disease epicenters can reduce infection.

Instead of removing all old stumps in an area, creation of root-free zones between diseased and healthy stands can be an effective method to contain armillaria root disease (Roth and others 1977). The mode of action differs depending on the type of infected stand. Armillaria root disease typically advances sporadically in waves along short fronts outward from infected stumps at an average rate of 3 feet (0.9 m) per year. In dense stands of young trees where old-growth stumps are common, establishment of a root-free area to serve as a barrier can be effective (Roth and others 1977). Keep in mind that treatment of the actual infected area by uprooting maybe expensive, detrimental to soil productivity, and ineffective.

Stands that have scattered infections require a different approach to contain armillaria root disease. If the mortality in dense stands is limited to small trees of low vigor and the stand is free of old stumps, then treatment is not required because the armillaria root disease is in an endemic state (Roth and others 1977). However, poorly stocked stands with infected old stumps do require some sanitation treatment to maintain the crop trees. Stumps and roots should be removed in an area extending no less than 33 feet (10 m) past the obvious boundary of the disease center because the root systems are most likely infected (Roth and others 1980; Williams and others 1986). Refer to Roth and others (1977) for marking and removal methods.

Prescribed fire may directly affect armillaria root disease activity in forests through the destruction of inoculum or indirectly through stress effects on the fungal mycelium (Reaves and others 1993). Kile and others (1991) suggested that high-intensity fires could directly destroy inoculum present in stumps as well as reduce rhizomorphs. However, Filip and Yang-Erve (1997) reported that armillaria root disease was significantly reduced following a fall burn in Oregon only at soil depths up to 3.2 inches (8 cm), but not at 11.8 inches (30 cm). This poses a problem because most of the woody roots that are naturally infected with armillaria root disease are at depths greater than 3.2 inches (8 cm). The authors suggest, however, that if the fire was intense enough or more penetrating, it could decrease armillaria root disease infection (Filip and Yang-Erve 1997). Probably the best use of prescribed fire to control armillaria root disease is to use

fire to increase the vigor of the trees instead of trying to destroy the fungus.

The use of chemicals to control armillaria root disease is only recommended in situations where high-value trees, such as in campgrounds, need protection. Chemical fumigants such as chloropicrin, methyl bromide, and carbon disulfide can reduce infection levels (Williams and others 1986). These fumigants should be applied around the base of the infected stem or placed in holes where trees have been uprooted to prevent the spread of armillaria root disease. Filip and Roth (1987) list several chemicals that failed to protect ponderosa pine from armillaria root disease in central Washington.

As with any native pathogen that attacks ponderosa pine in the Black Hills, managers need to realize that this is a natural process and part of the checks and balances of the ecosystem. The presence of armillaria root disease in the Black Hills does have some benefits. For instance, large-scale gaps formed in the forest create a variety of new habitats. The structural complexity of this landscape created by armillaria root disease and other disturbance factors provides a variety of plants that benefit wildlife. Furthermore, habitats for cavity-nesting birds are created as well as coarse woody debris habitat for small animals. In addition, since armillaria root disease has a low pathogenicity in hardwoods (Kallas 1997), regeneration of these trees is encouraged (Holah 1993), increasing the diversity of the landscape.

Red rot (*Dichomitus squalens*)

Red rot is caused by the fungus *Dichomitus squalens*, which produces a white-pocket rot in ponderosa pine. It is one of the major causes for loss of sound wood in commercial stands (Andrews 1971; Boldt and Van Deusen 1974). It has been estimated that up to one-fifth of the gross volume cut in virgin sawtimber in the Black Hills has been culled due to red rot (Andrews 1971). Red rot can infect pines of all age classes except seedlings and small saplings, thus it is problematic in second-growth stands because once it infects a young tree, it has several decades, depending on the rotation age, to develop and cause damage (Andrews 1971).

The life cycle of the red rot fungus starts on decaying dead material that is in close contact with the ground (Andrews 1971). The fruits are flat and white and typically appear 4 years after the material is infected. For the next 6 years, the sporophores develop annually on the decaying material during the summer rainy season. The combination of summer high temperatures and moist conditions favor fruiting, spore germination, and vegetative development of the red rot fungus (Andrews 1971).

The spores of the red rot fungus are wind disseminated and become established in living trees through the dead wood on branches or insect holes (Andrews 1971; Boldt and Van Deusen 1974). If the moisture regime is favorable, the spores will germinate immediately and develop a white, feltlike mycelial pad between the bark and wood. From the mycelial pad the fungus invades the wood of the branches and starts growing toward the bole through both wood and pith of the branch (Andrews 1971; Boldt and Van Deusen 1974). The red rot eventually will reach the heartwood of the trunk and cause the most damage.

Red rot fungus has two distinct stages: incipient and advanced (Andrews 1971). Both of these stages can be viewed in a board sawed from a decayed log. The incipient stage is identified by the reddish-brown discoloration of the infected wood, but with no change in structural integrity (fig. 22a). At the end of an infected log, incipient decay is often fan shaped or radiates out from the center of the log like spokes on a wagon wheel (fig. 22b) (Andrews 1971).

In the advanced stages of decay, the incipient stage typically surrounds the more decayed areas of the wood. The advanced stage is often poorly defined white pockets of discolored wood (fig. 22c). The strength and structural integrity of the wood are reduced, and long axes parallel to the grain appear to be filled with lintlike material (Andrews 1971). As the decay advances, the white pockets become more abundant until they join and give the infected wood the appearance of a fibrous white mass. In the end stages of decay, the white lintlike substance disappears and a bleached, grayish-brown, stringy decayed wood is left. Eventually, the stringy decayed wood will fall away and create a cavity in the wood (Andrews 1971).

Red rot is most damaging in wood products that are sawn into boards, where rotten portions of logs have to be discarded. It is less of a problem when logs are utilized for whole fiber products such as chips, particle board, or tongue-and-groove products. In these cases rot can effectively be removed in the manufacturing process. Utilization of red-rot-infested trees is limited by the economics of harvesting and transporting heavily infested trees with little recovery potential.

Red rot has some positive effects on forest ecosystems. The formation of cavities in the wood provides excellent habitat for the many cavity-nesting birds in the Black Hills. Furthermore, the presence of red rot in a forest adds structural heterogeneity in the form of small gaps within the forested landscape. Rot contributes to breakdown of woody debris, reduction of fuel loadings, and provides habitat for soil flora and fauna.

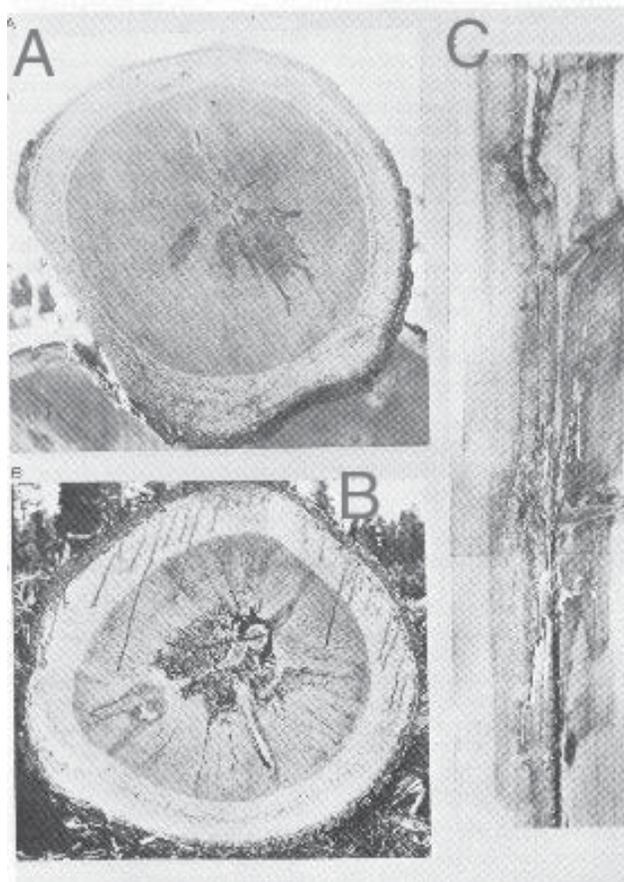


Figure 22: Example of various stages of red rot in ponderosa pine. (A) incipient decay at the end of a saw log; (B) advanced decay at the end of a saw log; (C) advanced decay bordered by incipient decay in the heartwood of a board (photo source: Andrews 1971).

Red rot in mature standing trees is often difficult to detect because the saprophytes rarely develop on living trees. In mature trees over 60 years old, examination of bark-covered branch stubs and branch debris on the ground at the base of the tree can give vital information on the extent of the infection (Andrews 1971). A branch must be pruned to determine if the fungus has reached the heartwood.

To determine if an immature tree is infected with red rot, try to break a branch close to the trunk (Andrews 1971). If the branch resists breakage or breaks with a crack, then the tree probably is not infected. If there is little resistance, however, and the branch breaks as if water logged, then decay is in an advanced stage. To determine if the red rot has reached the heartwood, prune the branch flush to the trunk and look for the decay.

Branches that have deadwood and retain their bark are primary infection vectors (Andrews 1971; Boldt and Van Deusen 1974). Trees that have large, dead, bark-covered branches are likely candidates for red rot

infection (Boldt and Van Deusen 1974). Infections in trees less than 20 years old are rare because few of these trees have dead branches large enough to be susceptible (Andrews 1971). Typically, infections are more related to the size of a dead branch than to the age of a tree. Since older trees typically have more dead branches, infections are usually more prevalent in older trees.

Control—There is no chemical control for red rot. The best method of controlling red rot in a forested stand is through silvicultural treatments. Pre-commercial thinnings should avoid small-branched trees and cut large-branched trees that are already infected or at risk for infection (Andrews 1971). In addition, intermediate cuts to improve volume growth rates should focus on the large-branched trees that are already infected or have the potential for infection (Boldt and Van Deusen 1974).

One of the best cultural treatments is to prune stands where average diameter is still small (less than 7 inches; <18 cm) (Boldt and Van Deusen 1974). Pruning eliminates future entrance points and branch infections, and can actually inactivate the red rot fungus in the decay columns (Andrews 1971). Removal of all branches on the butt log at the same time is desirable, but pruning may need to be conducted over several operations (Andrews 1971; Boldt and Van Deusen 1974). The earlier pruning is performed, the greater the protection from red rot infection (Boldt and Van Deusen 1974). Furthermore, pruning stubs are not subject to red rot infection and can increase the value of timber.

Western gall rust (*Peridermium harknessii*)

Western gall rust is caused by a fungus that forms galls on branches and stems of ponderosa pines in the Black Hills. Historically, the western gall rust was designated the scientific name *Endocronartium harknessii*, but recent studies have found *Endocronartium* to be incorrect and *Peridermium harknessii* is the accepted designation (Johnson 2000). Western gall rust is the second most troublesome disease of ponderosa pines (Boldt and Van Deusen 1974). Unlike other rusts, this fungus spreads directly from pine to pine without an alternate host (Johnson 2000). The fungus reduces tree growth, affects tree form, and causes tree mortality. It infects trees of all ages and sizes, but rarely kills an entire stand (Boldt and Van Deusen 1974; Johnson 2000).

Seedlings are quite susceptible to the gall rust because it girdles the stem with galls and can kill within a few years (Johnson 2000). Moreover, infections are most serious in plantations where mortality of young trees is common (Boldt and Van Deusen 1974). Unlike

most diseases that kill the weakest of the population, the gall rust infects vigorous trees more readily. The greater surface area of new shoots on the faster growing trees provides greater chance of infection (Boldt and Van Deusen 1974).

Western gall rust is an obligate parasite that grows perennially in the inner bark and xylem of the ponderosa pine host. This parasite reproduces using two types of spores, spermatia, which are rarely seen, and aeciospores (Johnson 2000). Aeciospores are produced under bark scales in whitish sacs called peridia in late spring or early summer the second or third year after infection. New spores are produced annually until the pine tissue dies, which can be up to 200 years. During periods of high humidity, the peridia rupture and expose masses of orange powdery spores, which are then dispersed by wind. Once the spores land on susceptible tissue, they need rainfall to germinate and cause a new infection (Johnson 2000).

Infections occur on twigs of the current year, branches, and stems (Boldt and Van Deusen 1974). It is difficult to determine if a tree has been infected during the first growing season of exposure. However, during the summer of the second year of infection, galls will develop on the infected area (fig. 23) and spores will be produced from the galls the following spring (Johnson 2000). Once the rust invades the pine bark, the cambial cells begin to divide rapidly resulting in a woody gall. The gall is soft at first, but as resins solidify, the gall becomes harder and decay resistant (Boldt and Van Deusen 1974).

The formation of galls on tree stems is the most damaging impact of western gall rust. This type of gall is often referred to as a canker (fig. 24) and can result in growth loss and cull. Cankers not only form weak points in the tree stem and increase the probability of wind breakage, but also create opportunities for other decay fungi to enter the tree stem (Johnson 2000). Stem cankers result in loss of merchantable length and mechanical weakness of lumber (Boldt and Van Deusen 1974). Gall formation on branches of mature trees is typically of slight importance, although it can girdle and kill a branch (Johnson 2000). Up to 100 galls may form on susceptible trees and reduce growth of the tree (Johnson 2000).

Control—There is no chemical control for western gall rust. Removal of infected trees during stand entries for thinning, timber stand improvement, and sanitation is the only practical method of reducing gall rust damage (Johnson 2000). Although pruning has been suggested to reduce western gall rust (Boldt and Van Deusen 1974), it does not seem practical (Johnson 2000). The presence of a few branch galls is not



Figure 23: Western gall rust canker on a branch in a ponderosa pine plantation in the Black Hills Experimental Forest.



Figure 24: Western gall rust canker on the bole of ponderosa pine in the Black Hills.

sufficient to cut a tree, but if a tree has a stem canker or branch gall within a few inches of the main stem it should be felled (Johnson 2000). If a main-stem canker encircles more than one-third of the circumference of a tree in a high-use recreational area, the tree should be removed to avoid potential injury (Johnson 2000). If the infected tree is located in a timber production area, compensation for future losses can be achieved by leaving a surplus of trees of precommercial thinning age. However, doing so on a large scale can potentially affect future growth and susceptibility to other pathogens.

Needle cast (*Elytroderma deformans*)

Needle cast is a fungus that defoliates ponderosa pine trees (Boldt and Van Deusen 1974). The fungus is widespread in the Black Hills, and the damage intensity varies among locations (Lundquist 1991). Needle cast typically occurs in areas where moisture is higher due to heavy dews and frequent fogs such as bottoms of draws and edges of meadows (Childs and others 1971). Infections are usually more severe in shaded thickets of reproduction, interiors of pole-sized stands, and lower crowns of pole-sized and larger trees on north to northwest slopes (Boldt and Van Deusen 1974; Childs and others 1971).

The needle cast fungus infects the needles, branches, and bark of ponderosa pine (Childs and others 1971). Infected needles appear red brown in the spring. The infection starts on the needles with long, dark, slender fruiting bodies appearing on the outer surfaces of red needles. The fruiting bodies appear as faint lines on

the outer surfaces of the needles and are visible by late spring or early summer. In late summer to early fall, the spores mature and are disseminated when the needles are wet and the temperature is favorable (Childs and others 1971).

The infection of branches and bark is perennial because the fungus can live in the inner bark and does not require the favorable weather conditions for spore infections needed for needle infection (Childs and others 1971). Infected branches form globular witches' brooms which are more dense and round than those caused by dwarf mistletoe (*Arceuthobium campylopodum*). Infected bark typically has visible lesions on the surface (Childs and others 1971).

The damage caused by needle cast is not as severe as other diseases, but it does weaken a tree's defense against other agents such as bark beetles or root rot (Childs and others 1971). Needle cast fungus is most damaging to trees with poor crowns. If the infection of a mature tree is moderately severe, the crown will gradually become thin, ragged, or short and the vigor of the tree will decrease (Childs and others 1971).

Damage inflicted by the fungus is classified on the amount of area diseased. If one-fourth of the needles on a twig are infected then it is considered little or no injury (Childs and others 1971). If one-fourth to three-fourth of the twig has needle infection then the tree is considered stressed and is likely to be killed by another agent such as bark beetles. If more than three-fourth of the needles are infected, then the tree will most likely die due to defoliation.

Control—Maintaining a healthy, vigorous stand is the best defense against the needle cast fungus, since fungicides are shown not to protect ponderosa pine from infection (Lightle 1955). Surveys for needle cast infections should be made in spring or early summer. If needle cast is detected in a stand, no immediate action is necessary because serious damage will not occur for several years (Childs and others 1971). If the stand is due for a thinning, thin the trees with trunk infections and trees that are within 6 feet (1.8 m) of a flagged tree. If mature trees are infected, salvage them when mortality rates or bark beetle populations start to increase in the stand. If a stand is lightly infected, remove only the trees with more than one-fourth of the crown flagged or killed. If the stand is moderately to heavily infected, cut the stand drastically and only leave trees with very good crowns (Childs and others 1971).

Diplodia tip blight (*Sphaeropsis sapinea*)

Diplodia tip blight (*Sphaeropsis sapinea*), once known as *Diplodia pinea*, attacks ponderosa pine trees

in the Black Hills. The spores of the diplodia tip blight fungus develop in small, black fruiting bodies known as pycnidia (Peterson 1981). Pycnidia form on fascicle sheaths and bark, but are most numerous on the base of needles and on scales of second-year cones. Pycnidia can be easily identified by looking for the ashen-gray needles of previous year infections. The spores of this fungus are disseminated from March to October during rainy periods (Peterson 1981). To germinate and penetrate the needles and shoots, the spores need high relative humidities (Peterson 1981). If it is a dry year, infection levels are usually very low.

Infection of ponderosa pine trees develops on both the shoots and second-year cones (Peterson 1981). One of the first indicators of infection on shoots is the presence of resin droplets; another is brown, stunted new shoots still encased in fascicle sheaths. Entire shoots can be killed rapidly by the fungus and infection on major branches can vary. Damage generally occurs first in the lower crown and can be detected in late May. The extent of the damage can be assessed by late June or July (Peterson 1981).

Infections that occur on ponderosa pine cones typically happen in late May and are limited to second-year cones (Peterson 1981). Infections occur on both healthy and stressed ponderosa pine trees. On older pines, numerous fruiting bodies develop on the cones first before the new shoots are infested. This build up on the cones leads to greater damage on older trees (Peterson 1981).

Although trees of all ages are susceptible to diplodia tip blight, mature trees are most susceptible due to the greater surface area of the cones (Peterson 1981). In fact, open-grown trees and trees along the edges of stands are most susceptible due to the abundance of cones typically found on these trees (Johnson and others 1983, 1985). Susceptibility is also increased if tissues are wounded by insects or hail, especially during May or June when vulnerability is highest (Peterson 1981). New shoots are susceptible during a 2-week period starting when buds begin to open until about mid-June (Peterson 1981). Repeated infections of diplodia tip blight can reduce growth, deform trees, and eventually kill the tree (Peterson 1981). Johnson and others (1983) reported that trees that were infected for 5 years showed a 20 percent reduction in growth.

Control—Since stressed and weakened trees are most susceptible to diplodia tip blight, maintaining tree vigor is important to prevent infection. Fungicides can reduce infection of new shoots during the 2-week period that shoots are highly susceptible, however, it does not prevent infection of seed cones. Peterson (1981) suggests two applications of 4-4-50 Bordeaux mixture

(4 lb. (1.8 kg) copper sulfate, 4 lb. (1.8 kg) hydrated lime, and 50 gallons (189 liter) of water each week for the 2-week period. The pruning of infected branches will probably not reduce the spread of the disease (Peterson 1981).

Fire

Historically, fire was a keystone ecological process that shaped the composition and structure of plant communities in the Black Hills. Over the past 100 years, however, fire has been suppressed in the Black Hills due to a combination of circumstance and a misunderstanding of the significance of fire. Forest density has changed markedly in many Black Hills landscapes over the past 120 years (fig. 25) as a result of fire suppression. No longer should fire be viewed solely as a destructive force, but instead as another tool managers can use to manage a healthy forest. To use this tool effectively, however, we must first understand the historical frequency and intensity of these fires.

Ponderosa pine is a fire-adapted species. It has evolved to coexist and flourish with fire, particularly

frequent, low-intensity underburns. The thick, well-insulated bark of large trees enables them to resist damage from these fires (aside from the occasional basal scars that leave a record of the fire, which allows reconstruction of the fire history). In the past, periodic low-intensity fires consumed small seedlings, pruned lower branches from large trees, and consumed concentrations of woody fuels on the forest floor; continuous fuels did not exist to allow such fires to burn completely through a forest. The result was a mosaic of conditions ranging from openings to groups of young seedlings to clumps and groups of older trees, including large orange-barked patriachs. If (or when) large crown fires did occur, they probably did not completely consume all trees within a landscape, but left sources of seed for the eventual recolonization of the burned areas.

All over the Western United States, numerous studies are documenting historical fire regimes within ponderosa pine forests using dendrochronological methods. Most of these studies have been performed in the Southwest, where presettlement fires were relatively low-intensity surface fires that occurred on average

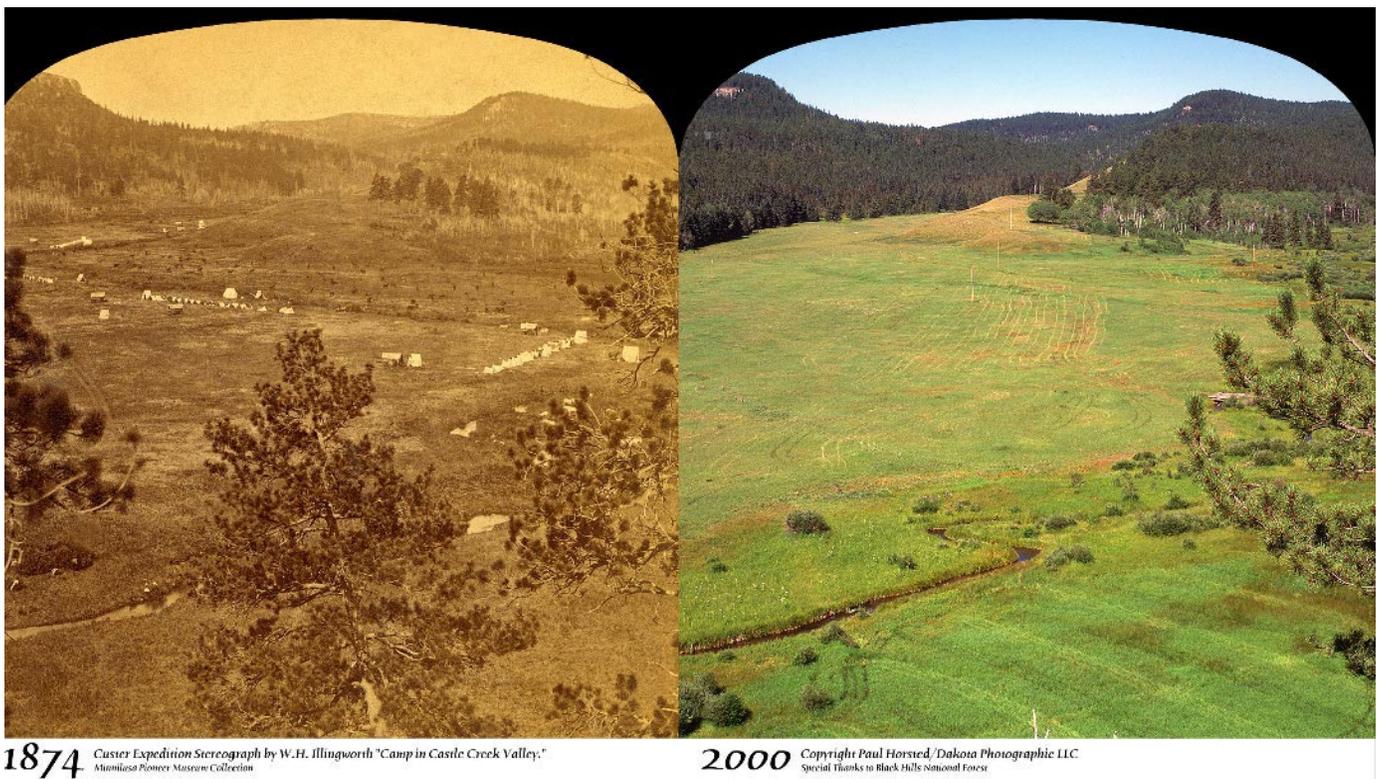


Figure 25: Increased density in ponderosa pine forests of the Black Hills region can be seen with this paired photograph taken by W. H. Illingworth in 1874 during Custer's Expedition and Paul Horsted in 2000. ©Paul Horsted/custertrail.com. Permission to use this photograph "Camp in Castle Creek Valley, 1874-2000" was graciously granted by Paul Horsted. The photograph is from a forthcoming book entitled "Exploring with Custer: The 1874 Black Hills Expedition," by Ernest Grafe and Paul Horsted, published by Golden Valley Press, Custer, SD, June 2002.

every 3 to 20 years (Brown and Sieg 1996). While understanding the historical fire regime in southwestern ponderosa pine forests is important, knowledge gained there must be adapted carefully to the Black Hills ecosystem.

To date, five studies have documented the historical fire regime of different regions within the Black Hills (fig. 26). Several factors influence fire regimes in forests, including: climate, elevation, topography, and fuel loads (Brown and others 2000). Therefore, it is important to look at fire regimes across the different regions. Most of these studies have taken place in National Monuments, National Parks, or wilderness areas within the Black Hills due to the availability of older trees for the dendrochronological analysis.

The study at the Devils Tower National Monument is representative of an area of open ponderosa pine forest and savanna with expansive areas of prairie northwest of the Black Hills proper (Fisher and others 1987). Annual average precipitation is 17.25 inches (43.8 cm) and elevation is 4,200 feet. (1280 m). Presettlement fire intervals were reported to be 27 years before 1770 and on average every 14 years between 1770 and 1900. After 1900, the frequency dropped to an average of 42 years due to fire suppression efforts and livestock grazing. The last reported area-wide fire for the Devils Towers area was in 1937 (Fisher and others 1987).

Three types of fires occurred at Devils Tower: spot fires, local fires, and area-wide fires. Spot fires, caused by lightning, were the most common and typically affected a single tree to a few trees. Low fuel loads or extinguishments by rainfall probably kept these fires from spreading (Fisher and others 1987). Lightning combined with conditions favoring the spread of fire probably ignited local fires. These fires covered a sizeable area and usually were extinguished by weather or low fuel loads (Fisher and others 1987). The largest fires were the area-wide fires, which typically burned throughout the area. The presettlement mosaic of fire frequency and different sized fires allowed the savanna-prairie boundary to remain in equilibrium. With the advent of fire suppression, however, dense forests have increased and prairie has decreased (Fisher and others 1987).

The study at the Jewel Cave National Monument in the south-central Black Hills is representative of the interior ponderosa pine forest (Brown and Sieg 1996). Elevations range from 5,200 to 5,800 feet (1,585 to 1,768 m) (Brown and Sieg 1996) with average annual precipitation approximately 18 to 19 inches (45.7 to 48.3 cm) (Driscoll and others 2000). Mean fire intervals in this area, calculated for the

years between 1388 and 1900 ranged from 20 to 24 years (fig. 26; table 2). These ranges are consistent with the mean fire intervals found by Fisher and others (1987) in their study of Devils Tower and with fire frequency data reported for ponderosa pine forests in the Northern Rocky Mountains (Arno 1976; Barrett and Arno 1982; Wright and Bailey 1982). Brown and Sieg (1996) also determined that the majority of fires were late growing season fires (July to August). These late-season fires coincide with the results from a study of historic lightning-ignited fires on the Northern Great Plains grasslands and pine savanna (Higgins 1984). Higgins also found that the majority (73 percent) of the 294 fires started by lightning occurred in July and August. The last recorded fire in the Jewel Cave area was a large-scale fire in 2000 due to arson (USDA Forest Service 2001a), but the previous fire was over 100 years ago in 1900 (table 2) (Brown and Sieg 1996).

The study at the Wind Cave National Park in the southeastern foothills of the Black Hills is representative of the ponderosa pine forest Northern Great Plains mixed-grass prairie ecotone (Brown and Sieg 1999). Elevations range from 3,600 to 5,020 feet (1,100 to 1,530 m) (Brown and Sieg 1999) and average annual precipitation is approximately 17 inches (43.2 cm) (Driscoll and others 2000). Season of fire was consistent with the Jewel Cave study, with fire scars indicating fires occurring later in the growing season or in the dormant season (Brown and Sieg 1999). However, mean fire intervals at Wind Cave sites ranged from 10 to 12 years (table 2), and were roughly twice as frequent as in the interior ponderosa pine stands of Jewel Cave (fig. 26; table 2). The authors suggest that the higher frequency of fire may be due to the lower elevation and warmer and drier environment of the Wind Cave area. This comparison of Wind Cave and Jewel Cave shows how differences in abiotic factors such as elevation and moisture can change a fire regime. The last extensive fire recorded in this area was over 90 years ago in 1910 (Brown and Sieg 1999).

The study at the Black Elk Wilderness Area in Upper Pine Creek Basin (Brown and others 2000) is representative of the high elevation (6,000 to 6,900 feet; 1,829 to 2,103 m) and rugged environment of the central Black Hills. The area is dominated by undisturbed forests of ponderosa pine, mixed with white spruce and aspen (Ryan and others 1994). Average annual precipitation is 20.8 inches (52.9 cm) (Ryan and others 1994). From 1580 to 1890, median fire interval was 22 to 23 years with a range 11 to 74 years for the area (fig. 26; table 2). The last fire recorded in this area was over 100 years ago in 1890 (Brown and others 2000).

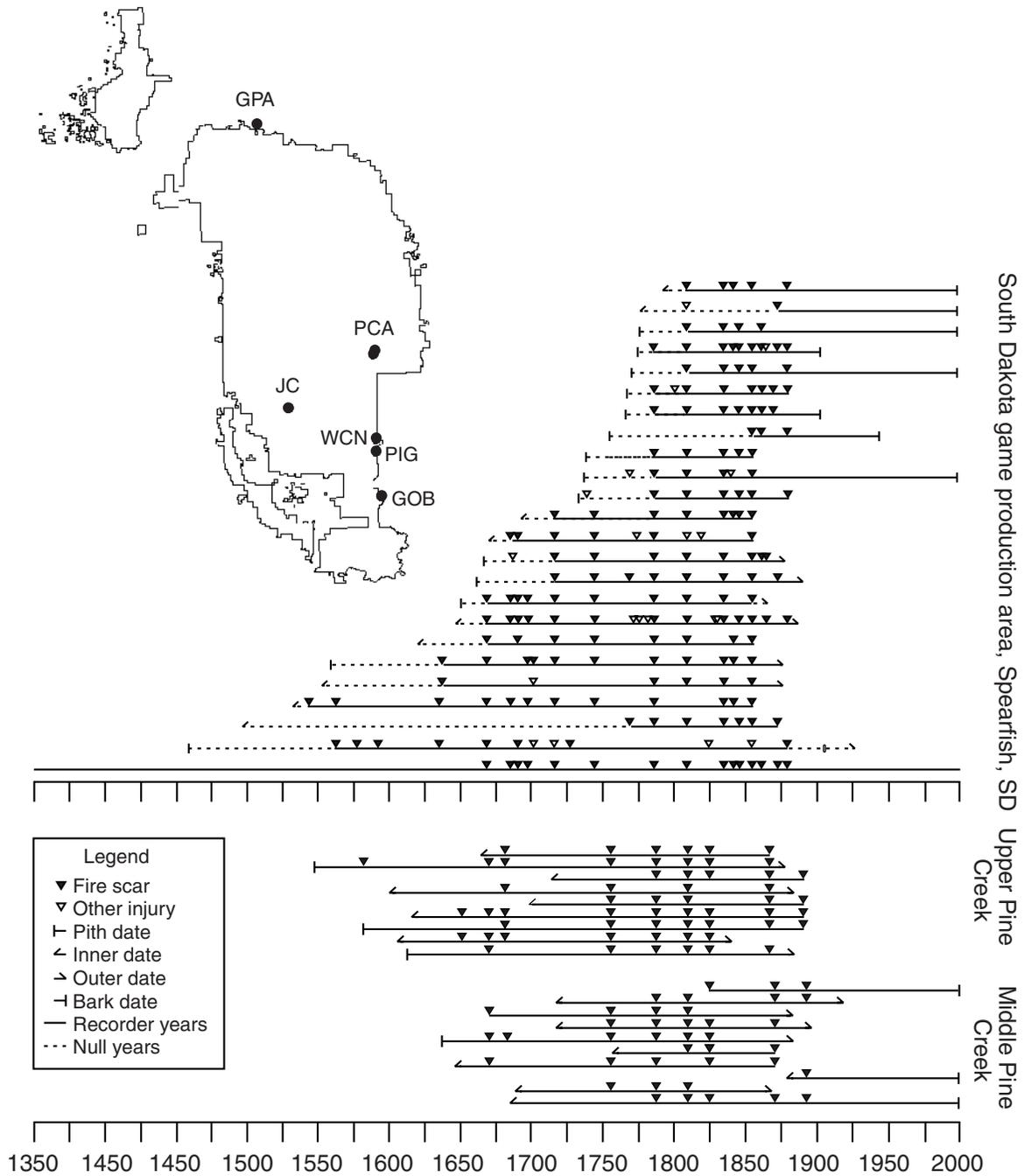


Figure 26: Locations and corresponding fire chronologies compiled from published fire histories of the Black Hills (Brown and Sieg 1996, 1999; Brown and others 2000; Wienk 2001). GPA = South Dakota game production area; PCA = Pine Creek Area; JC = Jewel Cave area; WCN = Wind Cave; PIG = Pigtail Bridge; GOB = Gobbler Ridge. Horizontal lines represent life span of individual trees. Fire scars are signified by triangles at the dates they were recorded. Open triangles are other injuries or questionable fire scars recorded within the ring series.

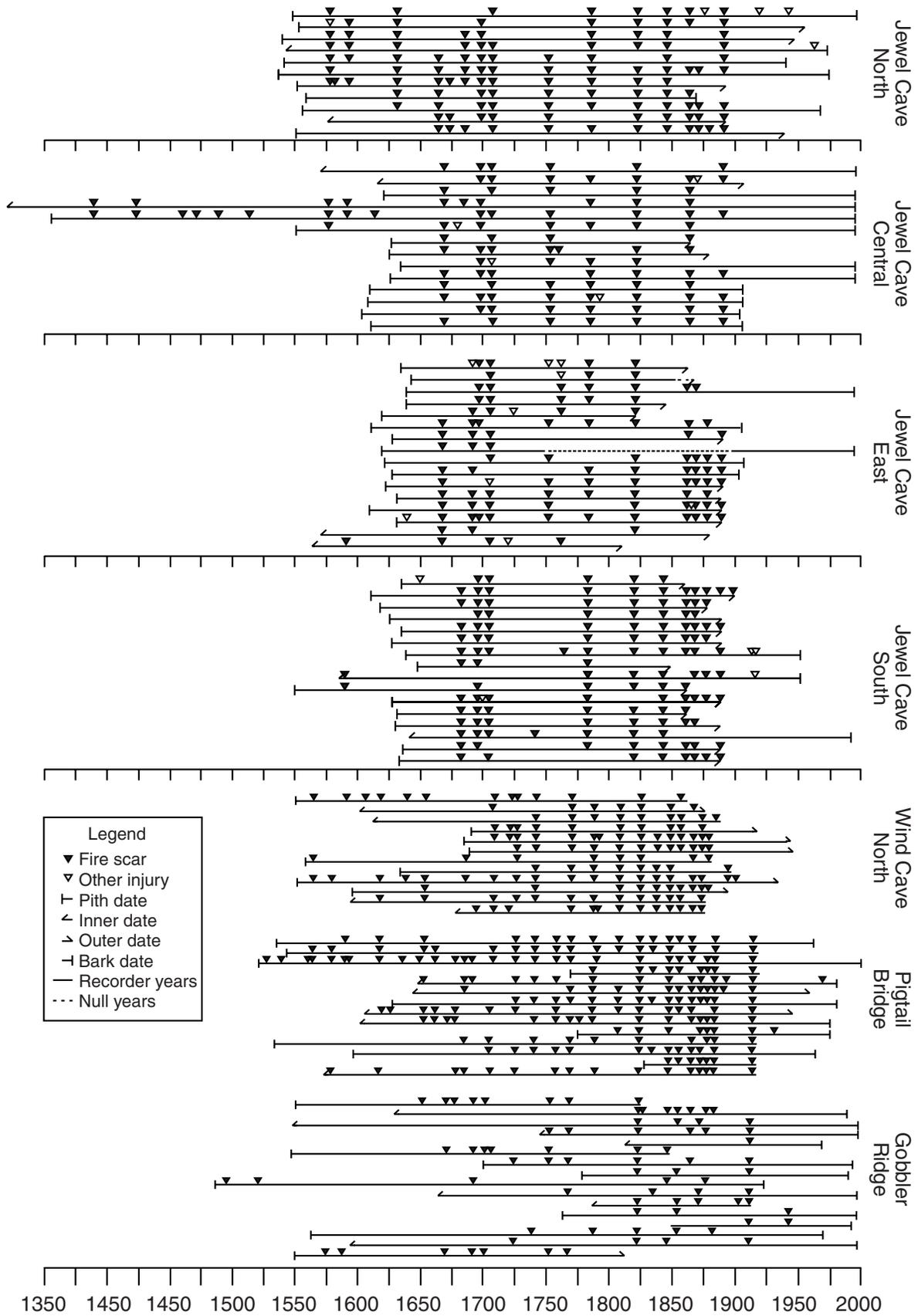


Table 2: Fire frequency, intervals, range, and time since last fire for 10 ponderosa pine sites in the Black Hills. Data compiled from published fire histories of the Black Hills (Brown and Sieg 1996, 1999; Brown and others 2000; Wienk 2001).

Location	Period of analysis	Number of fire intervals (years)	Mean fire interval (\pm SD)	Range of fire intervals (years)	Year since last fire ¹
Jewel Cave National Monument^a					
Jewel Cave North	1576 to 1890	16	19.6 \pm 13.5	4 to 45	104 (1890 to 1994) ²
Jewel Cave Central	1388 to 1890	22	22.8 \pm 17.6	1 to 63	104 (1890 to 1994) ²
Jewel Cave East	1591 to 1890	13	23.0 \pm 22.0	1 to 77	104 (1890 to 1994) ²
Jewel Cave South	1591 to 1900	13	23.8 \pm 23.1	7 to 93	94 (1900 to 1994) ²
Wind Cave National Park^b					
Wind Cave North	1564 to 1896	27	12.3 \pm 6.9	3 to 32	102 (1896 to 1998)
Pigtail Bridge	1528 to 1912	38	10.1 \pm 5.8	2 to 23	86 (1912 to 1998)
Gobbler Ridge	1652 to 1910	21	12.3 \pm 7.2	3 to 34	88 (1910 to 1998)
Black Elk Wilderness Area^c					
Upper Pine Creek	1580 to 1887	9	23 ³	11 to 74	110 (1887 to 1997)
Middle Pine Creek	1668 to 1890	7	22 ³	13 to 72	107 (1890 to 1997)
South Dakota Game, Fish, and Parks Area, Spearfish SD^d					
Badger GPA	1450 to 1998	26	13 \pm 10	1 to 43	119 (1879 to 1998)

¹ When studied was performed.

² Site was within the Jasper fire that occurred in 2000.

³ Median fire interval.

^a Brown and Sieg (1996).

^b Brown and Sieg (1999).

^c Brown and others (2000).

^d Wienk (2001).

The most recent study was performed in the northern Black Hills in Lawrence County, on the South Dakota Game, Fish and Parks Badger Game Production Area (GPA) west of Spearfish (Wienk 2001). Elevation of the site is 4,003 to 4,200 feet (1,220 to 1,280 m) and average annual precipitation is approximately 24 inches (61.0 cm) (Wienk 2001). The dominant vegetation is ponderosa pine. The majority of the fire scars were either late season or dormant season, which is consistent with the Jewel Cave and Wind Cave studies (Brown and Sieg 1996; 1999). Wienk's analysis of fire intervals from 1450 to 1998 yielded a mean fire interval of 13 years with a range of 1 to 43 years (table 2). This mean fire interval is similar to the interval reported for the foothills ponderosa pine stands of the Wind Cave study (fig. 26; table 2) (Brown and Sieg 1999). The last fire reported in the Badger GPA area was over 120 years ago in 1879 (Wienk 2001).

Examination of the dendrochronological data from these five sites within the Black Hills indicates that some fires were recorded on scarred trees within the same year (fig. 26), but it is impossible to ascertain if these fires were part of the same event or were separate events. Nevertheless, the evidence of fire acting

as part of the natural disturbance regime in the Black Hills is overwhelming. Shinneman and Baker (1997) proposed that both low-intensity and stand-replacing fires were significant components of this disturbance regime. Determining which type of fire occurred historically is not easy because fire scars detected by dendrochronology indicate only that a tree survived a fire. Whether the surviving tree was on the edge of a high-intensity fire or in the middle of a low-intensity ground fire is difficult to ascertain.

Inferences can be made about the occurrence of large-scale, stand-replacing fires by examining historical descriptions of early expeditions in the Black Hills, and by surveying the landscape for large areas of even-aged trees that have not been logged. Since the majority of the Black Hills has been logged within the past 100 years, we can only rely on historical descriptions. Shinneman and Baker (1997) examined several historical reports of the Black Hills and reported that many fires ranged from 7,500 to 22,000 acres (3,000 to 9,000 ha) in size, with a possibility of a very large fire of 50,000 acres (20,000 ha). Several observations by H. S. Graves in the 1899 forest inventory survey report indicate large-scale stand-replacing fires.

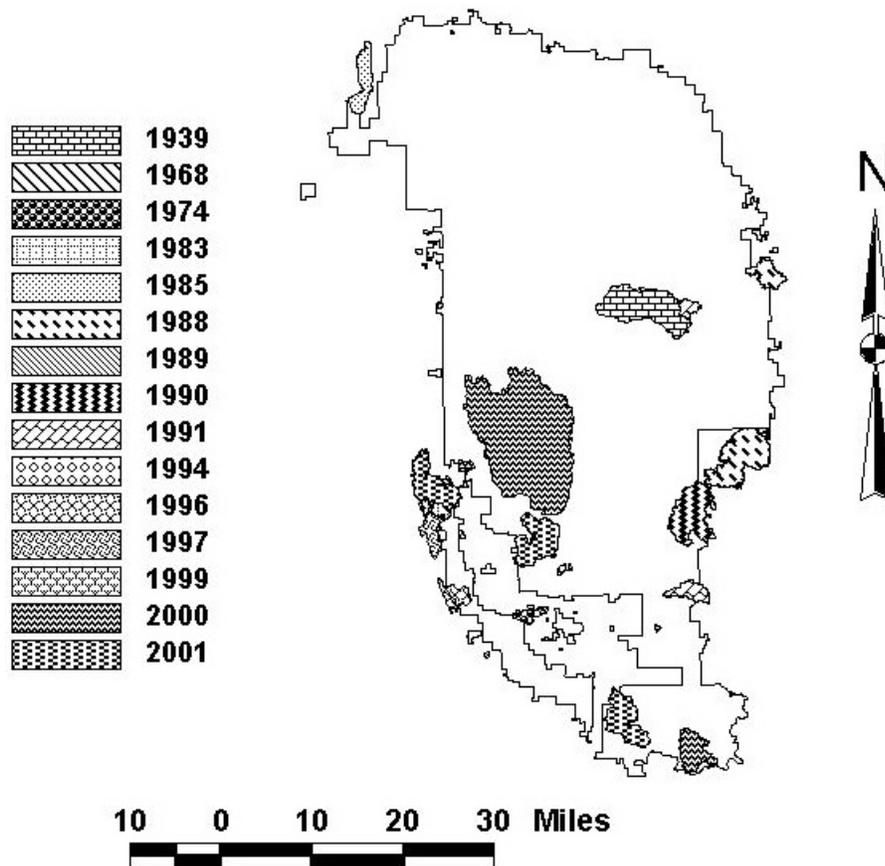


Figure 27: Some wildfires that have occurred in the Black Hills National Forest within the past 62 years. Data obtained from USDA Black Hills National Forest OnlineLinkage: [FTP://www.fs.fed.us/incoming/r2/bhnf/Core_Coverages/fire_history.zip](ftp://www.fs.fed.us/incoming/r2/bhnf/Core_Coverages/fire_history.zip)

For instance, Graves described a large forested patch of second-growth even-aged ponderosa pine approximately 100 years old on the limestone plateau (Graves 1899). In addition, Graves noted that patches of forest 100 to 110 years old could be found everywhere in the Black Hills. These stands could be a result of the 1785 fires recorded in the dendrochronological record (fig. 26) (Brown and Sieg 1996,1999; Brown and others 2000; Wienk 2001). Another even-aged forest stand around 190 to 200 years old documented by the Forest Service in the 1940s (USDA Forest Service 1948) can be traced to originate after a severe fire in the central Black Hills around 1730 to 1740 (fig.26) (Graves 1899; Progulske 1974; Raventon 1994). Evidence of a large-scale fire in 1842 (fig. 26) was confirmed by accounts reported by Native Americans that the “entire Black Hills were ablaze” around that time, and the presence of several clusters of 150-year-old trees in the northern Hills (Graves 1899; Raventon 1994; Shinneman and Baker 1997; USDA Forest Service 1948).

Within the past 100 years or so, large stand-replacing fires have occurred due to human and natural causes. In the 1890s, a severe 20,000-acre (8,100-ha) fire near Deadwood and Lead was thought to be started by settlers, and in 1898, a 20,000-acre (8,100-ha) fire near Crow Peak was caused by lightning (Progulske

1974). Throughout the 20th century (fig. 27), several fires burned areas from 7,400 to 22,000 acres (3,000 to 8,900 ha) (Raventon 1994). The most recent fire, known as the Jasper Fire, occurred in 2000 with approximately 83,500 acres (33,800 ha) burning in less than 6 days (USDA Forest Service 2001a). The intensity of this fire varied throughout the burned area, leaving pockets of living trees, which will certainly record this large-scale fire event for future generations.

Wind

Wind damage, common in the Black Hills (Lundquist 1995a; Raventon 1994), can range from a single tree to thousands of acres. Historical accounts of wind damage in the Black Hills indicate that tornados and chinooks can cause severe damage (USDA Forest Service 1948). Graves (1899) reported a large-scale windfall event in the late 19th century around Grizzly Creek. In May 1933, a tornado destroyed over 7,000 acres (2,800 ha) with a loss of 9 million board feet around the Rapid Creek area (USDA Forest Service 1948). In 1943, a chinook in the Black Hills region caused severe damage over an area 25 miles (40.2 km) long and 2 miles



Figure 28: A ponderosa pine snag topped by wind. This snag is still useful as a perch and for cavity nesters for roosts.

(3.2 km) wide (USDA Forest Service 1948). These events are only a few recorded instances within the past 100 years. More often, wind damage is restricted to individual stands or a small number of trees. Localized damage typically occurs in large pole and small sawtimber-sized stands (Boldt and Van Deusen 1974) and in mature to overmature stands during windstorms associated with heavy precipitation events (Alexander 1987a).

Wind damage on a smaller scale typically happens when a tree is already stressed by a predisposing agent such as suppression, fire, or shallow soil (Lundquist 1995a). The combination of the predisposing agent and the strong wind can result in a fallen tree, a broken stem (fig. 28), or an uprooted tree (Lundquist 1995a). Although tree mortality because of wind damage is not usually desired, each of these treefalls plays a role in ecosystem processes. A fallen tree creates coarse woody debris, which can provide wildlife habitat, insect habitat that also produces forage for birds, and will eventually decompose and

return nutrients to the soil. A broken stem can provide a snag for cavity nesters. Uprooted trees create pit and mound microsites (Stephens 1956). Pit soils are often shallow and are high in organic matter, while mound soils are deep with buried soil horizons present (Norton 1989). These microsites can influence plant distribution and productivity.

There are several things to keep in mind if minimizing wind damage is an objective. Although ponderosa pine is considered a wind-firm species (Alexander 1987a), Graves (1899) noted that trees on exposed ridges of micaceous schist or limestone formations were typically toppled. Blowdowns typically occur in higher elevations in exposed stands. Wind damage most often occurs on hillsides directly exposed to west winds or at the edge of high clearings where wind can be funneled down a valley (Raventon 1994).

Ratios of tree height to diameter have been shown to predict ponderosa pine susceptibility to wind and snow damage in the Northern Rocky Mountains (Wonn and O'Hara 2001). Ponderosa pine trees that have height:diameter ratios greater than 80:1 (units in feet) are more prone to damage than trees with lower ratios. Height:diameter ratios below the threshold level can be maintained with thinnings during early stand development. If stands are older and have borderline height:diameter ratios, a series of low-intensity thinnings is preferable to allow greater diameter growth while maintaining stand stability (Wonn and O'Hara 2001). If the density of a stand is dramatically reduced, then the residual trees will not utilize the available growing space to the full potential. Furthermore, dense stands typically have high height:diameter ratios due to competition. These trees do not have the bole and root systems to withstand the wind when opened up, and many of the residual trees are likely to be uprooted, broken, or bent (Boldt and Van Deusen 1974).

Snow

Ponderosa pine is also frequently damaged by snow in the Black Hills. In 1903, a particularly heavy snowfall downed and killed trees over a wide area (USDA Forest Service 1948). Snowbend and breakage occur more often in sapling and small pole-sized trees (fig. 29) (Boldt and Van Deusen 1974). Small saplings usually recover from snowbending, but larger saplings often fail to return to their original form. Pole-sized trees are in the highest danger of breakage. Early thinning of dense stands will strengthen the residual trees and prevent losses from heavy snow (Schubert 1974). Maintaining tree height:diameter ratios of less than 80:1



Figure 29: Snow damage in a ponderosa pine stand in the Black Hills.

significantly decreases the susceptibility of snow damage in ponderosa pine trees (Wonn and O'Hara 2001).

Overstory-Understory Relationships

Background

Historical accounts of the understory vegetation of ponderosa pine forests in the Black Hills suggest a productive and diverse understory that would be excellent for livestock grazing (Dodge 1965; Graves 1899; Newton and Jenney 1880; Parrish and others 1996). With the onset of fire exclusion and forest management a century ago, ponderosa pine forests in the Black Hills have become more dense and extensive (McAdams 1995; Parrish and others 1996). Dense ponderosa pine forests in conjunction with overgrazing in the early part of the century have changed the herbaceous and shrub communities of the Black Hills. Different overstory stocking levels result in various levels of competition for light, nutrients, and water, which influence understory productivity and composition. In general, as ponderosa pine overstory density increases, understory plant biomass and species diversity decrease (Bennett and others 1987; Clary 1988; Pase 1958; Uresk and Severson 1989, 1998; Uresk and others 2000; Wrage 1994).

Influence of Soil Type

Soil properties in conjunction with overstory density and precipitation influence understory production in the Black Hills. Bennett and others (1987) developed models using stepwise regressions to predict understory herbage production potential of six predominant forest soils (Pactola, Sawdust, Stovho, Trebor, Vanocker, and Virkula) in the physiographic areas of the central crystalline core and limestone plateau in the Black Hills. They reported that canopy cover was the most important independent variable for predicting understory production, although, herbage yield did not respond to canopy coverage equally for different soils. Soils that produced more herbage at low canopy coverage had a steeper decline in production under high canopy coverage. Stovho and Virkula soils had the highest production potential of 1,600 lb per acre (1,800 kg per hectare) with no cover and normal May to June precipitation. However, Stovho soils produced more graminoids while Virkula produced more forbs. Sawdust soils had the lowest production with only 490 lb per acre (550 kg per hectare) under similar conditions, with hardly any forbs. The Pactola soil sites had different understory production and composition based on aspect. The southern aspects were dominated by graminoids and had lower production yields than the northern shrub-dominated aspect. If the objective were herbage production, the southern aspect would

have produced higher yields. It was also noted that graminoids were more negatively sensitive to changes in precipitation patterns than forbs. The equations produced by Bennett and others (1987) can be used to estimate forage potential on these six forest soils types.

Understory Production

Crown cover effects—Several studies in the Black Hills have investigated how ponderosa pine crown cover influences understory production for livestock and wildlife grazing (Bennett and others 1987; Pase 1958; Wienk 2001; Wrage 1994). Dense ponderosa pine stands have little or no understory vegetation, but stands that are more open have varying amounts of graminoids, forbs, and shrubs depending on site characteristics. Generally, herbage production decreases with an increase in crown cover in ponderosa pine forests. For instance, total herbage produced in a clearcut was estimated to be 2,160 lb per acre (2421 kg per hectare) but only 40 lb per acre (45 kg per hectare) with 70 percent crown coverage (Pase 1958). Another study reported total herbage produced for an open stand (0–20 percent crown coverage) to range from 1428 lb per acre to 1784 lb per acre (1600 to 2000 kg per hectare), for an intermediate cover class (30–60 percent) 638 lb per acre (715 kg per hectare), and for the dense canopy cover class (~85 percent crown coverage) 244 lb per acre (273 kg per hectare) (Wrage 1994).

Basal area effects—In 1963, a long-term study was initiated in the ponderosa pine/bearberry habitat type within the Black Hills Experimental Forest to determine the effects various growing stock levels (GSL)

and size class of ponderosa pine had on forage production and diversity (Severson and Boldt 1977). The GSLs were 0, 20, 40, 60, 80, 100, and 120 in units of ft² per acre (0, 5, 9, 14, 18, 23, and 28 in units of m² per hectare) (Uresk and Severson 1989,1998; Uresk and others 2000). The unthinned controls had basal areas of 161 to 174 ft² per acre (37 to 40 m² per hectare) for the pole-sized (6 to 7 inches; 15.2 to 17.9 cm d.b.h.) stands, and 118 to 144 ft² per acre (27–33 m² per hectare) for the sapling-sized (3 to 4 inches; 7.6 to 10.2 cm d.b.h.) stands. Three replications of each GSL within each size class were established, and plots were thinned every 5 years to maintain treatment GSL. Understory production was measured in 1974, 1976, and 1981, allowing researchers to monitor the relationship of stand density to understory production and diversity over the course of several years (Severson and Boldt 1977; Uresk and Severson 1989,1998; Uresk and others 2000).

The GSL affects the production of herbage in the understory. Generally, the production of graminoids, forbs, and shrubs is similar between sapling and pole stands (fig. 30 and 31) (Uresk and Severson 1989). Total understory production is highest in the clearcuts and lowest in the unthinned stands (Uresk and Severson 1989). In years of drought, however, total annual production of understory vegetation was actually higher in the GSL 20 ft² per acre (GSL 5 m² per hectare) plot and not the clearcut (Severson and Boldt 1977). This might be due to the shading effect of the overstory lowering the soil temperature and evapotranspiration. In general, understory production was greatest when density was less than GSL 100 ft² per acre (GSL 23 m² per hectare) in sapling stands and when density was less

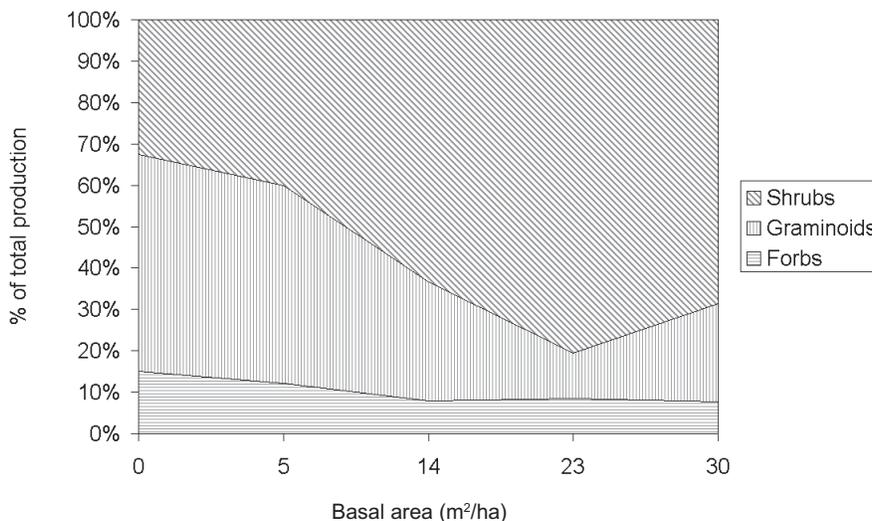


Figure 30: Percent of total understory production of different plant functional groups under various overstory densities in sapling-sized (3 to 4 inches [7.6–10.2 cm]) ponderosa pine stands in the Black Hills Experimental Forest (data for graph obtained from Uresk and Severson 1989).

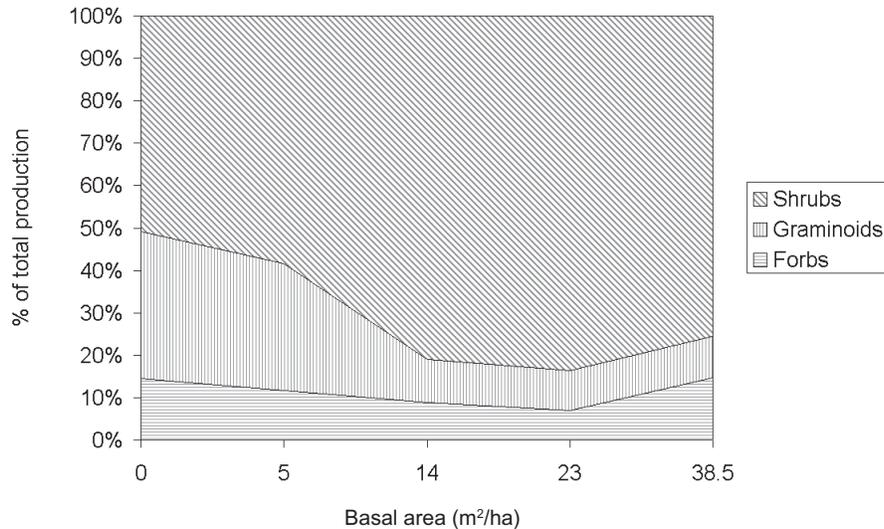


Figure 31: Percent of total understory production of different plant functional groups under various overstory densities in pole-sized (6 to 7 inches [15.2-17.9 cm]) ponderosa pine stands in the Black Hills Experimental Forest (data for graph obtained from Uresk and Severson 1989).

than GSL 60 ft² per acre (GSL 14 m² per hectare) in pole stands (Uresk and Severson 1998).

Overstory Influence on Functional Groups

In general, species richness declines with increasing canopy coverage (Wrage 1994) or basal area (Uresk and Severson 1989; Wienk 2001). Different densities of ponderosa pine overstory influence the species composition and abundance of the various functional groups in the understory (fig. 30 and 31). Uresk and Severson (1989) observed greater floristic diversity in ponderosa pine stands with low GSL, but suggested that total floral diversity would be greatest if all stocking levels, including clearcuts and unthinned dense stands, were present. To maximize the benefit to wildlife, these stands should be distributed across the landscape (Uresk and Severson 1998). Understanding how individual forage species within each stand respond to changes in stand density, and relating the response to the preferred species of fauna, would help the forest manager provide forage for a range of wildlife and livestock species.

Forbs—Generally, forb production is relatively low compared to shrub and graminoid production in understories of ponderosa pine forests in the Black Hills (fig. 30 and 31)(Severson and Boldt 1977; Uresk and Severson 1989,1998). The production of forbs is negatively affected by overstory canopy coverage (Wrage 1994) and basal area (Uresk and Severson 1989). For both sapling and pole-sized stands, forb production is highest where overstory density is low and decreases

substantially as the density increases (Uresk and Severson 1989). Pase (1958) reported 305 lb per acre (341.9 kg per hectare) of forb production on clearcuts compared to only 5 lb per acre (5.6 kg per hectare) under stands of 70 percent crown coverage. Similar patterns of forb production were noted by Uresk and Severson (1989).

Forb diversity is highest in areas without canopy coverage. Pase (1958) reported 63 species of forbs in areas with low crown coverage, but found few of the 63 persisting under stands with greater than 40 percent crown coverage. Forb species diversity was lower in the Uresk and Severson (1989) study than in the Pase (1958) study. Clearcuts were the most diverse, with 21 species in the pole stands and 18 species in the sapling stands. There were significantly fewer forb species in both pole and sapling stands that exceeded GSL 60 ft² per acre (14 m² per hectare) (Uresk and Severson 1989).

Graminoids—Graminoids are more sensitive to changes in canopy coverage and basal area than are forbs or shrubs (fig. 30 and 31). Several studies report a sharp decrease in graminoid production with an increase in ponderosa pine canopy coverage (Pase 1958; Wrage 1994) and basal area (Uresk and Severson 1989). Pase (1958) reported up to 1,730 lb per acre (1,939 kg per hectare) of graminoid production in a clearcut compared to 25 lb per acre (28 kg per hectare) under canopy coverage greater than 70 percent. Production estimates reported by Uresk and Severson (1989) were slightly lower in clearcuts than what Pase (1958) reported. Uresk and Severson also noted that graminoid production of 817 lb per acre (916 kg per hectare) was lower

in pole-sized stands than the 1,157 lb per acre (1,297 kg per hectare) reported for the sapling-sized stands. Furthermore, they reported a significant decrease in the number of graminoid species in stands that exceeded GSL 60 ft² per acre (14 m² per hectare) for both sapling- and pole-sized stands.

Shrubs—Shrub production is less sensitive to ponderosa pine canopy coverage and basal area. Shrubs make up 35 to 50 percent of total understory production in clearcuts (fig. 30 and 31) and up to 70 percent in dense stands. Estimates of shrub production vary between studies. Pase (1958) reported 10 lb per acre (11 kg per hectare) under crown coverage greater than 70 percent and 125 lb per acre (140 kg per hectare) in clearcuts. Shrub productivity differed between pole- and sapling-sized stands (Uresk and Severson 1989). In the sapling-sized stands, shrub productivity of 713 to 834 lb per acre (800 to 935 kg per hectare) was maintained up to a density of GSL 100 ft² per acre (23 m² per hectare) and then decreased to 205 lb per acre (230 kg per hectare) at GSL 118 to 144 ft² per acre (GSL 27 to 33 m² per hectare). Clearcutting maximized shrub production (1,200 lb per acre [1,344 kg per hectare]) after treatment in pole-sized stands. Similar productivity in the pole-sized stands was reported after sapling-sized stands were thinned to GSL 20 ft² per acre (GSL 5 m² per hectare) and 40 ft² per acre (GSL 9 m² per hectare), but denser treatments lessened shrub productivity.

There was a significant reduction in shrub species in pole stands that exceeded GSL 60 ft² per acre (GSL 14 m² per hectare), but no difference in species diversity in the sapling stands (Uresk and Severson 1989). Shrub diversity under different ponderosa pine densities varied with study site, but in general, within a study site, diversity did not differ, although dominance did differ. Uresk and Severson (1998) reported up to 14 different shrub species in the ponderosa pine understory, Pase (1958) reported 18, and Wrage (1994) counted only six.

Snags and Coarse Woody Debris

Long viewed as a waste of wood fiber and added fire hazard, snags and coarse woody debris are now being recognized for the benefits they provide for wildlife habitat, nutrient cycling (Graham and others 1994; McComb and Lindenmayer 1999), and forest structural complexity (Franklin and others 1997). Snags and coarse woody debris are produced by natural disturbances such as wind, fire, insect and pathogen attacks, and timber harvesting. The creation of snags and coarse woody debris depends upon the intensity, frequency,

and pattern of the disturbance. For instance, an epidemic of mountain pine beetles or a large fire can produce large numbers of snags and more coarse woody debris. Ice or wind can break treetops, resulting in fungal infections that in turn produce lower amounts of snags and less coarse woody debris. Thus, depending on the disturbance, various decomposition stages of snags and coarse woody debris can occur in the forest, providing habitat for several different organisms (McComb and Lindenmayer 1999).

A snag is defined as a dead or partly dead tree at least 4 inches (10.2 cm) d.b.h. and at least 6 feet (1.8 m) tall (Thomas and others 1979). Over time, snag characteristics change from the time the tree dies to when it falls to the ground (fig. 32). Snags can be either characterized as hard or soft depending on their state of decomposition. Hard snags have mostly sound wood on the outside, but may have heart rot decay on the inside. Snags in an advanced stage of decay and decomposition are commonly referred to as soft snags. Each stage in the decomposition of a snag has a particular value for certain wildlife. Therefore, the presence of sufficient numbers of snags with a variety of diameters and decay classes is critical in maintaining populations of several wildlife species (Raphael and White 1984; Scott and others 1977; Thomas and others 1979).

Snag utilization

Snags provide habitat for many forms of life. Fungi, mosses, and lichens use decayed wood as a growth substrate. Insects often use snags for cover and food throughout their development. At least 23 birds and 10 mammal species in the Black Hills depend on snags during a portion of their life history for nest sites, roosts, perches, dens, and foraging substrate (USDA Forest Service 1996a).

Animals that excavate their own nest cavity in a snag are known as primary cavity nesters. Primary cavity nesters have an important role in forested ecosystems because most excavate a new cavity each year. The cavities left behind will subsequently be used by secondary cavity nesters such as chickadees (family Paridae), wrens (family Troglodytidae), bluebirds (family Muscicapidae), flycatchers (family Tyrannidae), and swallows (family Hirundinidae), who do not have the excavating capability (Balda 1975; Cunningham and others 1980; McComb and Lindenmayer 1999). Mammals, such as the northern flying squirrel (*Glaucomys sabrinus bangsi*) and bats (family Vespertilionids), also utilize these cavities (McComb and Lindenmayer 1999). Some bird species such as the red-breasted

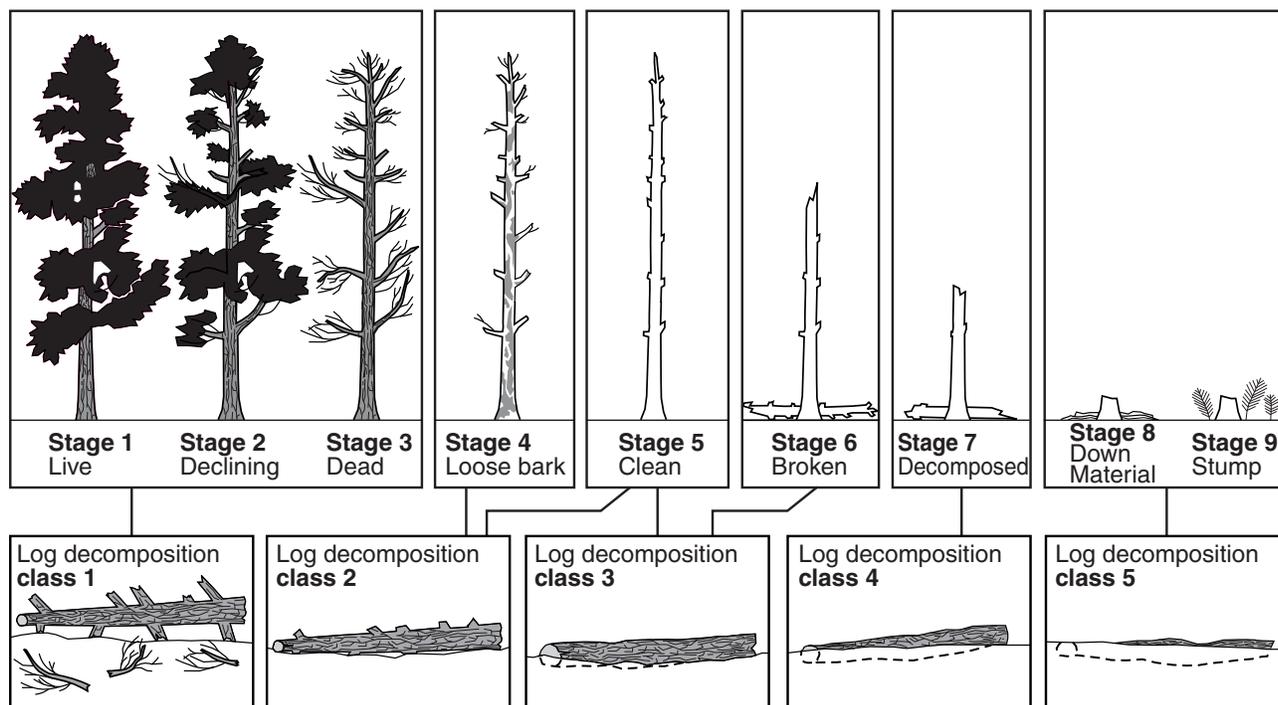


Figure 32: Various stages of the formation of snags and coarse woody debris (figure adapted from Maser and others 1979).

nuthatch (*Sitta canadensis*) are opportunistic nesters that utilize existing cavities or create their own nests in soft snags (Cunningham and others 1980).

Most cavity nesters in the Black Hills are year-round residents and make up a large proportion of the winter bird community (Rumble and others 1999). Cavities provide protection from predators and winter weather. The survival of some resident birds may depend solely on the availability of suitable snags (Sydeman and Guntert 1983). For instance, pygmy nuthatches (*Sitta pygmaea*) roost communally in groups of 10 to 14 birds (Cunningham and others 1980; Hay and Guntert 1983; Sydeman and Guntert 1983). Therefore, snags large enough to accommodate such a large group are required to aid in the survival of this species.

The availability of suitable snags directly influences the population levels of primary and secondary cavity nesters (Cunningham and others 1980; Thomas and others 1979) as well as species density and diversity (Raphael and White 1984). Since secondary cavity nesters rely on primary cavity nesters to provide nest sites, supplying enough snags for primary cavity nesting pairs is a good strategy. Thomas and others (1979) suggested up to three snags are needed for each primary cavity nesting pair because each year a pair might go through a number of false starts before finally excavating a cavity. Furthermore, since certain species nesting habitat are positively or negatively effected when snags are clumped, the spatial arrangement of snags

on the landscape should be diverse with both clumping of snags in small patches as well as even distribution (Thomas and others 1979).

Cavity-nesting species do not randomly pick snags for cavities; instead, a snag is chosen based on certain attributes such as tree species, diameter, bark coverage, and decomposition stage (Cunningham and others 1980; Lentile and others 2000; Raphael and White 1984; Thomas and others 1979). In the Black Hills, evidence of cavity use was highly correlated ($r^2 = 0.95$) with snag d.b.h. and was only observed in ponderosa pine, aspen, and paper birch (Lentile and others 2000). Evidence of forage use in snags greater than 3 inches (>7.6 cm) d.b.h. was found in ponderosa pine (46 percent), aspen (26 percent), paper birch (49 percent), bur oak (17 percent), white spruce (19 percent), and Rocky Mountain juniper (2 percent), although highest forage use was observed for trees greater than 10 inches d.b.h.. Since 98 percent of the snags greater than 10 inches (>25 cm) d.b.h. surveyed in the Black Hills are ponderosa pine (Lentile and others 2000), the importance of maintaining ponderosa pine snags for forage is crucial.

Snag Population Dynamics

Snag density—A survey in the summer of 1999 assessed snag densities across the entire Black Hills National Forest (Lentile and others 2000). The mean

density of snags regardless of tree species in areas with no recent evidence of logging (20 to 30 years) was reported to be 17 snags per acre in trees greater than 3 inches (>7.6 cm) d.b.h., 3.6 snags per acre in snags greater than 10 inches (>25 cm) d.b.h., and 0.33 snags per acre in snags greater than 20 inches (>51 cm) d.b.h.. Snag composition of snags > 10 inches (>25 cm) d.b.h. was predominately (more than 98 percent) ponderosa pine. Smaller diameter snags were still dominated by ponderosa pine (81 percent), but 19 percent of the composition was aspen, paper birch, oak, white spruce, and Rocky Mountain juniper. Ponderosa pine snag density ranged from 5.22 snags per acre in the 4- to 8-inch (10- to 20-cm)-diameter class to 0.44 snag per acre in the 20-plus-inch (51-cm)-diameter class. Proposed guidelines for snags in the Black Hills National Forest's Phase I Amendment to the Forest Plan call for an average retention of four ponderosa pine snags per acre greater than 10 inches (25 cm) d.b.h. (on north- or east-facing slopes) and two ponderosa pine snags per acre greater than 10 inches (25 cm) d.b.h. (on south- or west-facing slopes), of which 25 percent on all aspects must be greater than 20 inches (51 cm) in diameter (USDA Forest Service 2001b).

Snag persistence—Lentile and others (2000) also surveyed three randomly selected snag populations in the southern (Jewel Cave $n = 24$), central (Black Fox $n = 26$), and northern (Thompson's Gulch $n = 27$) areas of the Black Hills. Snags at each site were randomly chosen and date of tree death was determined to estimate the amount of time a snag persisted. Snag persistence ranged from 8 to 46 years for the southern site, 9 to 57 years for the central site, and 1 to 17 years for the northern site. Since the distribution of snag persistence was not uniform, Lentile and others (2000) calculated the median age of snag longevity, which would be a better estimate than the mean snag longevity. They reported that the median age of snag longevity (defined as the years since death of a tree that half of the snags from a particular site have persisted) was 17.5 years for the southern site, 24 years for the central site, and 3 years for the northern site. It should be noted that stand age of both the southern and central sites ranged between 150 to 400 years, while the northern site was 60 to 130 years. The lower values of snag longevity in the northern site were attributed to the age of the stands, which would contain few older snags.

Age of ponderosa pine at time of death is positively related to snag longevity in western Montana (Smith 1999a,b). Smith found that trees greater than or equal to 200 years old had a greater chance of standing for at least 10 years, while younger trees fell sooner. It should be noted, however, that these 200-year-old trees were

subjected to frequent, nonlethal underburns that created minor basal scarring and increased pitch flows in tree boles (Smith 1999a,b).

Coarse Woody Debris

Once a snag falls, it becomes coarse woody debris and is still valuable for wildlife habitat and forage, as well as a nutrient pool for the soil (fig. 32). Coarse woody debris is woody material derived from tree limbs, boles, and roots in various stages of decay (Graham and others 1994). In general, larger diameter coarse woody debris offers a broader range of habitats than smaller pieces. Coarse woody debris greater than 3 inches (7.6 cm) in diameter is suggested as a minimum because this size will still contribute to the forest floor and soil before completely decaying or being consumed by fire (Graham and others 1994). Furthermore, many species use the various stages of decomposition of coarse woody debris for habitat and forage. Providing many different sizes of coarse woody debris allows the various stages to occur in a forest.

The ecological role of coarse woody debris is multifaceted. First, coarse woody debris provides habitat for a variety of organisms, including reptiles, amphibians, birds, small mammals, insects, plants, and fungal and microbial species (Franklin and others 1997; Hassinger 1989; Maser and others 1979). Coarse woody debris also has the potential to hold large amounts of water that may be important for plants during dry periods, as well as for development of fungi that serve as forage for several small mammals (Hassinger 1989). Physically, coarse woody debris reduces soil erosion by acting as a barrier to soil movement (Franklin and others 1985), provides shade and disrupts airflow, insulating and protecting new forest growth, and can protect tree seedlings from livestock damage (Graham and others 1994).

Coarse woody debris is a long-term source of organic matter and nitrogen (Franklin and others 1997). As coarse woody debris decomposes or burns, it adds nutrients such as sulphur, phosphorus, and nitrogen into the soil (Graham and others 1994). In advanced stages of decay, it produces humus and buried residue that are excellent sites for the formation of ectomycorrhizal root tips (Harvey and others 1981). Ectomycorrhizal fungi provide a food source for many small rodents and larger predators, and aid woody plants in obtaining water and nutrients (Maser and others 1986; Reynolds and others 1992).

Past practices of slash disposal and site preparation after a logging operation have reduced the coarse woody debris available in the forest. It is essential

that managers recognize that coarse woody debris is not a waste of wood fiber, but a resource that promotes wildlife habitat, nutrient cycling, and structural complexity.

Management of Snags and Coarse Woody Debris

Providing enough snags across a range of forested habitats and in large diameter classes, as well as a commitment to providing a variety of decomposition stages of coarse woody debris, will take a concentrated effort. Plans for the management of snags and coarse woody debris must consider numbers, sizes, and dynamics of decay and persistence. Both hard and soft snags are needed, and new snags must be recruited periodically to replace those that have fallen. Managers should protect and retain standing dead trees and coarse woody debris that have occurred naturally. In stands without snags or coarse woody debris, trees can be killed in several ways to create snags. Wolfy, dead top, or deformed trees are excellent candidates for snags, provided they do not have butt or root rot. A commercially available power girdler with a rotary cutting head called the Li'l Beaver girdles trees between 4 and 12 inches (10 cm to 30.5 cm) d.b.h. efficiently, while a chain saw works best for trees larger than 12 inches (>30.5 cm) d.b.h. (Kilroy and Windell 1999). The disadvantage of girdling is that decay begins in the sapwood before the heartwood, and by the time the tree is rotten enough for an animal to excavate a cavity, the snag may be ready to fall. Girdling must be thorough to kill the tree (Bull and Partridge 1986). Herbicides have been used to kill trees, but Bull and Partridge (1986) reported trees killed in this manner typically fell before girdled trees and were seldom used for forage or nest habitat.

Bull and Partridge (1986) recommended topping trees 49 to 82 feet (15 to 25 m) above ground and removing lower limbs as the best technique to create snags in ponderosa pine. The topping technique is currently being tested in the Black Hills, but results (Smith 2001, personal communication) are not yet available. Another technique involves killing a tree by injecting a fungicide into the phloem and then inoculating the heartwood with a fungus (Hunt 1990). Although this method has not been tested, Hunt suggests that this method could prolong the longevity of a snag by preserving the sapwood while stimulating decay in the heartwood.

Increasing the decay resistance of a snag has been suggested as a method to prolong its longevity (Smith 1999b). Extractives such as terpenoids,

flavenoids, tropolones, and stilbenes, important to the naval stores industry in the 19th century, are now being investigated for their decay resistance properties in snags. Both sapwood and heartwood contain extractives (Hillis 1977; Zavarin and Cool 1991). When a tree is wounded, such as in a low-intensity fire, extractives are produced from resin canals in ponderosa pines (Zavarin and Cool 1991). Extractives are thought to be toxic to insects and diseases (Kurth 1952; Scheffer and Cowling 1966), thus increasing the decay resistance of the heartwood. Smith (1999b) found that pitch content was no higher in standing ponderosa pine snags than in broken snags. However, the testing of pitch content was exploratory, and further study is needed to better understand this relationship. Furthermore, she suggests that in future studies, sapwood and heartwood samples should be separated for chemical analysis.

Major Wildlife Species

Habitat Capability Models (HABCAP)

Habitat capability is defined as the capacity of a given area to meet the needs of a species, either seasonally or year round (Lyon and Christensen 1992) based on a combination of forage and cover. Habitat capability (HABCAP) models are frequently used to assess the consequence of management activities on the quality of wildlife habitat. These models represent the ability of an area to support a variety of wildlife species, but do not represent population levels. HABCAP models contain coefficients, ranging from 0 to 1.0, that rate the cover and forage capability of structural stages and forest type for each wildlife species. Coefficients of zero indicate conditions with no value and coefficients of 1.0 denote optimal conditions for that species. The Rocky Mountain Region of the USDA Forest Service has published a user guide for the HABCAP program (USDA Forest Service [n.d.]).

Initial use of the HABCAP model by the U. S. Forest Service Rocky Mountain Region utilized information from Towry (1984) and Hoover and Wills (1984). Towry (1984) discussed acreage for minimum viable populations, and ratings of structural stages within various ecosystems, to meet habitat requirements for numerous mammals and birds of Colorado. The Black Hills has modified the Rocky Mountain regional HABCAP model to create a HABCAP model specific to the Black Hills area. The adjustments were based on literature reviews, experience with Black Hills habitats and wildlife habitat associations, and empirical data

(Mills and others 1996; Rumble and Anderson 1995; Rumble and others 1999). The Black Hills HABCAP model utilizes ARC/INFO, a GIS program, to analyze habitat capability for species based on spatial data, and has the ability to map habitat capability results (USDA Forest Service 1996b). The vegetative structural stages described by Buttery and Gillam (1984) are used for the HABCAP model in the Black Hills (table 3).

The intended use of HABCAP output is as a planning tool for different management scenarios. The model is not intended to predict population levels because HABCAP does not include all the factors that affect population densities. The model only estimates habitat capability at one point in time and does not simulate change over time. Spatial distribution of cover and forage, except for deer and elk, is not incorporated into the model due to lack of information. Furthermore, the model assumes snags, down woody material, and

water are well distributed across the analyzed area (USDA Forest Service 1996b). Neither herbaceous understory composition nor subdominant tree species in mixed species forests are included in the model parameters. Topography is ignored and competition for forage, cover, or space among different animal species (both wild and domestic) is not addressed. Lastly, the model permits compensation for poor forage by good cover, which is not always true (USDA Forest Service 1996b).

Rocky Mountain Elk (*Cervus elaphus nelsoni*)

Prior to 1870, the Black Hills was home to the Manitoban elk (*Cervus elaphus manitobensis*) but by 1888, overhunting had led to extirpation from the area (Turner 1974). Between 1912 and 1914, the Rocky Mountain subspecies of elk (*Cervus elaphus nelsoni*)

Table 3: Cross-reference of ponderosa pine structural stages categorized by diameter breast height. Numbers in parentheses denote the numeric structural stage code.

D.B.H. (inches)	Buttery and Gillam (1984) ^{a,b}	Region 3 Vegetation Structural Stage ^{c,d}	Region 2 Structural Stage ^b	Black Hills NF Structural Stage ^e
0	Grass-forb stage (1)	Grass/forb/shrub (1)	Nonstocked (1)	Grass/forb/shrub (1)
0 to 1	Shrub-seedling stage (2)	Grass/forb/shrub (1)	Established (2)	Grass/forb/shrub (1)
1 to 2	Sapling-pole stage (3)	Seedling/sapling (2)	Small (3)	Seedling/sapling (2)
2 to 3	Sapling-pole stage (3)	Seedling/sapling (2)	Small (3)	Seedling/sapling (2)
3 to 4	Sapling-pole stage (3)	Seedling/sapling (2)	Small (3)	Seedling/sapling (2)
4 to 5	Sapling-pole stage (3)	Seedling/sapling (2)	Small (3)	Seedling/sapling (2)
5 to 6	Sapling-pole stage (3)	Young forest (3)	Medium (3)	Young forest (3)
6 to 7	Sapling-pole stage (3)	Young forest (3)	Medium (3)	Young forest (3)
7 to 8	Sapling-pole stage (3)	Young forest (3)	Medium (3)	Young forest (3)
8 to 9	Mature stage (4)	Young forest (3)	Medium (3)	Young forest (3)
9 to 10	Mature stage (4)	Young forest (3)	Large (4)	Mid-aged forest (4)
10 to 11	Mature stage (4)	Young forest (3)	Large (4)	Mid-aged forest (4)
11 to 12	Mature stage (4)	Young forest (3)	Large (4)	Mid-aged forest (4)
12 to 13	Mature stage (4)	Mid-aged forest (4)	Large (4)	Mid-aged forest (4)
13 to 14	Mature stage (4)	Mid-aged forest (4)	Large (4)	Mid-aged forest (4)
14 to 15	Mature stage (4)	Mid-aged forest (4)	Large (4)	Mature forest (5)
15 to 16	Mature stage (4)	Mid-aged forest (4)	Large (4)	Mature forest (5)
16 to 17	Mature stage (4)	Mid-aged forest (4)	Very large (4)	Mature forest (5)
17 to 18	Mature stage (4)	Mid-aged forest (4)	Very large (4)	Mature forest (5)
18 to 19	Mature stage (4)	Mature forest (5)	Very large (4)	Mature forest (5)
19 to 20	Mature stage (4)	Mature forest (5)	Very large (4)	Mature forest (5)
20 to 21	Mature stage (4)	Mature forest (5)	Very large (4)	Old forest (6)
21 to 22	Mature stage (4)	Mature forest (5)	Very large (4)	Old forest (6)
22 to 23	Mature stage (4)	Mature forest (5)	Very large (4)	Old forest (6)
23 to 24	Mature stage (4)	Mature forest (5)	Very large (4)	Old forest (6)
> 24	Old growth (5)	Old forest (6)	Very large (4)	Old forest (6)

^a Structural stage classification used for white-tailed deer (USDA Forest Service 1996a) and nongame birds (Mills and others 1996, 2000; Rumble and others 1999).

^b Overstory canopy coverage: A = 0 to 40 percent; B = 40 to 70 percent; C = more than 70 percent.

^c Vegetation structural stages used in the Southwestern Goshawk Guidelines (Reynolds and others 1992).

^d Overstory canopy coverage: A = 0 to 40 percent; B = 40 to 60 percent; C = more than 60 percent.

^e Phase I goshawk analysis structural stages for Black Hills National Forest.

was introduced to the area from Montana and Wyoming (Turner 1974). In 2000, the estimated elk population in the South Dakota section of the Black Hills was 3,895 (South Dakota Game Report 2000). An estimate for the Wyoming portion of the Black Hills in 1996 was approximately 500 elk (USDA Forest Service 1996a). Elk in the Black Hills region have been studied in Custer State Park (Millspaugh 1995; Millspaugh and others 1998; Rumble and others 2002), Wind Cave National Park (Bauman 1998; Wydeven and Dahlgren 1983), the central Black Hills (Hippensteel 2000; Rice 1988), and southern Black Hills (Rice 1988).

Elk in the Black Hills utilize a variety of habitats for forage, loafing, and escape cover throughout the year. Although most elk in South Dakota are considered non-migratory, seasonal movement among elk populations to different habitat types in response to snow depth (greater than 20 inches; >51 cm) and hunting pressure have been reported (Millspaugh 1995; Rice 1988; Bauman 1998). One issue of concern is the movement of elk from the Black Hills National Forest into Wind Cave National Park during the hunting season, which diminishes hunter success in the National Forest (Rice 1988) and places additional stress on food sources and competition with other ungulates within the National Park (Bauman 1998; Wydeven and Dahlgren 1985). Therefore, providing sufficient forage and security cover habitat for elk within the Black Hills National Forest might avoid or lessen this problem.

Habitat requirements

Historically, many wildlife biologists have accepted the idea that providing dense vegetative cover for thermal cover enhances the survival of wild ungulates by moderating the effects of harsh weather and minimizing the energy required for thermoregulation. The majority of studies supporting the thermal cover hypothesis are based on observational studies of habitat selection. However, a recent study in the Blue Mountains of northeastern Oregon tested the thermal cover hypothesis by monitoring body mass and composition of elk subjected to 1 of 4 levels of cover during 4 winter and 2 summer season-long experiments (Cook and others 1998). They concluded that thermal cover does not significantly improve the energetic status and productive performance of elk (Cook and others 1998). Instead, they suggest that observational studies of elk habitat selection may be more related to other needs such as forage or security. Furthermore, a test of the HABCAP model for elk in Custer State Park suggests that forage is three times more important than cover, and the modification of the model resulted in a better

fit between elk dispersion patterns and predicted habitat effectiveness (Rumble and others 2002).

Elk diets depend on forage availability and the time of year (Hippensteel 2000; Kufeld 1973; Wydeven and Dahlgren 1983). In the Central Black Hills, elk consumed graminoids, shrubs, and ponderosa pine in both the winter and summer, but consumption of forbs was very low (0.16 to 2.78 percent) in the winter and ranged from 4.2 to 24.9 percent in the summer (Hippensteel 2000). The large range in forb consumption between the summer of 1994 (24.9 percent) and the summer of 1995 (4.2 percent) in the central Black Hills elk herd was contributed to precipitation and forb availability (Hippensteel 2000). Elk in the Wind Cave National Park area were reported to feed primarily on forbs in the fall and winter and graminoids in the spring and summer (Wydeven and Dahlgren 1983). Furthermore, shrubs only contributed 7 to 9 percent of the elk's diet (Wydeven and Dahlgren 1983), as opposed to making up 30 percent of the central Black Hills elk diet (Hippensteel 2000).

Elk habitat selection throughout the year for forage, escape cover, and calving indicates the need for a variety of habitat types within the Black Hills. Openings created or maintained by wildfire and prescribed burns were extensively used year round as forage habitat unless disturbance prevented such use (Rice 1988). In general, annual habitat use patterns of elk showed high proportional use of meadow/range and fire-killed forests (Bauman 1998; Millspaugh 1995; Rice 1988) most likely a result of higher forage production and diversity (Uresk and Severson 1989). Openings less than 0.5 acre (0.2 ha) with limited canopy of aspen or bur oak were also heavily utilized for forage habitat (Rice 1988). In the summer, elk still utilized openings but increased their use of ponderosa pine forests (Bauman 1998; Millspaugh 1995). This increased use of forests was most likely due to the increase in human disturbance (for example, tourist activity, vehicles); however, Millspaugh and others (1998) suggest that this increased use was to provide relief from summer temperatures. Although Cook and others (1998) did not find any evidence that forest cover enhanced the growth and condition of elk in their summer experiments, they did note that elk under dense forest cover required less water. Whether or not the elk selected the ponderosa pine forests for summer bed sites based on a cooler microclimate or for security, the elk selected areas with canopy closures more than 54 percent, increased basal area, and more trees per acre than random sites within Custer State Park (Millspaugh and others 1998).

Elk habitat selection is influenced by several human-caused disturbances such as roads used by

motorized vehicles (Millsbaugh 1995; Rowland and others 2000; Rumble and others 2002; Ward and Cupal 1979), gunshots (Ward and Cupal 1979), and hunter activities (Millsbaugh and others 2000). Elk that sense danger respond with increased alertness accompanied by an increase in heart rate. If the disturbance is deemed as hazardous to the elk, it will typically run away. This increase in heart rate in combination with running away expends large amounts of energy and can negatively affect the elk's health, especially in times of physiological stress. Ward and Cupal (1979) reported an increase in heart rate at distances up to 1,300 feet (400 m) for gunshots, humans seen on foot, audible car horns, and stopped vehicles; however, moving high-speed traffic produced very few heart rate reactions.

Providing hiding cover that enables the elk to feel safe is considered a critical component of elk habitat (Millsbaugh and others 1998). In Custer State Park, elk tend to avoid areas with high hunter density and hide in dense vegetative cover to minimize the disturbance during the fall hunting season. Dense vegetative cover allows greater space overlap with hunters. In areas with low vegetative cover, elk and hunter space overlap is much larger. This was especially noticeable in the fire-killed areas of the Park, which has lower vegetative cover (Millsbaugh and others 2000). Rice (1988) observed the use of large openings in early fall ceased during daylight, and the use of openings less than 5 acres (2 ha) in size and next to dense ponderosa pine forests increased substantially during hunting season.

The effect of roads on elk movement, distribution, and habitat use appears to be a function of season and amount of traffic (Millsbaugh 1995; Rowland and others 2000; Rumble, in press; Ward and Cupal 1979). For instance, proximity to primary roads (more than 35 vehicles per week) in Custer State Park had a negative effect on the dispersion patterns of elk during the winter, while primitive roads (less than 7 vehicles per week) did not negatively affect elk (Rumble and others 2002). In northeast Oregon, female elk were reported to select areas away from open roads in both spring and summer (Rowland and others 2000). Calving elk in Custer State Park typically avoid areas within 660 feet (200 m) of primary and secondary roads (Millsbaugh 1995). Several studies have reported that the negative effects of roads decline as distance from the road increases (Rowland and others 2000; Rumble and others 2002). Negative effects of roads can be minimized by providing greater canopy coverage along roadsides (Unsworth and others 1998).

The spatial arrangement of forage and cover habitats is an important component to consider in elk management. Calving habitat typically occurs in

openings less than 1 acre (0.4 ha) in size, with no overstory more than 5 feet (1.5 m) tall and ground cover with herbaceous vegetation, shrubs, and some coarse woody debris. These openings typically have at least one side adjacent to dense ponderosa pine stands with high canopy coverage (Rice 1988). Elk in Custer State Park were found to use forage areas 330 feet or less (100 m) from cover proportionally more than forage areas more than 660 feet (200 m) from cover (Rumble, in press). Season also influences how far elk will venture into an opening. Rice (1988) noticed that during spring, summer, and fall, elk would rarely feed more than 300 feet (90 m) from the forest edge, but in winter, elk were observed in the middle of openings over one-fourth mile (0.4 km) from the forest edge.

Management guidelines

Providing adequate food sources while maintaining cover, through a mixture of open meadows and closed dense forests, will help maintain elk populations in the Black Hills. This balance of conditions at the stand scale can be achieved through multiaged management, either utilizing group selection or even-aged management of two to three age-classes. In either case, silviculture activities should emphasize an irregular spatial distribution of trees to allow canopy openings for understory production and groups of trees for security.

Understanding how different levels of forest density favor plant functional groups (fig. 30 and 31) will allow managers to manipulate forest cover to provide sufficient amounts of the preferred forage at different times of the year. Tall shrubs such as woods rose, snowberry, and western serviceberry on winter range can provide forage in years with deep snow. Management practices that benefit elk summer range should focus on northern slopes, while winter range should focus on southern slopes. Maintaining meadow/range areas within the forest canopy with prescribed fire can help prevent tree encroachment. Furthermore, prescribed fire has been shown to increase forage production in Montana (Koncerak 1996) and should be tested in the Black Hills.

Providing sufficient security cover for elk can help minimize human-related disturbance. Guidelines for elk hiding cover call for vegetation that is capable of hiding 90 percent of a standing adult elk from the view of a human at a distance of at least 200 feet (61 m) (Thomas and others 1979). Achieving these requirements could be difficult in mature stands in which only the boles provide screening. Smith and Long (1987) demonstrated that, for a mature lodgepole pine stand to meet the guideline requirements, the stand would be so dense that it would slow growth, reduce vigor,

and be susceptible to mountain pine beetle infestation. However, young stands that have full crowns were shown to provide hiding cover that met the guidelines. Therefore, providing patches of young stands and managing these stands with thinning operations that maintain low crowns for extended periods of time can provide suitable security cover.

Security cover should be in close proximity to forage areas. Meadow areas near roads should be concealed by a forested buffer zone of a least 1,300 feet (400 m) to mitigate the negative effects of roads and to ensure these areas are utilized by the elk. Known travel lanes used by elk should be maintained. In areas that lack cover, travel lanes can be provided with timber stringers or riparian zones. To remain effective, buffer zones should be actively managed to retain their desired attributes. Ideally, buffer zones should be irregularly structured forests with groups of trees of all ages and sizes to ensure their integrity through time. These conditions are best maintained through use of uneven-aged management.

Hunter disturbance can be lessened by controlling hunter density and access to areas with low vegetative cover (Millspaugh and others 2000). Retaining areas of dense vegetation within a landscape that has been subjected to wildfire will help mitigate hunter disturbance. In areas of high recreational activity such as hiking or camping, providing forested buffer zones will lessen human-caused harassment.

White-Tailed Deer (*Odocoileus virginianus dakotensis*)

The white-tailed deer population within the Black Hills is on the decline. Population estimates from the early 1990s report approximately 75,000 white-tailed deer lived in the Black Hills. Current estimates place white-tailed deer populations at 29,000 (South Dakota Game Report 2000). In the central Black Hills (Pennington, Lawrence, Crook, and Weston Counties), white-tailed deer herds are estimated to be decreasing at a rate of 10 to 15 percent annually (DePerno 1998; DePerno and others 2000). To effectively manage for white-tailed deer, managers need information on white-tailed deer habitat requirements in the Black Hills. In response to this need, several studies have investigated the specific habitat requirements (DePerno 1998; Kennedy 1992; Sieg and Severson 1996; Stefanich 1995; Thilenius 1972; USDA Forest Service and others 1982) and diet (Hill 1946; Hill and Harris 1943; Hippensteel 2000; Schenck and others 1972; Schneeweis and others 1972; Sieg and Severson 1996; USDA Forest

Service and others 1982) of Black Hills white-tailed deer populations.

White-tailed deer in the Black Hills migrate between summer and winter ranges (DePerno 1998). Deer migration from high-elevation summer ranges to low-elevation winter range typically occurs between August and February. Females typically migrate to summer ranges in May, about 3 weeks before giving birth (DePerno 1998). A map of the Black Hills National Forest with migratory patterns and summer, winter, and critical winter ranges is provided in the Coordination Guidelines for Timber Management in Black Hills Deer Habitat manual (USDA Forest Service and others 1982).

Habitat requirements

White-tailed deer population decline has been attributed to the deterioration of habitats used for cover and forage, especially on winter range (DePerno 1998; DePerno and others 2000; Griffin and others 1992). It has been suggested that the reduction in forage quality and availability because of increased ponderosa pine overstory density has lowered the reproductive potential of white-tailed deer in the Black Hills (Rice 1984). Although hunting white-tailed deer in the Black Hills is popular, DePerno and others (2000) reported that 71 percent of female white-tailed deer mortality in the central Black Hills was natural (malnutrition, predation, and sickness), and over 50 percent of the female mortality occurred in the spring coinciding with late spring snowstorms. Females are most susceptible then because they are typically in the third trimester of pregnancy and nutritionally stressed from poor-quality forage on fall and winter ranges (DePerno and others 2000).

Habitat and forage availability differs between the northern and central Black Hills, and therefore affects the habitat selection and diet composition of white-tailed deer populations within each region. Although ponderosa pine forests dominate both regions, the forested habitats utilized by deer populations in the northern Black Hills have lower basal area, lower average d. b. h., and lower total canopy coverage than the forested habitats available to central Black Hills deer populations (Hippensteel 2000). In addition, northern Black Hills forests have agricultural fields that supplement the natural forage and a prevalence of deciduous habitats with an understory shrub component, which contributes to better escape cover and browse (DePerno 1998; Kennedy 1992).

Forage availability and diet quality in the winter depend on winter severity and duration (Hippensteel 2000; Osborn 1994; Sieg and Severson 1996). In the

northern Black Hills, low-growing shrubs such as Oregon grape and bearberry are important forage during mild winters, but in severe winters with deep snow, ponderosa pine needles, bur oak twigs, and common juniper are consumed (Hill 1946; Hippensteel 2000; Schenck and others 1972; Schneeweis and others 1972; Sieg and Severson 1996). Although the highly nutritious and digestible bur oak acorn has not been identified as an important food source, it is probably consumed when available (Sieg and Severson 1996). In the central Black Hills, Oregon grape, bearberry, and common juniper are not present on the deer's winter range; instead, deer in this region rely on serviceberry, western wild rose, and ponderosa pine needles for food (Hippensteel 2000). In the southern Black Hills, ponderosa pine, true mountain mahogany, common juniper, bearberry, western snowberry, and Rocky Mountain juniper are important browse during winter (Schenck and others 1972). The winter-spring transition is a critical time for all Black Hills white-tailed deer. Although spring deer diets increase in graminoids and new leaves from trees and shrubs (Hill 1946), spring snowstorms frequently cover the graminoids and winter browse is still utilized.

Availability of high-quality forage in summer and fall for fat storage is essential for winter survival (Mautz 1978). Food sources on summer range are typically greater in quantity and quality than winter range. However, availability of plant functional groups is influenced by precipitation (Bennett and others 1987). Forbs have been reported to dominate the white-tailed deer diet in the Black Hills in the summer (Schenck and others 1972; Sieg and Severson 1996; USDA Forest Service and others 1982); however, in the central Black Hills, Hippensteel (2000) reported populations ate mostly graminoids and moderate amounts of shrubs and forbs during the two summers of her study. She suggests that lower precipitation increased graminoid availability and lowered forb availability.

White-tailed deer utilize a variety of habitat types and structural stages throughout the year for cover and forage (DePerno 1998; Kennedy 1992; Stefanich 1995; Uresk and others 1999). High-quality deer habitat is a function of a variety of habitats within its home range that provides cover and forage. During winter in the central Black Hills, white-tailed deer selected dense young ponderosa pine/deciduous mixed forests and open burned ponderosa pine/grass habitats in greater proportion than available (DePerno 1998). In addition, young and mid-aged open ponderosa pine forests, aspen/coniferous forests, and most structural stages of aspen were

utilized, while mid-aged ponderosa pine stands with moderate to dense canopy coverage and meadow habitats were avoided (DePerno 1998). In contrast, white-tailed deer in the northern Black Hills selected young and mid-aged ponderosa pine stands with high canopy coverage while avoiding the more open ponderosa pine stands within the same diameter class (Kennedy 1992).

During summer in the central Black Hills, white-tailed deer selected ponderosa pine/deciduous, aspen, aspen/coniferous, spruce, and spruce/deciduous habitats, and areas with juniper and shrubs, while avoiding grass/forb habitats and mid-aged ponderosa pine forests with moderate to dense canopy coverage (DePerno 1998). White-tailed deer in the northern Black Hills selected agricultural lands, open young ponderosa pine forests and open aspen/beaked hazelnut habitats while avoiding mid-aged ponderosa pine forests with moderate canopy coverage and pine/grass-forb stands (Kennedy 1992).

Fawning habitat requires horizontal and vertical screening cover to hide young deer from predators. Sites chosen by fawns typically have more vertical and horizontal obstruction for screening cover than those found on randomly selected sites (Uresk and others 1999). In the northwestern Black Hills, white-tailed deer fawns were observed in ponderosa pine stands with 45 percent horizontal screening coverage and in aspen/paper birch stands with 63 percent horizontal coverage (Stefanich 1995). In areas where aspen habitats are limited, such as the northeastern Black Hills, fawn beds were found in relatively open stands of ponderosa pine with a substantial percentage of grass cover and a relatively tall understory (Uresk and others 1999). Therefore, habitats with dense tall shrub/saplings, regardless of habitat type, provide great horizontal cover for fawns (Stefanich 1995).

Management guidelines

Enhancing cover and forage habitats utilized by white-tailed deer in the Black Hills can increase carrying capacity and aid in the stabilization of population levels. Maintenance and enhancement of deciduous forests and coniferous/deciduous mixed forests should be a major element in deer habitat management in the Black Hills. Various silvicultural treatments and prescribed burning can be used to create a variety of habitats that will provide forage and cover for white-tailed deer. Close proximity of forage and cover areas can reduce energy expenditures for travel and foraging. USDA Forest Service and others (1982) recommend that forage areas should be no further than 600 feet

(183 m) from the edge of the cover habitat for all deer habitat.

Deciduous habitats are important to white-tailed deer on both winter and summer range (DePerno 1998; Kennedy 1992). Aspen stands provide horizontal cover for fawning and abundant high-quality understory forage (DePerno 1998; Kennedy 1992; Stefanich 1995). Providing a variety of seral aspen stages will maximize cover and forage diversity (Severson 1982). Removal of some ponderosa pine overstory in areas where remnant aspen occur can increase the deciduous component as well as restore declining aspen stands. Aspen regeneration should be closely monitored, however, to insure that declining clones are recovering. Fencing, or other measures may be needed to allow sprouting to introduce a new age class into the clone before deer are allowed access.

Forage production can be increased through prescribed burning, ponderosa pine stand thinning, and pine litter reduction (DePerno 1998; Hippensteel 2000; Osborn 1994). Prescribed burning of browse species such as chokecherry, serviceberry, and aspen can be beneficial since these species are prolific root or crown sprouters (Wright and Bailey 1982). Young succulent sprouts are usually more nutritious than unburned mature plants (Sieg and Severson 1996). Thinning ponderosa pine stands allows more light to reach the forest floor and increases forage production (Uresk and Severson 1998). In mixed oak-pine stands, selective removal of pine trees may enhance reproduction of oak and associated shrubs (Sieg and Severson 1996).

Precipitation patterns and plant abundance should be monitored on deer summer ranges. In years with lower precipitation, plant functional groups on summer range can change and might have detrimental effects on white tail deer populations, especially in areas that are grazed by livestock and other wild ungulates.

Mule Deer (Odocoileus hemionus)

The current mule deer population for the South Dakota region of the Black Hills is approximately 12,000 (South Dakota Game Report 2000). The majority of mule deer are found in the southern Black Hills (Sieg and Severson 1996). Although white-tailed deer, elk, and mule deer coexist in the Black Hills, they occupy different ecological niches. Unfortunately, most detailed studies of habitat requirements of ungulate populations in the Black Hills focus on white-tailed deer. We have therefore attempted to utilize information on

mule deer habitat requirements from other geographical regions and have applied this information to Black Hills populations.

Habitat requirements

The quality and quantity of food sources on winter range is usually cited as the limiting factor to mule deer populations (Natural Resource Conservation Service 2000; Towry 1984). Mule deer extensively use higher elevation areas of the Black Hills during the summer months, but use the entire Black Hills habitat as year-round range (Wyoming Game and Fish Department 1985). Winter range is usually below 5,200 feet in elevation and often occurs in winter livestock range (Wyoming Game and Fish Department 1985).

Although habitat requirement studies of mule deer are limited for the Black Hills (Wydeven and Dahlgren 1985), some general conclusions can be made based on other geographical studies. Mule deer abundance is highest in shrubby/brushy land, riparian woodland, and at the edge of forest and meadow areas where shrubs are found (Natural Resource Conservation Service 2000). In the Wind Cave National Park, mule deer were most often observed in the shrubby draws, while the use of ponderosa pine habitats was low (Wydeven and Dahlgren 1985). On the Front Range of northern Colorado, mule deer utilize ponderosa pine forests with mountain mahogany understory or open areas where mountain mahogany is dominant (Natural Resource Conservation Service 2000). In the Missouri River Breaks study area in Montana, mule deer were found in ponderosa pine-Rocky Mountain juniper habitat types throughout the year (Mackie 1970). In the Badlands National Park in South Dakota, chokecherry habitat and Rocky Mountain juniper habitat types were used (Steigers 1981). Towry (1984) noted that aspen stands also provide an abundant source of forage for mule deer populations in Colorado.

Mule deer browse on a variety of woody plants, but also graze on forbs and graminoids (Kufeld and others 1973; Towry 1984). Kufeld and others (1973) reported that mule deer diets varied seasonally and depended on the availability of forage. Shrubs and trees make up the majority of mule deer diets throughout the year, especially in the fall and winter. Grasses are grazed in the winter and spring when available. Forbs are also consumed throughout the year, with the highest consumption during the summer and fall when available (Kufeld and others 1973). In the Wind Cave National Park, mule deer fed mostly on browse species such as sumac (*Rhus* spp.) and western snowberry, but also ate moderate amounts of forbs. Forbs were more important

in summer and browse most important in fall and winter (Wydeven and Dahlgren 1985).

Although mule deer cover requirements are not well understood, current guidelines estimate that 40 percent of a deer's use area should be in cover. Hiding cover for mule deer should be capable of hiding 90 percent of the standing adult mule deer at a distance of at least 200 feet (61 m). Broken terrain or topographic features are often used as escape cover by mule deer.

Fawning habitat is often found in areas with ground cover at least 2 to 6 feet (0.6 to 1.8 m) in height. Ground cover can consist of herbaceous vegetation mixed with low shrubs or small trees (Towry 1984). In the Badlands National Park in South Dakota, chokecherry habitat was used for fawning sites and early fawn-rearing activities (Steigers 1981).

Management guidelines

Not all ponderosa pine habitat types in the Black Hills are suitable mule deer habitat. Mule deer use of open shrubby habitats, riparian woodlands, aspen forests, and edges of forest and meadow areas indicate the need to maintain these habitats where they occur within the Black Hills landscape. Since the majority of mule deer diet throughout the year comes from woody plants, enhancement of browse is key to providing sufficient forage. Open habitats and browse availability can be maintained through prescribed burning. Mule deer seem to prefer foraging in burned areas, most likely due to the increased nutrient content and palatability of forage. Fire can be used to stimulate browse, creating openings in dense forests, and create mosaics of forage and cover habitat (Gruell 1986; USDA Forest Service 2002a). Since mule deer share winter range with livestock, care should be taken that these areas are not over grazed and damaged (Wyoming Game and Fish Department 1985).

Providing sufficient security cover for mule deer can help minimize human related disturbance. Close proximity of forage and cover areas can reduce energy expenditures for travel and foraging. Mule deer frequently use draws and riparian zones to travel, indicating the need for maintenance of these areas (Wyoming Game and Fish Department 1985). Guidelines for mule deer hiding cover call for vegetation that is capable of hiding 90 percent of a standing adult deer from the view of a human at a distance of at least 200 feet (61 m) (Thomas and others 1979). Providing patches of young stands and managing these stands with thinning operations that maintain low crowns for extended periods of time can provide suitable security cover. Maintenance of areas with low crowns and ground cover at least 2 to

6 feet (0.6 to 1.8 m) in height will also provide mule deer fawning habitat.

As with elk habitat, use of irregularly spaced, uneven-aged or multiaged management can maintain and enhance ponderosa pine forests for mule deer habitat. The requirement to maintain open stands for shrub production will require the sacrifice of some timber yield, however.

Merriam's Turkey *(Meleagris gallopavo merriami)*

Merriam's turkeys are not native to the Black Hills, but are an important game species. Eight turkeys from New Mexico were introduced into the Black Hills in March 1948 for hunting purposes. In 1950, 15 turkeys from Colorado were released and in 1951, 6 more turkeys from New Mexico were released (Petersen and Richardson 1975). By 2000, the population of turkeys in the Black Hills (South Dakota region only) was estimated to be around 16,000 (South Dakota Game Report 2000).

Habitat requirements

Most turkey populations in the Black Hills begin nesting in late April (Rumble and Hodorff 1993). Nesting attempts are positively related to precipitation from April to June and turkeys may attempt to nest up to three times throughout this period. Nest survival generally increases from first attempt to third attempt. An average turkey clutch size is 9.2 eggs with 86 percent of the eggs hatching successfully (Rumble and Hodorff 1993).

Because reproductive success is one of the many important features for population stability, the quality of nesting habitat can greatly affect the turkey population (Rumble and Hodorff 1993). Successful turkey nests are generally found on slopes 30 to 40 percent regardless of aspect in moderately open ponderosa pine forests with average basal area of 85 ft² per acre (19.4 m² per hectare) and 451 trees per acre (1,116 trees per hectare) (Rumble and Hodorff 1993). Turkeys show preference for nesting in areas that have not been cut or recently thinned (precommercial and commercial) within the past 2 years. Seed-tree cuts are infrequently used and clearcuts are never utilized for nesting habitat. Nest overstory coverage is often greater than 77 percent and often consists of shrubs such as common juniper or the combination of western snowberry, chokecherry, and Wood's rose. Horizontal cover within 5 feet (1.5 m) of the nest sites should be at least 9 inches (23 cm) tall. Required horizontal cover decreases rapidly within the first 16 feet (5 m) from the nest to 4 inches (10 cm) high (Rumble and Hodorff 1993).

Within a few days after eggs hatch, hens often move the poults up to 2 miles (3.5 km) to brood-rearing habitats, even if suitable habitat is close to the nesting site (Rumble and Anderson 1993a). Hens with poults less than 7 weeks old are typically found in natural openings with abundant herbaceous vegetation (Rumble and Anderson 1993a). Meadows with herbaceous vegetation provide a high abundance of invertebrates (Rumble 1990; Rumble and Anderson 1993a, 1996a), which provide large amounts of protein required by poults for proper growth and survival (Robbins 1983). Meadows range in size from 5 to 465 acres (2 to 188 ha) and are usually within 33 feet (10 m) of a dense (greater than 100 ft² per acre; 23 m² per hectare) forest (Rumble and Anderson 1993a). Although young poults usually use western snowberry for hiding cover, older poults (more than 7 weeks) utilize the dense forested stands (Rumble and Anderson 1993a, 1996b).

From late spring to summer, turkeys forage in ponderosa pine habitats with low basal areas and large amounts of understory cover consisting of grasses, forbs, and shrubs (Rumble and Anderson 1996b). Turkeys are also found in aspen/paper birch habitats, which contain an abundance of grasses, forbs, and shrubs (Rumble and Anderson 1993b). From May to September, turkeys consume herbaceous foliage, grass seed, forb seed, forb flowers, and insects (Rumble and Anderson 1996a). Smooth brome seeds dominate the diet from June to September. Peak consumption of forb seeds and flowers occurs during periods of flowering in spring and autumn. Insect consumption increases during early summer and declines after August. In times of drought, turkeys will increase their consumption of soft mast and arthropods (Rumble and Anderson 1996a). In years of good bur oak mast production, turkeys can be found in bur oak habitats for 2 to 4 weeks in September and October.

In early fall, turkeys can still be found in the aspen/paper birch habitat, as well as open ponderosa pine habitats (Rumble and Anderson 1993b). By October, however, turkeys move to dense ponderosa pine habitats with overstory canopy cover greater than 71 percent and basal areas greater than 140 ft² per acre (32 m² per hectare). From October to April, ponderosa pine seed and bearberry seeds, which can be found in great abundance under dense ponderosa pine habitats, make up the majority of a turkey's diet (Rumble and Anderson 1993b, 1996b). In years of a poor ponderosa pine cone crop, a greater proportion of bearberry fruit and grass leaves and seeds comprise the turkey diet throughout the winter (Rumble and Anderson 1996a), and stands with lower basal area and more understory vegetation are preferred.

Roosting trees provide perches and a resting place for turkeys, and are an important habitat component for sustaining turkey populations (Hoffman and others 1993; Rumble 1992). In the Black Hills, turkeys select roost sites that have trees with large d.b.h., low tree density, and high basal area (Rumble 1992). Roost trees averaged 13.8 inches (35 cm) d.b.h., but trees 9 inches (23 cm) d.b.h. or greater were used. Average basal area of roost sites ranged from 82.8 ft² per acre to 109 ft² per acre (19 to 25 m² per hectare). Trees with layered horizontal branches spaced at 2- to 3-foot intervals that allow easy access for turkeys were characteristic of roost trees (Hoffman and others 1993; Rumble 1992). Roosts were usually located near the tops of slopes or on ridges. Roosts with western and northern aspects were located on the lower portions of slopes, while roosts on eastern or southern aspects were found on upper slopes. Selected roost sites had limited or no recent timber activity within the past 5 years (Rumble 1992).

Management guidelines

The key to providing Merriam's turkey habitat is to have a diversity of structural and stocking conditions within the Black Hills ponderosa pine landscape. Merriam's turkeys utilize different structural habitats for nesting, brood rearing, roosting, and foraging. Broods are reared in open meadows adjacent to dense ponderosa pine forests. Nesting habitat is found in open stands, with an understory component that provides cover. Summer habitats are found in open ponderosa pine and aspen stands with low canopy coverage. Winter habitats typically consist of dense, mature ponderosa pine stands with high canopy coverage. The use of these different habitats by Merriam's turkey throughout the year enables managers to utilize various silvicultural treatments to manage and create the structural stages needed.

Rumble and Hodorff (1993) suggest that nest site availability does not appear to limit Merriam's turkey populations in the Black Hills, and that specific management to enhance nest site availability is not necessary. However, suitable nesting habitat consisting of moderately open ponderosa pine forests with average basal area of 85 ft² per acre (19.4 m² per hectare) and a shrubby understory should be maintained.

Providing suitable habitat for poults is key for maintaining turkey populations. Poults younger than 7 weeks old prefer meadows and openings with large amounts of herbaceous vegetation. Hoffman and others (1993) suggest that the openings should be long and narrow with an irregular edge, and not exceed 240 feet (73 m) in width. Dense ponderosa pine stands

(greater than 100 ft² per acre or 23 m² per hectare) should be left along the meadow edges to provide escape and loafing cover for the hens (Rumble and Anderson 1993a). Trees encroaching into small openings or natural meadows and slash from a created opening should be removed promptly to encourage quick herbaceous vegetation recovery. Because poult require at least 1,200 lb per acre (1,345 kg per hectare) of herbaceous vegetation along forest/meadow edges (Rumble and Anderson 1996c), grazing by livestock should be monitored (Rumble and Anderson 1993a). Grazing should not exceed 50 percent of the herbaceous vegetation, and be prohibited during drought years. A rest-rotation grazing system is recommended because moderate grazing does stimulate new herbaceous growth, but re-entry into a previously rested pasture should be delayed until July 15 (Hoffman and others 1993).

Summer habitats should have low basal area to maintain high production levels of grasses, forbs, and shrub. Ponderosa pine stands with basal area around 61 to 78.4 ft² per acre (14 to 18 m² per hectare) are consistent with habitats chosen by turkeys without poults (Rumble and Anderson 1993b). Although the use of seed-tree regeneration harvests, which typically leave basal areas less than 39 ft² per acre (9 m² per hectare), should not impact summer habitats (Rumble and Anderson 1996a), care should be taken not to degrade nesting habitat (Rumble and Hodorff 1993).

Most limiting to the turkey population in the Black Hills is the availability of winter habitat. Winter habitats should be abundant and well distributed at the lower elevations and on southern aspects because snow depth limits the turkey's foraging ability. Ponderosa pine trees that are greater than 9.8 inches (>25 cm) d.b.h. provide better winter habitat than smaller trees with comparable overstory cover (Rumble and Anderson 1995). Silvicultural management that reduces stands to basal areas less than 78 ft² per acre (18 m² per hectare) will decrease the quality of winter habitat (Rumble and Anderson 1996b). Providing some ponderosa pine habitat with overstory canopy coverage greater than 70 percent and basal area greater than 139 ft² per acre (>32 m² per hectare) will ensure quality winter habitat (Rumble and Anderson 1993b). Mature, seed-bearing ponderosa pine trees should be maintained to provide winter forage.

Management of roost sites should be based on roost sites, rather than individual roost trees. Roosting habitat should be dispersed throughout the forest and can be located in dense winter habitat (Rumble 1992). Timber management that reduces stands to growing stock levels of less than 96 ft² per acre (22 m² per hectare) and tree diameters of less than 9.8 inches (<25 cm)

will degrade roosting habitat. Maintaining portions of the forest with basal areas greater than 91.5 ft² per acre (21 m² per hectare) with tree diameters that average from 9.8 to 13.8 inches (25 to 35 cm) should be adequate for roosting habitat. Mixed ponderosa pine/white spruce forests greater than 9.8 inches (>25 cm) d.b.h. and greater than 40 percent canopy cover also provide suitable roosting habitat (Rumble and Anderson 1995). Stands managed for roost sites should be situated on the upper third of a slope with layered horizontal branches (Rumble 1992) and be located on easterly aspects (Hoffman and others 1993).

Enhancement of deciduous tree habitats, such as aspen and bur oak, is important for nesting habitat, escape cover, and forage. Mature bur oak stands provide nutritious acorn mast. These bur oak stands should be maintained at basal areas greater than 35 ft² per acre (> 8 m² per hectare) and should be adjacent to ponderosa pine stands that have basal areas greater than 80 ft² per acre (>18.4 m² per hectare). Restoration of aspen stands will increase forage and provide sufficient escape and nesting cover.

High road densities and the frequent use of them by humans can cause turkeys to abandon some habitats. Hoffman and others (1993) suggest that roads should be eliminated from meadows wherever possible, and necessary new roads placed along the edge of openings to minimize disturbance.

Northern Goshawk (*Accipiter gentiles atricapillus*)

The northern goshawk occurs throughout North America in coniferous and mixed forests. Concern that the goshawk population may be declining in the Western United States has lead the U.S Forest Service to designate the goshawk as a sensitive species and management indicator species for the Black Hills as well as other areas in the West (Reynolds and others 1992; USDA Forest Service 1996c, 2001b). As a result of this designation, management recommendations for the northern goshawk were developed for the Southwestern Region of the Forest Service (Arizona, New Mexico) by a scientific committee (Reynolds and others 1992). These authors identified three critical components of goshawk nesting home range (about 6,000 acres; 2,400 ha): nest area (30 acres; 12 ha), post-fledging family area (420 acres; 170 ha), and foraging area (5,400 acres; 2,185 ha) (Reynolds and others 1992). This forage-based habitat management guideline identifies forest structural characteristics for goshawk nesting, post-fledging, and foraging habitats (Reynolds and others 1992).

One concern is that the management recommendations for the goshawk in the Southwestern United States are applicable only to forests for that region. Specifically, ponderosa pine in the Southwestern United States differs from the Black Hills ponderosa pine forest in structure, disturbance regimes, precipitation patterns, and site productivity. Furthermore, only two studies have briefly investigated goshawk ecology in the Black Hills (Bartelt 1977; Erickson 1987). Adaptation of southwestern guidelines to fit the Black Hills ecosystem is needed as well as further studies on goshawk ecology in the Black Hills to ensure proper management.

Habitat requirements

Nesting habitat has been suggested as an important component of goshawk management (USDA Forest Service 2000, 2001b). The goshawk nest area is approximately 30 acres (12 ha) and may include more than one nest (Reynolds and others 1992). Suitable nesting habitat for the goshawk is essential for reproductive success since goshawks have high site fidelity and may still use lower quality habitat and not produce young (USDA Forest Service 2000). In addition, all of the actions and behaviors associated with courtship, incubation, and the nestling stage occur in the nest area (Kennedy and others 1994; Reynolds and others 1992).

A number of factors have been identified for goshawk nest site selection in the Black Hills (Bartelt 1977; Erickson 1987). Goshawk nest sites in the Black Hills have been found in mature ponderosa pine, white spruce, or near stands of maturing aspen, but only mature ponderosa pine were actually selected as nest trees (Bartelt 1977; Erickson 1987). Tree sizes at nest sites ranged from 8 to 20 inches (20 to 51 cm) d.b.h. with a mean of 16 inches (41 cm) (Bartelt 1977; Erickson 1987). Nest trees ranged from 12 to 23 inches (31 to 58 cm) and were usually the largest trees in the stand (Erickson 1987). Nest site locations contained over 120 ft² per acre (27.5 m² per hectare) basal area with average canopy closure of 70 percent. Although nest sites were found in dense forests, they are often in close proximity to forest openings or roads. Bartelt (1977) reported approximately half of the nests he located were within 33 ft (10 m) of a 0.125 to 0.250 acre (0.05 to 0.10 ha) opening. Most nest sites were on gentle (less than 15 percent) west to northeast slopes, above elevations of 5,085 feet (1,550 m), within 0.75 mile (1.25 km) of open water, and included dead and down woody material (Bartelt 1977; Erickson 1987).

The impact of human activities such as logging, tourism, and road traffic on nesting habitat in the Black

Hills is unclear. The timing of human activity near nest sites is likely related to the disturbance of nesting goshawks. Nest abandonment seems to increase with an increasing amount of disturbance, and the probability of nest abandonment is greatest just before, during, or just after incubation (Bartelt 1977). Goshawks establish their nest sites in the Black Hills around mid- to late March. Eggs are laid around mid- to late April, and eggs hatch around late May. Young goshawks fledge in the Black Hills around mid- to late July, sometimes later if a nest was reinitiated because the first nesting attempt failed (Bartelt 1977).

The largest impact from logging on the nest site would be from the change in forest structure surrounding the nest area (Bartelt 1977). On the North Kaibab Plateau of Arizona, Reynolds (USDA Forest Service 2000) stated that goshawk fecundity decreased when seed-tree cuts were within one-half mile (0.8 km) of a goshawk nest. The use of large machinery in logging operations might disturb goshawk nesting success, although a study in northern Arizona reported that logging trucks more than 1,300 feet (>400 m) from a nest site did not disturb the nesting goshawk (Grubb and others 1998).

The post-fledging family area (PFA) encompasses approximately 420 acres (170 ha) surrounding the nest area (Reynolds and others 1992) and is used by the goshawk from the time the young leave the nest until they are not reliant on the parents for food (up to 2 months). The PFA provides hiding cover for goshawk fledglings while providing habitat for prey and foraging opportunities for the adults. The PFA contains a diversity of seral stages that provide patches of dense, large trees intermixed with small trees and shrubby understories for hiding cover. In addition, the well-developed understory and down woody debris provide habitat for prey populations. The PFA for Black Hills goshawks has not yet been determined, so guidelines from other regions must be used until further studies can be completed.

The tenet of management for foraging areas is based on providing habitat for the goshawk's prey species. The southwestern goshawk guidelines call for the foraging area to surround both the nest area and the PFA, and be approximately 5,400 acres (2,185 ha) (Reynolds and others 1992). The foraging area comprises a variety of landforms, forest canopy types, and vegetation structural stages. Goshawks need an open understory to detect and capture prey from their tree perches (Reynolds and others 1992). Snags, downed logs, woody debris, openings, large trees, and herbaceous and shrubby understories are all required to provide critical habitat

for the various birds, small mammals, and invertebrates eaten by the goshawk (Reynolds and others 1992). Providing habitat for the prey and the goshawk will require a mosaic of interspersed vegetative structural stages over a large area.

In the Black Hills, goshawks have been reported to eat insects, rodents, rabbits, and birds (Bartelt 1977; Erickson 1987). Specific prey include: red squirrels (*Tamiasciurus hudsonicus*), least chipmunks (*Tamias minimus*), thirteen-lined ground squirrels (*Spermophilus tridecemlineatus*), bushy-tailed woodrats (*Neotoma cineria*), voles (*Microtus* spp.), white-tailed jackrabbits (*Lepus townsendii*), Nuttall's cottontails (*Sylvilagus nuttallii*), ruffed grouse (*Bonasa umbellus*), northern flickers (*Colaptes auratus*), black-backed woodpeckers (*Picoides arcticus*), three-toed woodpeckers (*Picoides tridactylus*), American robins (*Turdus migratorius*), Townsend's solitaire (*Myadestes townsendi*), black-headed grosbeaks (*Pheucticus melanocephalus*), evening grosbeaks (*Coccothraustes vespertinus*), gray jays (*Perisoreus canadensis*), dark-eyed juncos (*Junco hyemalis*), and Merriam's turkey poults (Bartelt 1977; Erickson 1987; USDA Forest Service 2000).

Management guidelines

Management guidelines for the goshawk in southwestern ponderosa pine and mixed-species forests are described in great detail in Reynolds and others (1992). Included in the guidelines are attributes of stand conditions for nest sites, post-fledging family area, and foraging area, and how to obtain these features with silvicultural techniques. Additional guidelines are given concerning habitat attributes required for several goshawk prey species. Similar assessments are needed for optimum management of goshawk populations in the Black Hills. Recommendations for management of goshawk habitat in the Black Hills are currently under review (USDA Forest Service 2001b).

A balance of coniferous forest structural stages across the entire landscape was suggested as a long-term solution for providing prey and goshawk habitat in the Southwestern United States (Reynolds and others 1992) and for the Black Hills region (USDA Forest Service 2000). The examination of historic conditions and the range of natural variability of ponderosa pine distribution can aid in defining this balance of structural stages for the Black Hills region (USDA Forest Service 2000).

Continuous management will be needed to obtain and maintain the various structural conditions the goshawk requires for all three of the critical habitat components. Management of large areas of the same structural stage should be avoided. Instead, Reynolds and

Graham (personal communication, 2001) feel that small groups and clumps of trees are a critical component of the spatial heterogeneity needed for properly functioning goshawk habitat, regardless of the region. These patches and clumps should vary in size, shape, and structural stage to mimic natural vegetative patterns and patch sizes (USDA Forest Service 2000). Retention of at least one green ponderosa pine tree greater than 20 inches (>51 cm) d.b.h. per acre will aid in providing snags and large-diameter coarse woody debris for prey species.

Maintaining small groups of trees in the desired mix of what Reynolds calls Vegetation Structural Stages (VSSs) will be especially difficult in the higher structural stages that require high canopy closure (Shepperd and others 2001). Several issues must be considered: first, the time needed for a stand to grow from one VSS class to another will vary depending on stocking levels and site class. Second, trees in open-spaced groups or widely spaced clumps will grow into larger VSS classes quicker than those in denser clumps. Also, small open spaces are conducive to rapid seedling establishment in the Black Hills. These seedlings will create a dense sapling understory, which is not always desirable for goshawk habitat. Managing ponderosa pine forests for goshawk habitat will thus take continuous silvicultural activity to establish and maintain the desired mix of VSSs.

Uneven-aged group and irregularly spaced individual-tree selection systems can be well suited to accomplish and maintain these goals. Uneven-aged silviculture systems offer the flexibility to create and monitor conditions over large areas while minimizing the need for delineation and record keeping (Shepperd and others 2001). Under uneven-aged management, each goshawk territory could be designated as a site, or audit unit, for record-keeping purposes. Stocking growth projections of all VSSs in the unit could be handled as one stand. However, even-aged management of a diverse goshawk habitat would require breaking each group or VSS into a separate stand. Such stands might need to be as small as a fraction of an acre to meet goshawk diversity requirements. Breaking the landscape into many small even-aged stands to accomplish goshawk management objectives would not guarantee that stocking over the entire habitat would be regulated, and would infinitely complicate record keeping and database management.

Nongame Birds

The Black Hills is host to numerous bird species that are typically found in the Eastern, Western, and

Northern United States. The location of the Black Hills in the Northern Great Plains allows this area to host over 190 species of birds that are either migrants or permanent year-round residents (American Bird Association 1994). Haldeman (1980) identified some 111 species of nongame birds in the Black Hills National Forest, and developed habitat matrices based on vegetation type and successional stages within these types. Since bird communities are influenced by vegetative structure, several studies have investigated the effects of timber harvests on bird populations in the Black Hills (Dykstra 1996; Dykstra and others 1999; Mills 1994). These studies have aided in the development of habitat capability models (HABCAP) for the Black Hills National Forest to help predict consequences of land management decisions on both resident birds (Mills and others 1996) and wintering birds (Rumble and others 1999).

Composition and structure of vegetative communities influence bird species richness (in other words, the number of species). In the Black Hills, bird species richness is highest in stands of pure aspen and aspen with limited ponderosa pine (less than 10.5 ft² per acre; <2.4 m² per hectare), while stands of pure ponderosa and stands dominated by ponderosa pine with some aspen have slightly lower bird species richness (Mills 1994; Mills and others 2000; Rumble and others 2001).

Although species richness was not shown to differ significantly among ponderosa pine structural stages, bird composition does differ among the structural stages (Dykstra 1996; Dykstra and others 1999; Mills 1994; Mills and others 2000). Individual bird species are adapted to specific habitat structures that allow successful foraging and nesting. Timber harvests that reduce the amount of tree foliage within a forest can negatively affect birds that depend on crown vegetation for nesting and foraging (Szaro and Balda 1979). However, the same process of losing foliage and biomass to timber harvests can benefit other species of birds that prefer open or edge habitats. Thus, understanding how tree density and coverage influence bird abundance will allow land managers to manipulate forested stands to encourage or discourage the presence of different bird species (Dykstra 1996).

The following species discussions include some representative ground, foliage, and cavity nesters that utilize ponderosa pine, aspen, and mixed ponderosa pine/aspen habitats within the Black Hills, although some bird species do use white spruce habitats. Residency and possible HABCAP coefficient references for the bird species discussed in the following section are listed in table 4. Vegetative structural stages listed in the following bird discussion are based on the Buttery and

Gillam (1984) classification scheme; refer to table 3 for diameters and crown coverage values, as well as a crosswalk for current structural guidelines now applied in the Black Hills National Forest and other Forest Service regions.

Ground nesters

Dark-eyed Junco—The dark-eyed junco (*Junco hyemalis*) is found in all structural stages of ponderosa pine and aspen/paper birch forests throughout the year, but abundance is highest in open-canopied forests of ponderosa pine (3A and 4A) and aspen (3A) (Dykstra and others 1999; Mills 1994; Mills and others 1996, 2000; Rumble and others 2001). Abundance of dark-eyed juncos declines in both forested habitats as overstory cover increases (Mills and others 1996, 2000). During the summer, abundance in the shrub-seedling (2) stage of aspen/paper birch is high. In the winter, however, use declines significantly (Rumble and others 1999). Dark-eyed juncos build their nests on the ground or occasionally up to 8 feet (2.4 m) above ground in a shrub or tree. Nests on the ground are often concealed under herbaceous vegetation or large-diameter woody debris. Most food is obtained from the ground and includes some insects, but mostly seeds (DeGraaf and others 1991).

MacGillivray's Warbler—The MacGillivray's warbler (*Oporornis tolmiei*) is considered an aspen/paper birch obligate (Mills and others 2000; Rumble and others 2001) and prefers open canopied stands or low- to intermediate-density forests with a shrub understory (DeGraaf and others 1991). Abundance is highest in the sapling-pole (3A) aspen/paper birch habitat, with moderate use of sapling-pole (3B) and shrub/seedling (2) aspen/paper birch stands (Mills and others 1996, 2000). This warbler also uses habitats with serviceberry, snowberry, ninebark, wild spirea, and riparian willows (*Salix* spp.) and alders (*Alnus* spp.) (DeGraaf and others 1991). Ponderosa pine stands and meadows were not chosen by the MacGillivray's warbler (Mills and others 1996). The MacGillivray's warbler builds nests 2 to 5 feet (0.6 to 1.5 m) off the ground attached to plant stems in brushy habitat. It forages close to the ground, eating insects gleaned from vegetation (DeGraaf and others 1991).

Ovenbird—Ovenbirds (*Seiurus aurocapillus*) are found mostly in aspen/paper birch habitats and in ponderosa pine stands with deciduous components (Dykstra and others 1999; Mills 1994; Mills and others 2000; Rumble and others 2000b, 2001). In aspen/paper birch habitats, abundance is highest in sapling-pole (3B and 3C) stands with moderate to high

Table 4: Residency and available habitat capability (HABCAP) models for birds found in the Black Hills, South Dakota.

Common name	Scientific name	Residence	HABCAP- summer	HABCAP- winter
Ground nesters				
Dark-eyed junco	<i>Junco hyemalis</i>	Permanent ^a	Mills and others 1996	Rumble and others 1999
MacGillivray's warbler	<i>Oporornis tolmiei</i>	Neotropical ^b	Mills and others 1996	n/a
Ovenbird	<i>Seiurus aurocapillus</i>	Neotropical ^b	Mills and others 1996	n/a
Virginia's warbler	<i>Vermivora virginiae</i>	Neotropical ^c	Not developed	n/a
Townsend's solitaire	<i>Myadestes townsendi</i>	Permanent ^a	Mills and others 1996	Rumble and others 1999
Foliage nesters				
Gray jay	<i>Perisoreus canadensis</i>	Permanent ^a	Mills and others 1996	Rumble and others 1999
Yellow-rumped warbler	<i>Dendroica coronata</i>	Summer ^a	Mills and others 1996	n/a
Western tanager	<i>Piranga ludoviciana</i>	Neotropical ^a	Mills and others 1996	n/a
Chipping sparrow	<i>Spizella passerina</i>	Summer ^a	Mills and others 1996	n/a
Pine siskin	<i>Carduelis pinus</i>	Permanent ^a	Not developed	Not developed
Red crossbill	<i>Loxia curvirostra</i>	Winter ^a	n/a	Rumble and others 1999
Swainson's thrush	<i>Catharus ustulatus</i>	Neotropical ^a	Not developed	n/a
Ruby-crowned kinglet	<i>Regulus calendula</i>	Neotropical ^a	Mills and others 1996	n/a
American robin	<i>Turdus migratorius</i>	Summer ^a	Mills and others 1996	n/a
Dusky flycatcher	<i>Empidonax oberholseri</i>	Neotropical ^b	Mills and others 1996	n/a
Warbling vireo	<i>Vireo gilvus</i>	Neotropical ^a	Mills and others 1996	n/a
Cavity nesters				
Brown creeper	<i>Certhia americana</i>	Permanent ^a	Not developed	Not developed
Mountain bluebird	<i>Sialia currucoides</i>	Summer ^a	Mills and others 1996	n/a
Black-capped chickadee	<i>Parus atricapillus</i>	Permanent ^a	Mills and others 1996	Rumble and others 1999
Northern flicker	<i>Colaptes auratus</i>	Summer ^a	Mills and others 1996	n/a
Lewis' woodpecker	<i>Melanerpes lewis</i>	Summer ^c	Not developed	n/a
Black-backed woodpecker	<i>Picoides arcticus</i>	Permanent ^a	Not developed	Not developed
Downy woodpecker	<i>Picoides pubescens</i>	Permanent ^a	Not developed	Not developed
Three-toed woodpecker	<i>Picoides tridactylus</i>	Permanent ^a	Not developed	Not developed
Hairy woodpecker	<i>Picoides villosus</i>	Permanent ^a	Mills and others 1996	Rumble and others 1999
Red-naped sapsucker	<i>Sphyrapicus nuchalis</i>	Summer ^a	Mills and others 1996	n/a
Red-breasted nuthatch	<i>Sitta canadensis</i>	Permanent ^a	Mills and others 1996	Rumble and others 1999
White-breasted nuthatch	<i>Sitta carolinensis</i>	Permanent ^a	Mills and others 1996	Rumble and others 1999
Pygmy nuthatch	<i>Sitta pygmaea</i>	Permanent ^c	Not developed	Not developed
Others				
Brown-headed cowbird	<i>Molothrus ater</i>	Summer ^a	Not developed	n/a

^a South Dakota Ornithologists' Union (1991).

^b Ehrlich and others (1988).

^c DeGraaf and others (1991).

canopy coverage (Mills 1994; Mills and others 1996, 2000). Ovenbirds are also found in mature (4C) and multistoried/old growth (5) ponderosa pine stands with a deciduous component (Dykstra and others 1999; Mills 1994; Mills and others 2000; Rumble and others 2001). Ovenbirds build nests that look like a Dutch oven. These nests are often located in a small depression in the ground near the roots of a tree or by the side of a fallen log. It gleans insects, small snails, slugs, earthworms, and spiders from the forest floor (DeGraaf and others 1991).

Virginia's Warbler—A breeding population of the Virginia's warbler (*Vermivora virginiae*) was recently found in the southwestern region of the Black Hills. This population is the most northeasternmost known

breeding population and extends the breeding range by over 125 miles (200 km). The birds were found in a pine-juniper-shrub habitat that was relatively dense with shrubs, steep, and among medium to large ponderosa pine trees (Swanson and others 2000). Virginia's warblers are ground nesters, and often conceal the nest with overhanging vegetation (DeGraaf and others 1991). Insects are foraged on the ground and foliage. Insects are also caught by hawking, a foraging technique in which a bird attacks from a perch on short flights to capture flying prey (DeGraaf and others 1991).

Townsend's Solitaire—Townsend's solitaires (*Myadestes townsendi*) utilize the sapling-pole, mature, and multistoried/old growth structural stages of

ponderosa pine forests, but their habitat preference changes between the summer breeding season and winter (Mills and others 1996; Rumble and others 1999). During the summer, abundance is greatest in the sapling-pole (3A, 3B, and 3C) stage of ponderosa pine, moderate in the mature (4A, 4B, and 4C) stands, and lowest in the multistoried/old growth (5) stage of ponderosa pine (Dykstra and others 1999; Mills and others 1996, 2000). However, in winter, abundance is greatest in mature (4A and 4B) stands, moderate in multistoried/old growth (5) and mature (4C) stands, and low in sapling-pole (3A, 3B, and 3C) ponderosa pine stands (Rumble and others 1999). Abundance in aspen stands is low; aspen provides low habitat capability in both summer and winter (Mills and others 1996, 2000; Rumble and others 1999). Townsend's solitaires build nests on the ground, often hidden at the base of a tree or among the roots of a fallen tree. During the summer, the Townsend's solitaire hawks for flying insects or gleans food from the ground, foliage, and fruiting stems. During the winter, juniper berries, seeds, and other fruits are eaten (DeGraaf and others 1991).

Foliage Nesters

Gray Jay—Gray jays (*Perisoreus canadensis*) utilize the sapling-pole, mature, and multistoried/old growth structural stages of ponderosa pine forests, but their habitat preference changes between the summer breeding season and winter (Mills and others 1996; Rumble and others 1999). In the summer, optimal habitat includes mature (4B and 4C) and multistoried/old growth (5) ponderosa pine (Mills and others 1996, 2000). In the winter, optimal habitat for gray jays is found in the mature (4C) ponderosa pine stands, although sapling-pole (3C) and mature (4B) ponderosa pine stands are utilized (Rumble and others 1999). Nests are typically well concealed under heavy cover near the trunk about 10 to 30 feet (3 to 9 m) above the ground (DeGraaf and others 1991). Gray jays are omnivorous and eat insects, conifer seeds, berries, young birds, small mammals, lichens, fungi, and carrion (DeGraaf and others 1991).

Yellow-rumped Warbler—Yellow-rumped warblers (*Dendroica coronata*) are one of the most common species within all ponderosa pine structural stages (Dykstra and others 1999; Mills 1994; Mills and others 2000), but abundance declines as coniferous tree density and overstory cover decreases (Mills and others 1996, 2000). Abundance is low in the aspen/paper birch habitats (Mills and others 1996), although this habitat is utilized more frequently when a small component of ponderosa pine is present (Rumble and

others 2001). The yellow-rumped warbler builds its nest away from the tree bole on a horizontal branch about 15 to 20 feet (4.6 to 6.1 m) off the ground. The nest is usually screened from above by clumps of needles. Food is obtained by gleaning insects from foliage or hawking (Szaro and Balda 1979).

Western Tanager—In the northern Black Hills, the western tanager (*Piranga ludoviciana*) is most common in mature (4B and 4C) and multistoried/old-growth (5) ponderosa pine forests (Mills and others 2000). In the central Black Hills, they prefer multistoried (5) ponderosa pine stands, partially due to the bur oak and aspen/paper birch understory (Mills and others 2000). Moderate numbers of western tanagers are found in sapling-pole (3B) and mature (4A and 4B) ponderosa pine stands (Mills and others 1996, 2000). Western tanagers are also found in aspen habitats with a small amount of ponderosa pine (Rumble and others 2001). Western tanagers are sensitive to fragmentation and are typically absent from stands less than 25 acres (<10 ha) in size (Rumble and others 2000). Nests are built away from the trunk on a horizontal branch. Its main diet is insects either gleaned from foliage or caught in the air (DeGraaf and others 1991).

Chipping Sparrow—Chipping sparrows (*Spizella passerina*) live in both ponderosa pine and aspen/paper birch habitats (Rumble and others 2001). Abundance is highest in sapling-pole (3A, 3B) and mature (4A, 4B) ponderosa pine stands and aspen/paper birch (2, 3A and 3B) habitats (Mills and others 2000). Abundance decreases in habitats with dense canopy coverage (greater than 70 percent) and multistoried/old growth (5) ponderosa pine stands (Dykstra and others 1999; Mills and others 2000). Nests are typically built in trees about 3 to 10 feet (0.9 to 3 m) above ground and are well concealed by the foliage. Chipping sparrows are ground foragers, typically found in open meadows gleaning insects and seeds (DeGraaf and others 1991).

Pine Siskin—Pine siskins (*Carduelis pinus*) inhabit ponderosa pine and mixed ponderosa/aspen forests in the Black Hills (Rumble and others 2001). Pine siskins have been observed in sapling-pole (3A and 3B), mature (4B and 4C), and multistoried/old growth (5) ponderosa pine stands (Mills 1994; Dykstra and others 1999). Nests are usually covered with foliage on a branch about 15 to 35 feet (4.6 to 10.7 m) off the ground. Pine siskins glean insects and seeds from the ground, foliage, and off bark (DeGraaf and others 1991).

Red Crossbill—The red crossbill (*Loxia curvirostra*) is a short-distance migrant that breeds in the Black Hills during the winter (late December to March) (South

Dakota Ornithologists' Union 1991). The red crossbill relies heavily on ponderosa pine seed for its food. Although ponderosa pine forests with 71 to 100 percent canopy coverage have greater seed production than less dense forests (Rumble and Anderson 1996b), habitat suitability was shown to decline slightly as canopy coverage increases above 40 percent in the sapling-pole structural stage (Rumble and others 1999). Mature (4A, 4B, 4C) ponderosa pine forests provide some suitable habitat. Aspen/paper birch habitats with less than 70 percent overstory canopy coverage are unsuitable habitats. Moderate habitat capability for the aspen/paper birch habitat, with 71 to 100 percent canopy coverage, depends on the presence of ponderosa pine (Rumble and others 1999, 2001). Nests are built on branches of coniferous trees where suitable seed crops are available (DeGraaf and others 1991).

Swainson's Thrush—The Swainson's thrush (*Catharus ustulatus*) is found in both ponderosa pine and aspen/paper birch habitats of the Black Hills (Mills and others 2000; Rumble and others 2001). Abundance is limited to mature (4B) and multistoried/old-growth (5) ponderosa pine stands and sapling-pole (3A, 3B, 3C) aspen/paper birch stands (Dykstra and others 1999; Mills and others 2000). Nests are built 2 to 20 feet (0.61 to 6.1 m) off the ground near the bole of a tree on a horizontal branch. Its diet consists of insects and small fruits, which are gleaned from the forest floor, foliage, and branches (DeGraaf and others 1991).

Ruby-crowned Kinglet — Optimal habitat for the ruby-crowned kinglet (*Regulus calendula*) in the Black Hills occurs in mature (4C) and old-growth ponderosa pine (5) structural stages. Sapling-pole (3A, 3B, and 3C) ponderosa pine structural stages were found to be unsuitable habitat (Mills and others 1996). These birds are also found in aspen/paper birch habitats with moderate to dense canopy cover (Mills and others 1996) and in aspen/paper birch habitats with a small component of ponderosa pine (Rumble and others 2001). Nests are usually suspended from a branch 15 to 60 feet (4.6 to 18.3 m) above the ground. Insects and spiders are its main source of food, and are obtained by gleaning or hawking (DeGraaf and others 1991).

American Robin—American robins (*Turdus migratorius*) live in both ponderosa pine and aspen/paper birch habitats in the Black Hills (Mills and others 2000; Rumble and others 2001). Sapling-pole (3A and 3C) aspen, sapling-pole (3B) ponderosa pine, and mature (4B) ponderosa pine stands are used frequently (Dykstra and others 1999; Mills and others 2000), but aspen sapling-pole (3B) stands are considered optimal habitat (Mills and others 1996, 2000). Although meadows do not provide suitable nest sites, they are used

extensively for feeding (Mills and others 1996). Thus, the juxtaposition of meadows and suitable nesting sites is an important factor to consider. Nests are built in trees or shrubs, and mud is required for nest materials. American robins are ground foragers that eat both fruits and insects (DeGraaf and others 1991).

Dusky Flycatcher—The dusky flycatcher (*Empidonax oberholseri*) is considered an aspen/paper birch obligate (Mills and others 2000) and prefers open canopied stands or low to intermediate density forests with a shrub overstory (DeGraaf and others 1991). The shrub-seedling (2) stage of aspen is considered optimal habitat, although all structural stages of aspen provide suitable habitat (Mills and others 1996, 2000). Ponderosa pine stands and meadows are unsuitable habitat for the dusky flycatcher (Mills and others 1996; Rumble and others 2001). This bird usually nests in the lower portion of a tree, and eats flying insects (DeGraaf and others 1991).

Warbling Vireo—This aspen/paper birch obligate inhabits all aspen structural stages (Mills 1994; Mills and others 2000). The warbling vireo (*Vireo gilvus*) is also observed in all structural stages of ponderosa pine, but most often when the pine is associated with a small inclusion of aspen (Mills and others 1996; Rumble and others 2001). This bird nests and gleans most of its food (insects) from the mid to upper branches of deciduous trees. Nests are built away from the trunk and usually over 20 feet (greater than 6.1 m) above ground (DeGraaf and others 1991).

Cavity nesters

Brown Creeper—The brown creeper (*Certhia americana*) is classified as a sensitive species by the U.S. Forest Service Rocky Mountain Region (USDA Forest Service 2002b). Brown creepers select stands that are dense and have large-diameter trees (Dykstra and others 1999; Mills and others 2000; Rumble and others 1999, 2000). Even in stands within the same size category, brown creepers will choose stands with higher densities of large diameter (greater than 15 inches; >38 cm) ponderosa pine (Rumble and others 2000). In the northern Black Hills, brown creepers occur in high abundance in multistoried/old growth (5) ponderosa pine stands (Dykstra and others 1999). In the central Black Hills, brown creepers are observed in mature (4A, 4B, 4C) and multistoried (5) ponderosa pine stands, but abundance is highest in the mature (4B and 4C) ponderosa pine stands (Mills and others 2000). The brown creeper builds nests between loose bark and the trunk of a live, dead, or dying tree, but will sometimes utilize a natural or old woodpecker cavity. Most food

is obtained by exploring tree trunks and branches for insects and larvae (DeGraaf and others 1991).

Mountain Bluebird—Optimal habitat for the mountain bluebird (*Sialia currucoides*) is found in open habitats such as sapling-pole (3A) and mature (4A) ponderosa pine stands, as well as shrub-seedling (2) and sapling-pole (3A and 3B) aspen stands (Mills and others 1996, 2000). Rumble and others (2001) observed this species most often in pure aspen stands. Mountain bluebirds also feed in wet and dry meadows in the Black Hills (Mills and others 1996), but their abundance in these habitats is limited by suitable nest sites in adjacent habitats (Mills and others 1996). The availability of suitable snags for cavities can limit mountain bluebird abundance (Mills and others 1996). As a secondary cavity nester, it usually nests in old woodpecker holes or natural cavities in dead trees. Artificial nest boxes are also utilized. Insects make up the majority of the mountain bluebird's diet, and high perches are an important habitat requirement for the capture of these insects (DeGraaf and others 1991).

Black-capped Chickadee—The black-capped chickadee (*Parus atricapillus*) is abundant in all structural stages of both the ponderosa pine and aspen/paper birch habitats, but is most abundant in multistoried/old growth (5) ponderosa pine and sapling-pole (3B) aspen/paper birch habitats (Mills and others 1996, 2000). The black-capped chickadee is a cavity nester and will either excavate its own nest in soft decayed wood or use an old cavity of another bird (DeGraaf and others 1991). As with most cavity nesters, abundance might be dictated by the amount and quality of snags or advanced wood decay rather than by vegetative structural stage (Mills and others 1996). In winter, black-capped chickadees require suitable cavities in snags greater than 6 inches (>15 cm) d.b.h. to maintain thermoregulation for roosting (Balda 1970; Cunningham and others 1980). Insects, seeds, and fruits are gleaned from the ground up to the tops of trees (DeGraaf and others 1991).

Northern Flicker—Northern flickers (*Colaptes auratus*) utilize both ponderosa pine and aspen/paper birch habitats in the Black Hills (Mills and others 2000; Rumble and others 2001). Northern flickers are most abundant in the shrub-seedling (2) and sapling-pole (3A-3C) aspen stands and in open mature (4A) ponderosa pine stands (Mills and others 2000). The use of the shrub-seedling aspen stands is limited to areas with suitable snags (Mills and others 1996). Northern flickers are also observed in the sapling-pole (3A and 3C), mature (4B and 4C), and multistoried/old growth (5) ponderosa pine stands (Dykstra and others 1999; Mills and others 2000). The use of a variety of structural

stages and vegetative habitat types suggests that northern flicker abundance is probably determined more by the density of snags, rather than the age or overstory coverage of a stand (Mills and others 1996). Northern flickers excavate a new cavity annually in a dead tree or near the top of a broken-off live tree. Abandoned nests are typically reused by secondary cavity nesters. Northern flickers are ground foragers that consume seeds, fruits, and insects, especially ants (DeGraaf and others 1991).

Lewis' Woodpecker—The Lewis' woodpecker (*Melanerpes lewis*) is listed as a sensitive species by the U.S. Forest Service Rocky Mountain Region (USDA Forest Service 2002c). Open mature (4A and 4B) and multistoried/old growth (5) ponderosa pine forests and burned, partially logged forests are valuable nesting habitat (Saab and Dudley 1998; Tobalske 1997; USDA Forest Service 2000). Saab and Vierling (2001) reported that conditions created by stand-replacing wildfire in a southwestern Idaho ponderosa pine forest appeared to be a potential source habitat of Lewis' woodpeckers, whereas unburned ponderosa pine forests appear to be a sink habitat. Large burned areas are colonized within a year after fire, and abundance increases for several years (USDA Forest Service 2000). Lewis' woodpeckers require large snags in an advanced stage of decay to excavate a nest (Raphael and White 1984; Tobalske 1997). In southwestern Idaho, nest trees average 18 inches (46.7 cm) in diameter (Saab and Vierling 2001). Lewis' woodpeckers will also use natural cavities and old cavities created by other woodpeckers (USDA Forest Service 2000). Unlike other woodpeckers that obtain food by drilling into trees, the Lewis' woodpecker catches insects by hawking from perches in dead trees or stumps (DeGraaf and others 1991).

Black-backed Woodpecker—The black-backed woodpecker (*Picoides arcticus*) is classified as a sensitive species by the U.S. Forest Service Rocky Mountain Region (USDA Forest Service 2002d). The black-backed woodpecker is a primary cavity nester and requires snags for nesting, foraging, and roosting. The management of this bird relies on the maintenance of forest diseases and other mortality-causing agents that lead to beetle infestations for food and heart rot for cavities (Goggans and others 1989; Marshall 1992). In the Black Hills, the black-backed woodpecker is observed in sapling-pole (3A and 3B), mature (4B and 4C) and multistoried/old growth (5) ponderosa pine stands (Dykstra and others 1999; USDA Forest Service 2000). The black-backed woodpecker forages on larvae of bark beetles and pupae of wood-boring insects (Clark and others 1989; Marshall 1992; USDA Forest Service 2000) from tree boles by scaling or

flaking bark (Bull and others 1986). Populations are irruptive and abundance is high in areas where insects proliferate such as stand-replacement burns and beetle-killed areas (Clark and others 1989; USDA Forest Service 2000). Fire suppression and postfire salvage logging can adversely affect this species (Dixon and Saab 2000). However, retaining 42 to 50 snags per acre (104 to 123 snags per ha) greater than 9 inches (greater than 23 cm) d.b.h. in clumps can minimize the impact of postfire salvaging (Wisdom and others 2000).

Downy Woodpecker—Abundance of the downy woodpecker (*Picoides pubescens*) throughout the year is similar across structural stages and canopy coverage of both ponderosa pine and aspen stands (Mills and others 2000; Rumble and others 1999, 2001). This cavity nester, which rarely reuses an old cavity or cavities of other birds, will excavate a new cavity each year near the top of a dead tree, or in a live tree if heartrot is present. The abundance and distribution of suitable snags is probably most important in determining habitat suitability. The majority of its diet is wood-boring larvae, which are obtained by excavating and drilling into the bark of trees (DeGraaf and others 1991).

Three-toed Woodpecker—The three-toed woodpecker (*Picoides tridactylus*) is classified as a sensitive species by the U.S. Forest Service Rocky Mountain Region (USDA Forest Service 2002e). The three-toed woodpecker chooses mature (4) and multistoried/old growth (5) ponderosa pine and white spruce stands (USDA Forest Service 2000). Recently burned areas or areas infested with bark beetles, disease, or heart rot provide foraging and nesting habitat (Goggans and others 1989; USDA Forest Service 2002e). This primary cavity nester excavates nests each year in large diameter (9 to 20 inches; 23 to 51 cm) dead or live trees that have heartrot (Clark and others 1989; Goggans and others 1989; Scott 1980). Once the three-toed woodpecker abandons the nest, it is often used by a variety of secondary cavity-nesting species (Finch 1991). Three-toed woodpeckers forage on wood-boring beetle larvae, and are attracted to areas with high concentration of beetles (USDA Forest Service 2000; Villard 1994).

Hairy Woodpecker—Hairy woodpeckers (*Picoides villosus*) utilize both ponderosa pine and aspen/paper birch habitats in the Black Hills throughout the year (Mills and others 1996, 2000; Rumble and others 1999, 2001). Hairy woodpecker abundance is generally highest in the open to moderate canopy coverage stands of sapling-pole (3A and 3B) and mature (4A and 4B) stands of ponderosa pine. In addition, abundance is high in multistoried/old growth (5) ponderosa pine and

in the moderate to high canopy coverage of sapling-pole (3B and 3C) stands of aspen/paper birch (Dykstra and others 1999; Mills and others 1996, 2000). The similarities of hairy woodpecker abundance among vegetation types and canopy coverages is most likely associated with the availability of suitable snags rather than the structural stage of the stand (Mills 1994; Mills and other 1996). For instance, the occurrence of hairy woodpeckers in the shrub-seedling (2) stage of aspen was related to the presence of snags (Mills and others 1996). Hairy woodpeckers excavate nests in live or dead ponderosa pine and aspen trees greater than 10 inches (>25 cm) d.b.h.. Live trees typically have heartrot present (DeGraaf and others 1991). An important habitat requirement for this species is large-diameter woody material for forage (Rumble and others 1999). Its main food source, wood-boring beetles, is obtained by excavating and drilling holes into the bark of trees. It also eats nuts, fruits, and other insects (DeGraaf and others 1991).

Red-naped Sapsucker—This medium-sized woodpecker (*Sphyrapicus nuchalis*) is considered an aspen/paper birch obligate in the Black Hills, although it does utilize ponderosa pine forests (Dykstra and others 1999; Mills and others 2000; Rumble and others 2001). The red-naped sapsucker creates nests in live aspen trees, especially ones that are susceptible to heart rot (Crockett and Hadow 1975). Red-naped sapsucker abundance is related to suitable snag availability. Mills and others (2000) reported that structural stages with at least four aspen/paper birch snags greater than 6 inches (>15 cm) in diameter per acre (11 snags per ha) had a high red-naped sapsucker abundance. Furthermore, Daily (1993) suggests that the degree of heart rot is more important in determining habitat capability than the vegetative structural stage. In the Black Hills, red-naped sapsucker abundance is highest in sapling-pole (3B and 3C) aspen stands, although some are found in the other structural stages of aspen (Mills and others 1996, 2000). Red-naped sapsuckers are also found in mature (4C) and multistoried/old growth (5) ponderosa pine stands (Dykstra and others 1999; Mills and others 1996). Red-naped sapsuckers drill rows of holes through the outer bark of trees and eat the tree sap and cambium. They also eat the insects that are attracted to these holes (DeGraaf and others 1991).

Red-breasted Nuthatch—The red-breasted nuthatch (*Sitta canadensis*) is found at higher elevations during summer and in lower elevation ponderosa pine and deciduous forests during winter (Haldeman 1980; South Dakota Ornithologists' Union 1991).

Throughout the year, red-breasted nuthatches are most abundant in mature (4B and 4C) and multistoried/old-growth (5) ponderosa pine stands (Crompton 1994; Dykstra and others 1999; Mills 1994; Mills and others 2000; Rumble and others 1999). Red-breasted nuthatches are also found in sapling pole (3) ponderosa pine stands with the highest abundance in areas with high canopy coverage (Mills 1994; Mills and others 2000). Sapling-pole aspen/paper birch habitats also provide some habitat, especially in areas with high canopy coverage and where ponderosa pine is present (Mills and others 1996; Rumble and others 2001). The red-breasted nuthatch nests in natural cavities, old woodpecker cavities, or excavates its own in soft snags (DeGraaf and others 1991). The high abundance of red-breasted nuthatches in mature and multistoried/old-growth ponderosa pine forests is most likely related to its diet and need for large-diameter snags for nests (Mills and others 2000). Ponderosa pine seed makes up the majority of the red-breasted nuthatch's diet, and seed production in ponderosa pine stands is highest in areas with increased basal area and canopy coverage (Rumble and Anderson 1996b).

White-breasted Nuthatch — Both coniferous and mixed coniferous/deciduous forests can provide habitat for the white-breasted nuthatch (*Sitta carolinensis*) (Mills and others 1996, 2000; Rumble and others 2001). Abundance during the summer is greatest in sapling-pole (3B) and mature (4B) ponderosa pine stands (Dykstra and others 1999; Mills 1994; Mills and others 1996, 2000). In winter, multistoried/old growth (5) and ponderosa pine stands with more than 70 percent canopy coverage provide moderate quality habitat. Open bur oak habitats provide the best habitat for white-breasted nuthatches during the winter (Rumble and others 1999). White-breasted nuthatches are secondary cavity nesters, and use natural cavities or old woodpecker cavities (DeGraaf and others 1991). Food is obtained by gleaning insects from the bark of trees and seeds from the ground. In the fall and winter, acorn mast is an important food source (DeGraaf and others 1991).

Pygmy Nuthatch—The pygmy nuthatch (*Sitta pygmaea*), found in the lower elevations of the southern and eastern sections of the Black Hills, has been classified by the Rocky Mountain Region of the Forest Service as a sensitive species (USDA Forest Service 1996c). The pygmy nuthatch is associated with mature (4A and 4B) and old growth (5) ponderosa pine forests with canopy coverage less than 70 percent (Clark and others 1989). The pygmy nuthatch is a primary cavity nester and requires soft snags and old trees over 19 inches (>48 cm) d.b.h. for nesting (Clark and

others 1989; DeGraaf and others 1991). Since pygmy nuthatches are communal nesters, clumps of snags would benefit the population. In Arizona, Scott (1979) reported pygmy nuthatch populations declined in thinned stands where snags were removed, but were stable in adjacent thinned stands where snags were conserved. Clark and others (1989) suggest three to five snags per acre (8 to 12 snags per ha) that are at least 19 inches (48 cm) d.b.h. for roost cavities. Insects are its primary food source, but it also consumes conifer seeds (DeGraaf and others 1991). Food is obtained by gleaning insects from the bark of trees or hawking insects from the air.

Others

Brown-headed Cowbird—The brown-head cowbird (*Molothrus ater*) does not build its own nest; instead, it lays its eggs in the nests of other songbirds. Some authors attribute recent declines in songbird populations to parasitism of nests by the brown-headed cowbird (Whitcomb and others 1981). In the Black Hills, the brown-headed cowbird is less abundant than most permanent resident species (Mills 1994). Abundance is similar among all structural stages of ponderosa pine and aspen, as well as all the canopy coverage categories (Dykstra and others 1999; Mills and others 2000; Rumble and others 2001). Brown-headed cowbirds eat insects and seeds gleaned from the ground (DeGraaf and others 1991).

Management guidelines

Managers concerned with sustaining and increasing Black Hills bird diversity should start by maintaining within- and between-stand structural diversity to provide for habitat needs of birds during both the breeding season and the winter (DellaSalla and others 1996; Huff and others 1991; Mills and others 2000). Ponderosa pine forests that have deciduous components such as bur oak or aspen can provide more habitats for some wildlife species than are typically found in a monotypic coniferous stand (Dykstra 1996; Glenn-Lewin 1977; Mills and others 2000; Rumble and others 2001).

Cavity nesters comprise 30 percent of the breeding bird richness in the central Black Hills (Mills and others 2000). Some cavity nesters that require large snags, such as the pygmy nuthatch and brown creeper, might be negatively impacted if snag guidelines do not provide for them (Dykstra 1996). Distribution, abundance, and condition of snags as well as deciduous trees beneath the ponderosa pine canopy should be included in forest inventories to better quantify habitats for these birds (Mills and others 2000). Snag data should be stratified by vegetation structural stage to ensure

adequate snag density and size across all classes (Mills and others 2000). The largest snags should be retained during timber harvests because birds prefer them, the snags survive longer, and better winter thermal cover is provided (Cunningham and others 1980). Snags less than 6 inches (<15 cm) d.b.h. should also be retained because they are important forage sites for cavity-nesting species (Baker 1973). Firewood gathering should be monitored carefully in areas of low snag density because this activity can negatively affect cavity nester populations.

The importance of postfire forests in conifer-dominated systems of the West should not be ignored. Fire affects bird nesting and foraging habitat by creating snags, altering insect communities, reducing foliage, and altering forest structure (Brown and Smith 2000; Smith 2000). Kotliar and others [in press] summarized the influence of fire and salvage logging on avian communities in conifer-dominated forests of the West. In general, wood drillers (black-backed woodpeckers and three-toed woodpeckers) and aerial insectivores (mountain bluebirds) were more abundant in early postfire forests, while bird species that occur most often in unburned forests were usually foliage and bark gleaners.

Postfire salvage logging can be detrimental to cavity nesters that use postfire habitats. Snag reduction may seriously impact black-backed and three-toed woodpeckers, which rarely use partially logged postfire forests (Kotliar and others, in press). This effect can be ameliorated if proper measures are taken to retain a diversity of snag sizes in various stages of decay. Furthermore, varying the spatial distribution of snags within salvaged forest can provide nesting and foraging habitats for both bird species that prefer high densities of snags and bird species that don't.

Tree density and coverage influence bird species composition. Harvests of mature and old growth ponderosa pine stands would negatively affect red-breasted nuthatches, western tanagers, gray jays, brown creepers, and Swainson's thrush, which require high amounts of overstory canopy coverage (Mills and others 2000). However, species that require open overstory canopy cover, such as northern flickers, dark-eyed juncos, mountain bluebirds, and chipping sparrows would benefit from such harvests (Mills 1994; Mills and others 1996, 2000). Forests are dynamic. Any natural disturbance, management activity, or even ecologic succession will have both positive and negative effects on the birds living in them. Maintaining as much diversity as possible at the landscape scale is likely the best means of balancing the winners and losers and maintaining optimal bird populations.

Silviculture of Ponderosa Pine in the Black Hills

Silvics of Black Hills Ponderosa Pine

Several unique characteristics of the Black Hills make the area well suited for management with different silvicultural techniques. The intermediate shade tolerance of ponderosa pine allows regeneration under partial shade as well as full sunlight. Pine seedlings can establish under a variety of overstory densities, ranging from 0 to 60 ft² per acre (0 to 14 m² per hectare) and in some cases above that density where gaps occur in the forest canopy (fig. 33). Ponderosa pine has large, heavy seeds with relatively small wings that disperse fairly close to parent trees (1 to 1.5 tree height). Ponderosa pine seed is produced almost every year, with abundant crops every 2 to 5 years (Boldt and Van Deusen 1974), although seed production on the Limestone Plateau and some portions of the Bearlodge Mountains has been sporadic in recent years. In areas with prolific seed production and favorable climate, natural regeneration of ponderosa pine can be quite successful.

Frequent rain showers throughout the growing season, which lasts from early March to August, is the major climatic factor contributing to the prolific growth and establishment of ponderosa pine. Greenhouse studies (Noble and others 1979) have shown that seed germination and growth are optimal at temperatures above 50 °F (15 °C) and where rainfall exceeds 1 inch (2.5 cm) a month. These conditions are usually met throughout the growing season in the Black Hills (fig. 34). In contrast, average precipitation from March to July in the Colorado Front Range and Southwest is insufficient to establish ponderosa pine seedlings (fig. 35). Furthermore, temperatures in the Front Range and Southwest do not usually reach the 50 °F (15 °C) germination threshold at the same time when rains do occur. Consequently, ponderosa pine regenerates only sporadically in these areas (Shepperd and others, submitted).

Regeneration Cuts: Even-aged

Two-cut shelterwood

The two-cut shelterwood, even-aged method is very well suited for Black Hills ponderosa pine. This natural regeneration method has been the most extensively used within the Black Hills, and proven to be most reliable in managing these forests.



Figure 33: An even-aged ponderosa pine stand in the Black Hills Experimental Forest with an average basal area of 64 ft² per acre. Note the scattered ponderosa pine regeneration in the understory.

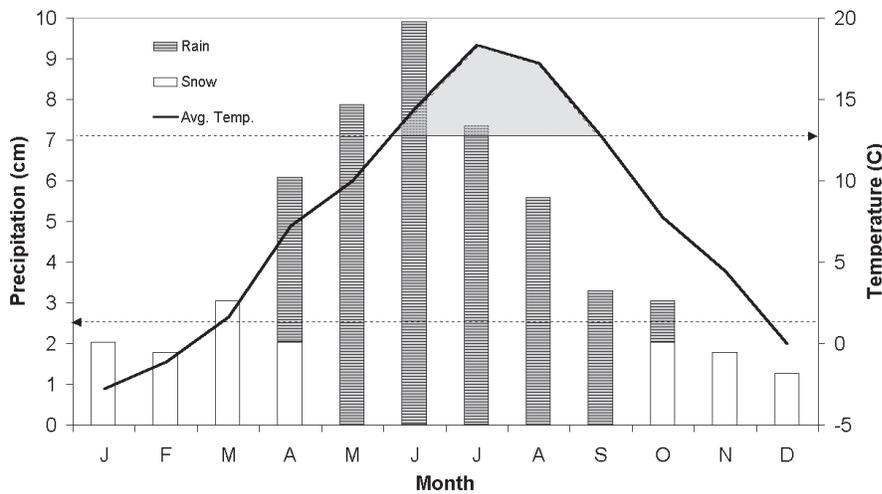


Figure 34: The average precipitation and average temperature in the Black Hills over an entire year (Shepperd and others 1983). Dotted arrows indicate the minimum precipitation and temperature required for ponderosa pine seed germination (Noble and others 1979). Shaded gray area indicates period in the Black Hills where conditions are optimal for seed germination.

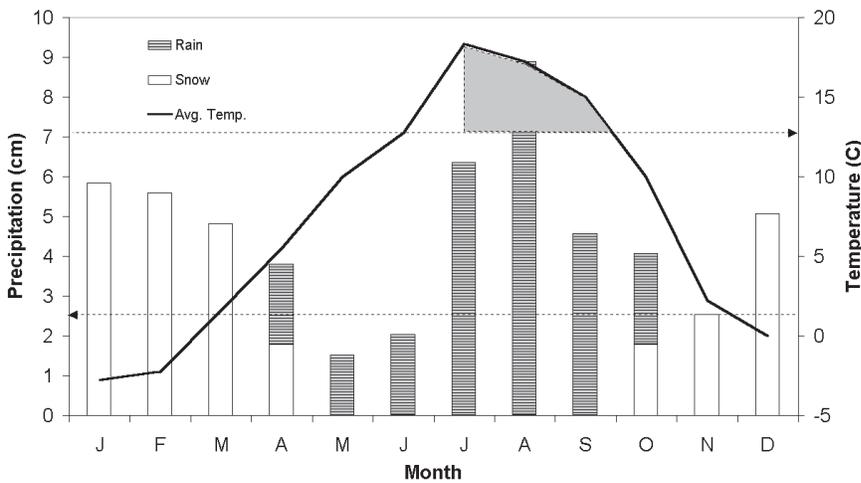


Figure 35: The average precipitation and average temperature in the Rocky Mountains and Southwest United States over an entire year (Shepperd and others 1983). Dotted arrows indicate the minimum precipitation and temperature required for ponderosa pine seed germination. Shaded gray area indicates period in these regions where conditions are optimal for seed germination.



Figure 36: A ponderosa pine shelterwood stand with dense regeneration ready for the removal of the overstory in the Black Hills National Forest.

In a two-cut shelterwood, basal areas are generally reduced below 60 ft² per acre (14 m² per hectare) in an initial seed cut using a marking regime that leaves a uniform canopy of the biggest and healthiest trees for a seed source. Abundant natural regeneration is usually produced within 5 to 10 years. Under a classic shelterwood, all remaining overstory trees are removed once a new generation of trees is established. The overstory can be harvested anytime after sufficient seedlings are present to establish a new forest. Ideally, the overstory should be removed while the new trees are small to minimize damage to them during logging. In practice, however, the overstory has either not been completely removed or removal has been delayed until new seedlings have grown to large sapling or pole-size trees. In addition, some residual overstory trees are also left in Black Hills shelterwood systems for wildlife habitat, snag recruitment, and other purposes.

A classic two-cut shelterwood allows sufficient sunlight to reach the forest floor to establish seedlings, yet provides enough shade to ameliorate the harsh microclimates of full canopy openings. The residual overstory trees provide an even distribution of seed from healthy, desirable genotypes. Scarification that occurs with logging is usually adequate to provide the mineral soil seedbed favored by ponderosa pine. Except for extremely harsh sites, these factors combine to establish abundant understories of several thousand seedlings per acre (fig. 36) (Boldt and Van Deusen 1974).

Three-cut shelterwood

Three-cut shelterwood is primarily recommended for situations where overstory trees are susceptible to windthrow, or where additional growth of a portion of the overstory canopy is desired before the forest is regenerated. Removing too many trees at once in a high wind risk situation can result in the loss of seed trees before the new regeneration becomes established. To lessen this risk, an initial preparation cut is first made to open the overstory somewhat and allow the residual trees to develop windfirmness before the seed cut is made. Usually no more than one-third of the existing stand basal area is removed under the initial entry of a three-cut shelterwood. Residual basal areas are kept above 60 ft² per acre (14 m² per hectare) to minimize regeneration after the initial entry. Following the development of natural windfirmness or the loss of trees that were susceptible to windthrow (usually 10 to 15 years), a seed cut is then applied to regenerate the forest as in a two-cut shelterwood system. Areas that have exhibited unusual amounts of past windthrow, evidenced by many downed trees with large root masses tipped up, windy ridge tops, saddles in ridges, and other places susceptible to wind are candidates for the three-cut shelterwood system.

Using the shelterwood system for managing Black Hills ponderosa pine provides a reliable and quick means to reestablish an even-aged forest. There has been a long, successful history of shelterwood use in the Black Hills. Shelterwood management provides the

seedbed treatment, growth environment, and seed source necessary to successfully reestablish ponderosa pine. In addition, understory forage for wildlife and livestock is increased in many habitat associations under a shelterwood system.

Extensive use of both two- and three-cut shelterwood systems can create uniform forests over large areas of a landscape, which may not meet diversity requirements for wildlife habitats and other resource values. Classic shelterwood systems also have the appearance and environment of regenerated clearcuts following the overstory removal. Careful layout and scheduling of cuts is therefore necessary to avoid conflicts with other resource values.

Three-step shelterwoods have the disadvantage of the extra expense of the third entry and the extra time needed to regenerate the forest. The additional entry may also contribute to soil compaction and erosion, unless properly executed. However, these factors are potentially offset by increased growth of the trees remaining after the first cut.

Seed-tree method

The seed-tree method has also been used successfully in the Black Hills. The seed-tree method is

similar to the two-step shelterwood, except only a few trees per acre are left to provide seed for regeneration (fig. 37). One feature of the seed-tree method that can be either a potential advantage or disadvantage is that heavy ponderosa pine seed does not get uniformly distributed throughout the stand. Regeneration may then be spotty and clumpy, especially if the seed trees themselves are not uniformly spaced. However, such a structure may actually be desirable under current holistic management philosophy. If so, the seed-tree method can provide an open, irregular stand structure. The seed-tree method can be modified to both regenerate the forest and provide reserve trees for wildlife and snag recruitment in a single entry by simply never removing the seed trees.

The low density of residual trees in a seed-tree cut allows abundant understory forage production for livestock and wildlife. Seed-trees are more susceptible to windthrow, however, because they are more isolated and widely separated than trees in the shelterwood system. The seed-tree method should therefore not be used in high wind risk areas. Seed-trees also look artificial, especially when regularly spaced and applied to large areas of the landscape. Nor does the seed-tree method



Figure 37: A ponderosa pine seed-tree cut in the Black Hills.

provide suitable habitat for closed-forest and canopy-dependent birds and animals.

Clearcut method

From a natural regeneration standpoint, clearcutting is the least desirable of any of the even-aged systems that could be practiced in the Black Hills. The primary reason is that effective seed fall distance is so short that only very small clearcuts could be successfully regenerated. The clearcut method is generally recommended only for ponderosa pine stands that are heavily infested with dwarf mistletoe, which does not occur in the Black Hills.

Clearcutting does provide openings for some wildlife species (fig. 38), and can be used to open roadside vistas within pine forests for a period of time until the regeneration is established. Clearcutting can certainly be used in the Black Hills in a patch clearcut mode where the cuts are small enough to allow seeds from surrounding stands to reach the opening. The size of the opening should normally be restricted to about twice the 250-foot (76 m) dispersal distance of ponderosa pine seed, if a seed-dispersing stand exists all around the perimeter (Boldt and others 1983). Strip clearcuts

that are oriented perpendicular to the direction of prevailing winds can also be quite successful, as experience on the Black Hills Experimental Forest has demonstrated (fig. 39).

Clearcuts can look artificial from a distance or from the air unless the cuts are properly designed and shaped to blend into the natural character of the landscape. Geometric shapes should be avoided. Openings should be irregularly shaped, and be placed in the landscape to mimic natural disturbance patterns. Production of grasses and forbs for domestic livestock and wildlife is maximized in clearcuts, at least for a few years until the regeneration is established and reaches sapling size. Clearcuts benefit bird and mammals that prefer openings, such as deer and elk, provided the clearcuts are not too large (Wallmo and Schoen 1981).

Regeneration Cuttings: Uneven-Aged Silviculture

Uneven-aged regeneration methods can meet the requirements for a variety of conditions needed under today's management philosophy. Uneven-aged management allows for the creation of forests of various



Figure 38: Structural diversity within the forested landscape created by applying a patch clearcut in the middle of a dense forest.



Figure 39: A ponderosa pine strip clearcut in the Black Hills Experimental Forest.

structural and spatial distributions, and allows managers to create conditions that can meet a variety of resource needs, such as wildlife habitat, scenic values, forage, recreation areas, timber, roadside travel corridors, and so forth. Uneven-aged management mimics the later stages of succession, primarily the understory reinitiation stage (Guldin 1996; Oliver and Larson 1996). Basically, the older age cohorts in the overstory die at the same rate as new age cohorts are established. This creates a forest of diverse canopy structure that may be uniformly stocked, or contain numerous gaps and clumps of trees.

While even-aged silviculture is designed to raise a crop of trees to maturity and then harvest and replace them with a new forest, uneven-aged silviculture is designed to create a desired condition in the forest landscape and then maintain that condition through time by periodic entries to remove trees of various sizes and in various patterns. The goal is to eventually attain a forest comprised of trees of all ages and sizes. While this diverse structure is probably not achievable initially in any major area in the Black Hills, existing conditions are such that many areas can be moved toward an uneven-aged structure with one or two entries, or cutting cycles.

Stocking control

Control of stocking or density under uneven-aged management is achieved through one of two methods: either the BDQ method, which relies on a basal area, diameter distribution, and a “Q” ratio, or the SDI³ (Stand Density Index) method (Long 1995),

which distributes stocking of a forest in a variety of SDI classes containing trees of various sizes. Both of these methods require a stocking guideline that is usually based on basal area as well as upper and lower diameter limits to achieve the desired stocking goal.

Under the BDQ system, the manager first selects the “Ds,” or upper and lower target diameters that define the range of tree sizes to be managed. Normally, the upper diameter limit is dictated by the maximum tree size the manager wants to retain on the site (whether it is actually present or not). The lower diameter limit is usually the smallest diameter class tallied in the stand inventory.

The “Q” curve is used to control the distribution of tree diameters between upper and lower target diameter classes. By definition, the Q factor is the ratio between the number of trees per acre in one diameter class and those in the next smaller class (Alexander and Edminster 1987). At values larger than 1.0 its computation always results in a distribution of tree sizes that follows a negative exponential decay function. In other words, each diameter class in the target forest

³ Stand density index (SDI) is a relative measure of stand density that provides a relationship between stand basal area, trees per unit area, average stand diameter, and stocking of a forested stand (Reineke 1933). The premise of SDI is that if the carrying capacity for tree growing space is fully occupied, an increase in average stand diameter can only occur if there is a reduction in the number of trees per unit area.

will always contain greater numbers of trees than the next larger class. The intent of such a distribution is to ensure that there will always be sufficient numbers of smaller trees to grow into each successive larger diameter class, even with normal rates of mortality. Increasing the Q factor will increase the density of smaller trees in the desired forest. Decreasing the Q factor will lower the density of smaller trees (fig. 40). Therefore, changing the Q factor can drastically alter the

appearance of the target forest, regardless of the desired residual basal area.

The final factor that controls stocking under the BDQ method is “B,” or the residual basal area that the growing stock is reduced to following each cutting cycle. Raising or lowering the residual basal area changes the number of trees desired in each diameter class (and thus the overall stocking), but the Q ratio is unaffected (fig. 41). Specific methods for calculating

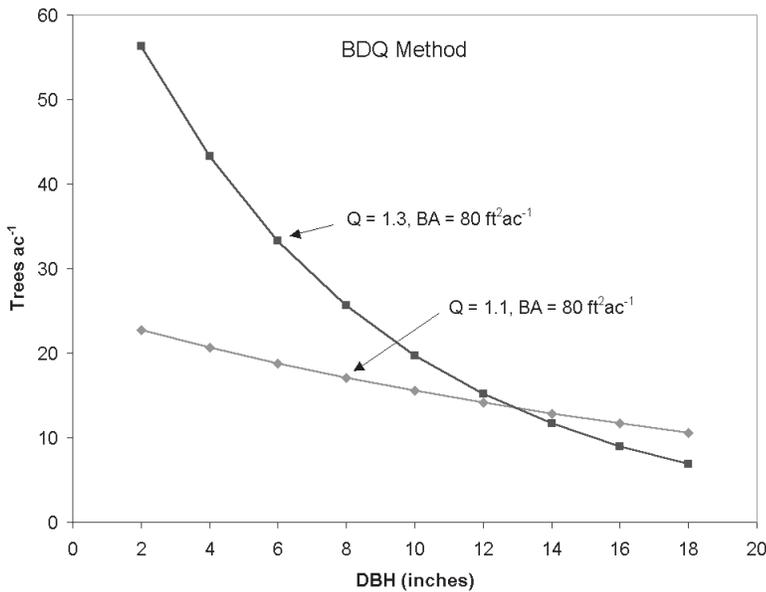


Figure 40: Influence the “Q” factor has on trees per acre in each diameter class for a stand with similar basal area.

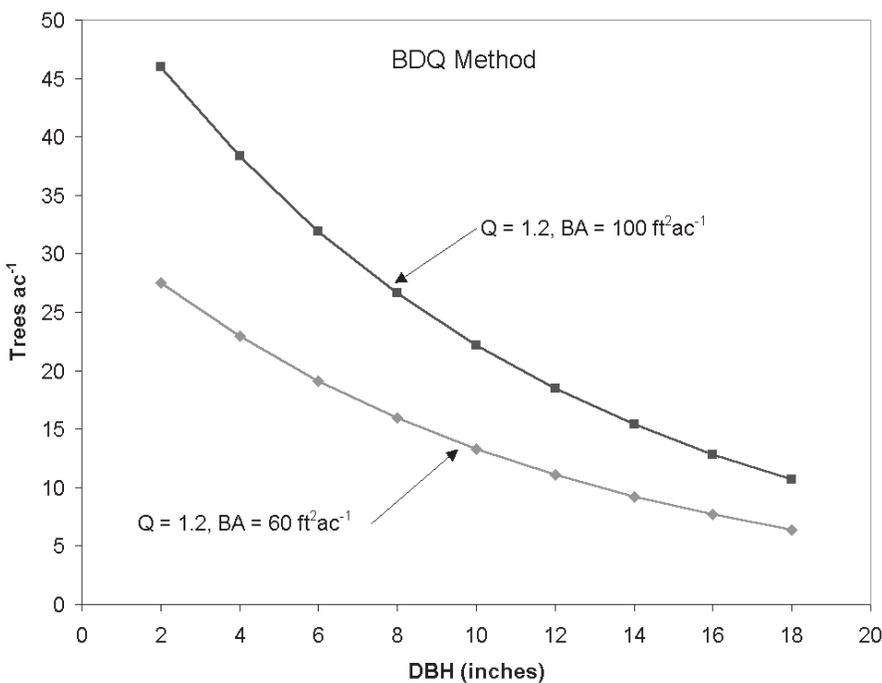


Figure 41: Influence that residual basal area has on trees per acre in each diameter class under identical “Q” factors.

desired stocking under the BDQ method are presented in Guldin (1996) and in Alexander and Edminster (1987). Desired stocking can easily be calculated with a computer spreadsheet and rudimentary cruise data.

The SDI method of calculating desired stocking in an uneven-aged forest also utilizes upper and lower diameter limits, but apportions stocking in each diameter class according to Stand Density Index, rather than using a Q factor and residual basal area (Long 1995). A total residual SDI is first selected (usually about one-third of maximum SDI for the species). This SDI is then apportioned evenly across all diameter classes in the target stand, and used to calculate the number of desired residual stems per acre in each class. The result is a distribution of stems across all diameter classes not unlike that achieved using the BDQ method, but without a constant ratio between the number of trees in each class. Changing the residual target SDI for a stand will raise or lower the overall stem density, but will not change the shape of the distribution of trees among classes (fig. 42). The target SDI can be apportioned differentially rather than evenly among diameter classes to vary the distribution, and avoid dense understories (Long 1995). However, apportionment must not result in greater numbers of trees in larger diameter classes than in smaller classes. Regardless of how the total SDI is apportioned, overall stocking for the target stand will remain within the desired SDI stocking limits.

Another method of allocating growing stock in complex multiaged or multistructured stands is the Multiaged Stocking Assessment Model MASAM (O'Hara and Valappil 1999). This approach relies on fundamental relationships between leaf area and growing space to predict the growth response under a specified allocation of growing stock, and could potentially be used with many existing Black Hills ponderosa pine stands.

There are several complicating factors in using the MASAM as presented in O'Hara and Valappil (1999). Local allometric relationships must be developed to estimate leaf area index (LAI) of designated structural cohorts of the forest canopy. These allometric relationships have not yet been developed for Black Hills ponderosa pine. Evaluations of alternate prescriptions require repeated runs of the MASAM and cannot be done in a spreadsheet as with the BDQ and SDI methods. MASAM does not produce yield or stand tables that can be utilized to develop timber sale plans or marking guides. If these limitations can be overcome, MASAM could potentially be a useful tool in managing complex ponderosa pine stands in the Black Hills.

By manipulating either the target stocking or the distribution of trees in the various size classes, both the BDQ and the SDI methods can create forests of infinitely different structures. Managers must rely on their knowledge of silvics and common sense to choose a desired condition that is both attainable and sustainable.

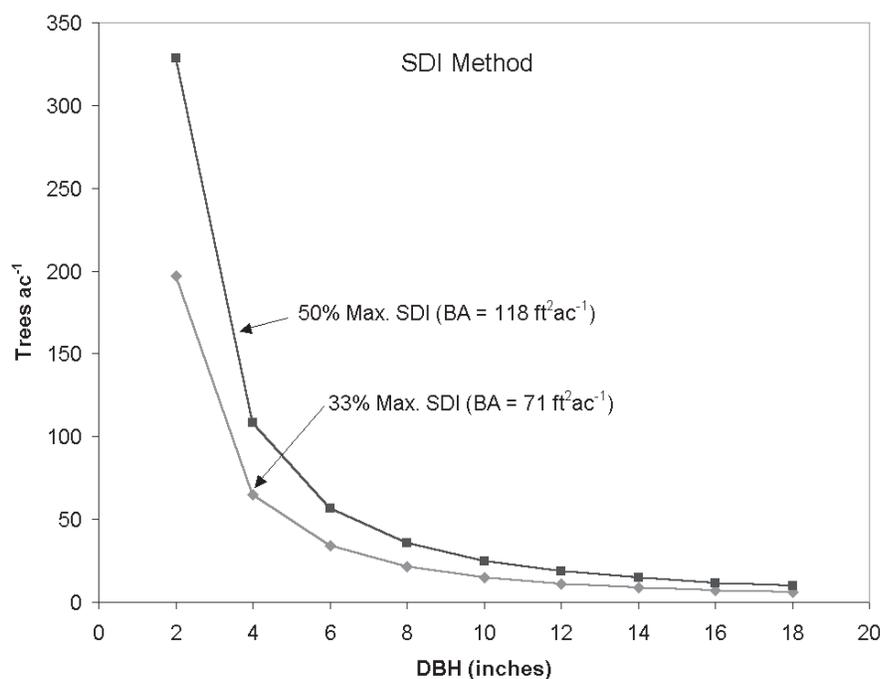


Figure 42: Distributing stocking in an uneven-aged stand using the SDI method. A change in the residual target SDI will change overall stem density, but the proportional distribution of trees among diameter classes will remain the same.

For example, a residual stocking target greater than 80 to 100 ft² (18.3 to 23 m² per hectare) of basal area per acre in Black Hills ponderosa pine forests is probably not sustainable. Such forests would grow slowly and be highly susceptible to mountain pine beetle attack. Conversely, selecting a high Q factor (or differentially apportioning SDI into smaller diameter classes) would also result in unattainable stocking (not enough light through the overstory canopy to establish pine regeneration and grow new trees into larger diameter classes). Such dense understories would also serve as live fuel ladders, making those forests extremely susceptible to crown fire.

In practice, the goal of most uneven-aged prescriptions in Black Hills ponderosa pine is to create a relatively open diverse forest that has a lot of structural and spatial diversity, and is not susceptible to stand-replacement fires or catastrophic insect attack. This can usually be accomplished using residual basal areas under 60 ft² per acre (13.8 m² per hectare) with moderate stocking in the smaller diameter classes, while retaining some large trees. Cutting cycles of 20 to 30 years are usually appropriate, although initial applications of uneven-aged management in the Black Hills indicate that a supplemental pre-commercial thinning may be needed between entries to control excessive regeneration before it gets too large (fig. 43).

Another factor controlling the selection of a residual target condition for uneven-aged management is the current stocking of the forest. While it is possible to eventually convert a strictly even-aged forest to an uneven-aged condition after several cutting cycles,

uneven-aged management is best suited to forests that already contain several age classes and have some structural diversity. In these cases, the desired target condition should approximate the current diameter distribution as much as possible, removing surpluses from overstocked classes and leaving understocked classes to grow. Diameter class distributions of existing forests almost never agree with the Q curve, even in stands that have previously been placed under uneven-aged management. Unforeseen mortality, insect and disease damage, and differences in microsite conditions and genetics can all be responsible. Therefore, surpluses and deficits among diameter classes should be balanced so that the target stocking (basal area or SDI) is maintained. Strictly cutting surplus diameters back to the target stocking will result in an overharvest, or understocked condition whenever deficit diameter classes exist.

Single-tree versus group selection

Under single-tree selection, trees are managed as individuals within a population where trees are selected for harvest or retention on a tree by tree basis in each entry, or cutting cycle. The ideal result from single-tree selection is a uniformly stocked forest containing trees of a variety of sizes.

Under group selection, the mix of trees of various ages and sizes in the forest is accomplished by manipulating the trees in groups such that openings, groups, and clumps of similar sizes are distributed throughout the landscape. Group selection is not simply creating small holes in the forest, or harvesting small patch clearcuts from the forest in periodic

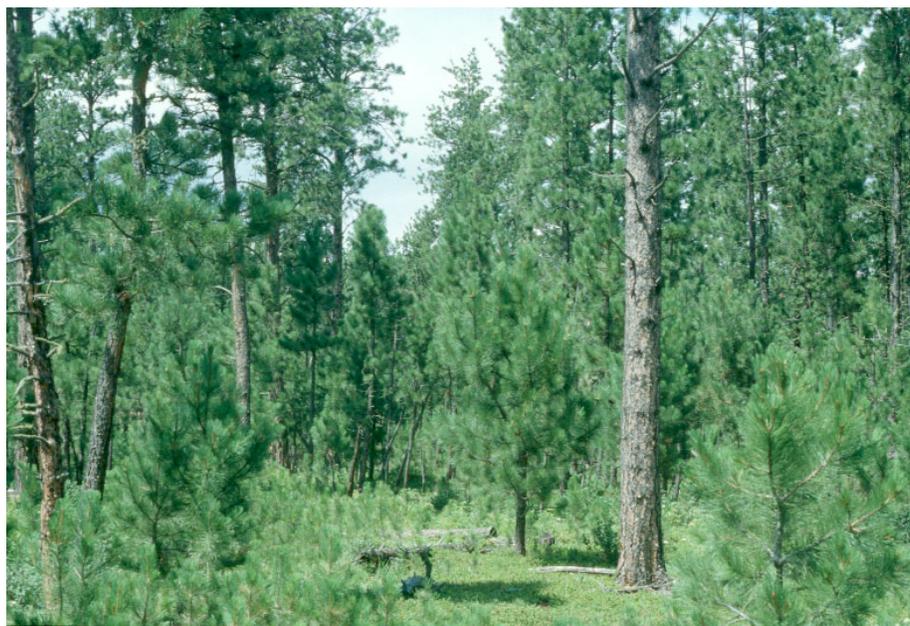


Figure 43: Uneven-aged ponderosa pine stand in the Black Hills Experimental Forest.

entries. The intervening matrix of forest between harvested groups and clumps of trees is also managed to insure that overall stocking of the forest is maintained in the desired distribution of age and size classes.

Regardless of the method used to control stocking, both group and individual-tree selection systems should be implemented so that a sustainable (and desired) condition are maintained through time. Either the BDQ or SDI method can be used to regulate overall stocking under both single-tree and group selection. This enables a very diverse forested landscape to be created and maintained using a single set of inventory records. Uneven-aged silviculture can provide an infinite variety of forest structures and conditions for wildlife habitat guidelines, fiber production, aesthetics, developed recreation areas homesites, and roadside corridors, where maintenance of the forest environment through time is paramount.

Another advantage of uneven-aged management is that the forest structure itself provides a record of past disturbance and management activity, so it is easy to interpret what condition the forest is in, and what additional entries or manipulations are needed to return it to the desired condition. Unless clear records have been retained, it is often impossible to tell what the intentions of previous management entries were under even-aged systems. This is especially true where residual overstory trees were retained for other resource uses.

Layout, marking, and utilization under uneven-aged systems

Uneven-aged management requires skill on the part of tree markers, timber inventory personnel, and sale-layout and sale-administration personnel to achieve the desired objectives. Layout of an uneven-aged prescription begins with a well-designed inventory of the forest. Calculating the desired BDQ or SDI stocking distributions requires reliable data by diameter classes in all areas of the stand. Grouped or clumped stands may require greater numbers of inventory plots to obtain statistically accurate data. The total number of trees to be removed in the stand can be calculated once target stockings have been selected for each diameter class. These totals are then used to mark the stand. Computing a ratio or percentage of trees to mark in each diameter class can help guide marking personnel in selecting trees and avoid undermarking or overmarking a particular diameter class.

The above system can be used in marking both group and individual tree selections by simply tallying all trees removed in a group (or subtracting the equivalent per area stocking by diameter class if groups are designated by a painted boundary). Individual trees can also

be removed from the area between harvested openings to maintain the overall desired stocking goal. Treating an uneven-aged forest in this manner ensures that stocking is maintained throughout diverse sites, and allows the entire area to be tracked as a single site record in inventory databases. Although stocking may vary considerably within a group selection area, and result in differential growth and susceptibility to insect and disease attack in some portions of the stand, such discrepancies should average out over the site and not deviate significantly from the growth that would be expected in a uniformly stocked uneven-aged stand.

Any uneven-aged management harvest entry will produce a variety of product sizes, which will require a complete spectrum of wood product utilization to be economically viable. Typically, an uneven-aged management entry will produce material throughout the range of diameters within the site. Where natural regeneration is prolific, as in Black Hills ponderosa pine forests, many small stems will have to be removed in each entry. Markets or ways of economically disposing of this material will need to be developed. It therefore may be desirable to discourage regeneration by carrying slightly higher residual stocking, or avoiding excessive scarification during logging. An additional pre-commercial thinning entry or prescribed burning between cutting cycles may also be needed to remove excessive saplings.

Uneven-aged management also requires careful logging to avoid damage to residual stems, many of which will be maintained on site for several cutting cycles. Harvest equipment and techniques need to be configured to handle a variety of log sizes and product classes. Removing material in two stages may be more efficient to accommodate these needs. Designating permanent skid trails may be a practical way to avoid compacting soil under the 20- to 30-year cutting cycles in Black Hills uneven-aged ponderosa pine forests.

Hybrid Systems: Two-Aged/Three Aged

Today's resource demands are such that maintaining stands in two or more even-aged classes is desired. Black Hills ponderosa pine forests lend themselves well to management under two- or three-aged systems. These hybrid forests are not strictly even-aged or uneven-aged, but have a canopy that would be two to three storied in nature, comprised of trees of two or three distinct age classes (fig. 44). Each of these age classes would be managed as an even-aged cohort within these forests. Older age classes would be harvested at some point in time when they mature, and a new replacement age class would be established under



Figure 44: Two-aged ponderosa pine stand in the Black Hills.

the remaining canopy. These systems have the advantage of keeping a forested environment on a site at all times, and are somewhat less complicated than applying a pure uneven-aged system. A more regimented forest structure is maintained than in an uneven-aged system. While these systems theoretically could be applied in a group or clumped mode with more than three age classes, a point is reached in which they become indistinguishable from an uneven-aged system and probably should be treated as such.

Intermediate Operations

One thing unique to the Black Hills ponderosa pine is the prolific establishment of regeneration under both even- and uneven-aged management. It is certainly possible to achieve more regeneration in the understory than is desired under any of the systems discussed above. The need for intermediate operations to control density of Black Hills ponderosa pine is easily illustrated by simulating the growth of a typical forest under a number of even- and uneven-aged scenarios (fig. 45). Without periodic silvicultural intervention, most stands will quickly grow into stocking levels that are either unhealthy, or are undesirable for many resources.

The Black Hills National Forest has an increasing precommercial thinning backlog that is evident throughout many of the forests that have been managed or entered in the last 20 or 30 years. Many of these forests contain dense, very thick understories containing several thousand stems per acre (fig. 46).

As these stems grow, they are becoming more susceptible to catastrophic wildfire, disease, and insect attacks. Managers should plan for periodic density control to alleviate some of the current overstocking problems and avoid future problems.

Precommercial thinning

Precommercial thinning is an absolute necessity in the Black Hills under all even- and uneven-aged silviculture regeneration methods to maintain desired density attributes for managed stands. Thinning should be performed as early as possible from both a biologic and economic standpoint. As dense young trees increase in size, the cost of cutting and removing them increases as well as the biomass of fuel loadings created. Early thinning is also the most cost-effective way to maximize productivity over a rotation in even-aged management, or between cutting cycles in uneven-aged stands.

Commercial thinning

Commercial thinning is an intermediate silviculture entry into an even-aged forest that removes sufficient trees of marketable size to cover the direct costs of harvest (Ford-Robertson 1971). Commercial thinning is intended to improve the health, vigor, and future growth capacity of a forest that is not yet ready to be regenerated. Establishment of new seedlings is not a goal. Commercial thinning can be designed to satisfy a number of management goals, each of which can determine the types, numbers, and patterns of trees that are removed. Commercial thinning can remove

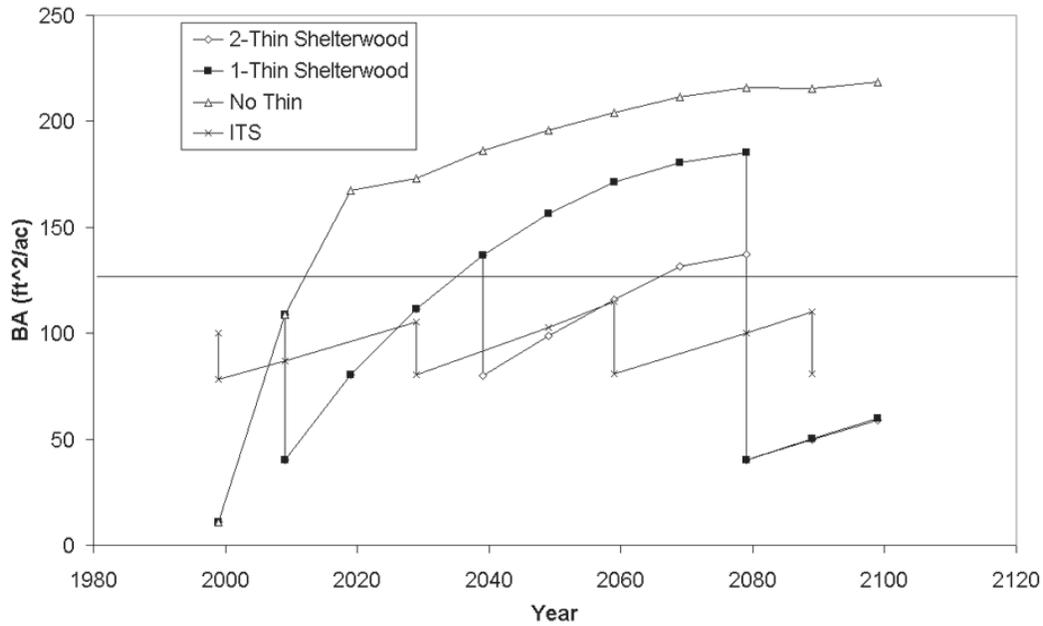


Figure 45: Forest Vegetation Simulator (Crookston 1997) projections of basal area growth over time for selected Black Hills ponderosa pine management schemes. The even-aged simulations all began with 2000 1-inch trees present in the year 2000. The 1-Thin option precommercially thinned to 40 BA in 2010. The 2-Thin option added a second commercial thinning to 80 BA in 2040. A seed-cut shelterwood harvest was applied to both options in 2080. Initial conditions for the uneven-aged individual tree selection (ITS) option assumed 100 BA distributed across 2- to 26-inch d.b.h. classes under a $Q = 1.3$. Stocking was reduced to 80 BA in each of four entries at 30-year intervals, beginning in 2000. All projections used Site Index = 70. The horizontal line at 120 BA represents stocking levels at high risk of attack by mountain pine beetles (Schmid and others 1994).



Figure 46: An extremely high-density stand of naturally regenerated ponderosa pine in the Black Hills Experimental Forest.

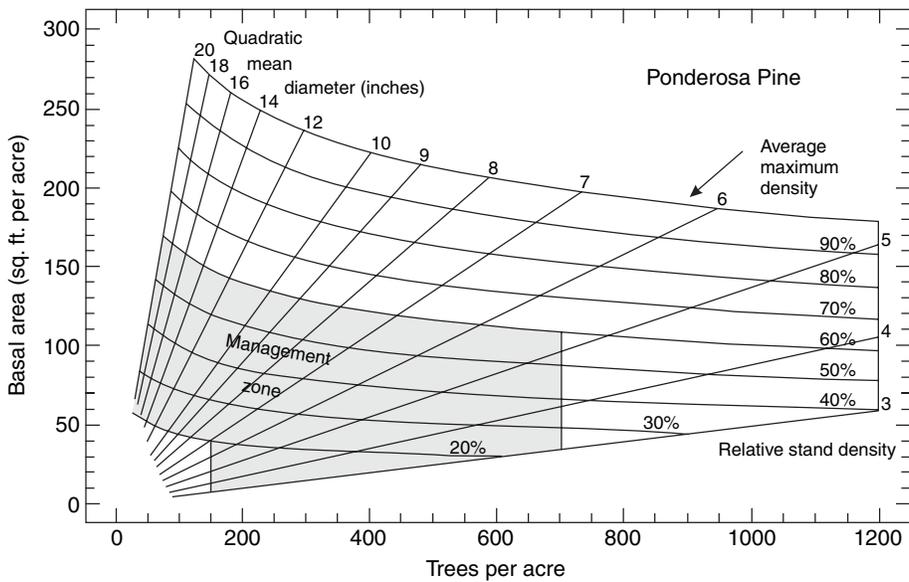


Figure 47: Gringrich stocking curves for ponderosa pine in the Forest Service's Rocky Mountain Region (R2) (Forest Service Manual, R2 Supplement 2409.17-92-2 1992).

smaller, suppressed trees, those with poor form or disease, or remove undesirable species from a mixed-species forest. The number of trees removed can vary widely, but is usually dictated by a residual stocking guideline determined by specific management objectives for the site. One such guideline that has been adopted by the U.S. Forest Service is the Gringrich thinning curve developed for Black Hills ponderosa pine (fig. 47). Stocking levels within the management zone illustrated on this Gringrich chart are considered suitable for a variety of resource uses within the Black Hills. Stocking levels below the management zone are considered less than fully occupied (for example, not growing at acceptable levels for a managed forest on the site). Thinning to these levels would create trees of open-grown crown form and would result in the

establishment of new seedlings in the openings between trees, thus creating a new age class (not a desirable result of an intermediate thinning under even-aged management).

Stocking levels above the management zone on the Gringrich curve are not considered sustainable for many resource uses. Forests at these densities grow considerably more slowly and are susceptible to density-dependent mortality, insect and disease attack, or stand-replacing wildfires. Understory vegetation is almost nonexistent, resulting in depauperate habitats for many wildlife species.

Although not absolutely necessary to maximize fiber yield (fig. 48), at least one commercial thinning would be advantageous in maximizing product yield on many even-aged stands in the Black Hills. Most

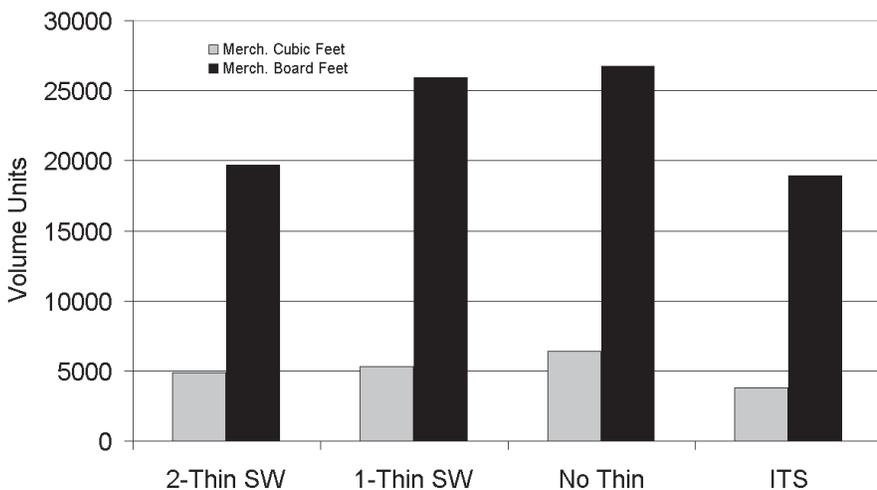


Figure 48: Total projected merchantable cubic and board foot yields over the 100-year period from the Black Hills ponderosa pine management schemes shown in figure 45.

even-aged ponderosa pine forests in the Black Hills will require at least one thinning entry in a rotation to remain healthy. The level and timing of thinning is a factor of the specific stand conditions, management guidelines for the site, and specific site growth characteristics. SUPPOSE, the PC-based Forest Vegetation Simulator (FVS) growth and yield model (Crookston 1997), can be used to evaluate thinning alternatives on a site-by-site basis.

Maximization of merchantable board foot yield

Tree spacing in precommercial thinning and subsequent commercial thinning all affect merchantable board foot (MBF) yield. Using the FVS growth simulation model, we compared total rotation MBF yields under several scenarios for a typical even-aged Black Hills ponderosa pine stand. The following stocking criteria were used: site index = 62; initial stocking = 3,000 trees per acre; planted at year 0, reaching d.b.h. at year 10; 120-year rotation. All scenarios were based on precommercial thinning at year 30. We attempted to commercially thin if a minimum 1,000 MBF per acre could be removed without decreasing the overall MBF yield. Our goal was to maximize yield without undue risk to mountain pine beetle loss. We judged stands to be at risk if basal areas exceeded 120 ft² per acre (27.6 m² per hectare) and quadratic mean diameter (QMD) was greater than 9 inches (22.9 cm).

Merchantable board foot yield was maximized at the 9 foot by 9 foot (2.7-m by 2.7-m) precommercial thinning spacing with no subsequent commercial thinning (table 5). However, the extremely high basal areas

under this scenario would make it highly susceptible to mountain pine beetle (fig. 18). In addition, tree size was too small at the 9 foot by 9 foot (2.7-m by 2.7-m) spacing to allow a commercial thinning before the end of the rotation. We were able to model a commercial thinning with the 10 foot by 10 foot (3.1-m by 3.1-m) spacing, but the stand was still susceptible to beetles and the overall yield dropped slightly (table 5). The 12 foot by 12 foot (3.7-m by 3.7-m) precommercial thinning spacing with a subsequent commercial thinning to remove all MBF volume at age 80 proved to be the best compromise between maximizing merchantable board foot yield and minimizing mountain pine beetle risk. Under this scenario, board foot volumes were 3,900 MBF less than under the 9 foot by 9 foot (2.7-m by 2.7-m) spacing (table 5). If a more open stand is desired, a precommercial thinning to 20 foot by 20 foot (6.1-m by 6.1-m) will produce the largest diameter trees with little risk of beetle infection, but total MBF yield will be about one-third less than the previous scenario (table 5). We tried a number of additional thinning scenarios at various spacings, but none performed better than the ones listed here.

We should note that these scenarios only apply to the conditions listed above and are included here for illustrative purposes only. Readers should not assume that these results would directly apply to every ponderosa pine stand in the Black Hills. Irregular spacings, site conditions, ages at thinning, presence of additional age classes, and so forth, will all affect yields. We therefore urge managers to utilize FVS to model scenarios for their local conditions.

Prescribed burning

Prescribed fire has not been used to a great extent to eliminate regeneration in the Black Hills, but it is a technique that should be explored further. Prescribed burning would best be done when seedlings are quite small and fuels ladders are either infrequent or non-existent to avoid loss of larger trees. Under even-aged systems, prescribed burns can be done while seedlings are essentially in the grassy stage or less than 1 foot tall to eliminate the densest concentrations of seedlings. In uneven-aged systems, excess regeneration established after one cutting cycle could be removed by burning prior to the next entry cycle. For example, an individual-tree uneven-aged stand that was harvested on the Black Hills Experimental Forest in the mid-1980s is currently overstocked in the understory size classes, even though it is only halfway through its cutting cycle (fig. 43). Careful underburning in this stand could have eliminated many of these trees before they got large enough to become a problem.

Table 5: End of rotation per acre, tree density (TPA), basal area (BA), quadratic mean diameter (QMD), and total merchantable board foot yields (MBF) for various precommercial tree spacings (feet) and two subsequent commercial thinnings.

Treatment	TPA	End of Rotation Values		
		BA	QMD	MBF
No Thin	627	230	8.2	8,652
7x7	554	236	8.8	14,549
9x9	452	227	9.6	16,848
10x10	356	195	10.0	15,891
12x12	247	157	10.8	14,462
15x15	163	122	11.7	12,945
20x20	96	85	12.8	9,251
10x10 Com. Thin	288	154	9.9	15,542
12x12 Com. Thin	174	117	11.1	13,473

Prescribed burning can also efficiently remove fuel loadings from logging slash and stimulate understory forage production. In areas with heavy needle and litter accumulations around the base of ponderosa pine trees, the fuel loadings should be raked away from the base because these fuels can smolder at the base of the trees for long periods of time, and damage pine roots (Sackett 1980). Proper timing of prescribed burns to maximize soil moisture can alleviate some of this damage.

Careful burning under the right conditions can scorch the lower branches of trees without consuming the entire crown and killing the tree. Such scorching can kill lower branches and effectively prune the tree, improving its form and decreasing the chance of crown ignition from future fires (fig. 49).

Uniform prescribed burning could reduce structural diversity of forests, but burning in a manner that does not blacken the entire forest floor can actually increase diversity. Therefore, leaving portions of a stand

unburned will likely come closer to mimicking the natural fire regimes that occurred in the Black Hills prior to European settlement. For the same reason, the loss of occasional overstory trees, or occasional groups of trees, during a prescribed burn should not be considered detrimental. Unlike mechanical silviculture treatments, prescribed burning is not exact. It cannot guarantee the removal or survival of individual trees. It can only attempt to improve the present condition of the forest. Nor can burning be applied in as timely a fashion as mechanical treatments. Weather, fuel and soil moisture conditions, and regional smoke restrictions can severely restrict when burning can occur, leading to delays in treatment of several years or more. Prescribed fire should therefore not be considered a panacea in the management of ponderosa pine, but only one of many tools that can be used to manage pine ecosystems in the Black Hills.

Artificial Reforestation

Principles

Natural regeneration is the preferred method of regenerating ponderosa pine trees in the Black Hills due to the relative abundance of precipitation throughout the growing season, as well as the dependable seed production of ponderosa pine in this region. However, natural regeneration is sometimes not an option due to a large-scale fire, tornado, or other catastrophic disturbance that eliminates seed sources within the area to be reforested. Both seeding and planting can be used in the Black Hills. Successful seeding depends on several conditions, including a suitable seedbed, ample seed supply, and an environment conducive for germination and establishment of seedlings. Planting success depends on healthy local seedling stock, proper soil moisture, and correct handling and planting techniques.

Tree Improvement Program

The Black Hills National Forest has a tree improvement program in progress to enhance the productivity of the forest by establishing well-adapted, fast-growing, pest- and disease-resistant trees on productive sites. The program includes activities for seed collection to develop an adequate seed inventory of source-identified seeds, establishment of seed production areas to provide seeds for specific zones within the Black Hills National Forest, and efforts to locate the superior ponderosa pine parent trees identified by Van Deusen and Beagle (1970) for use in progeny studies.



Figure 49: Prescribed burns can lessen the risk of crown fires by killing and pruning lower branches.

The majority of the Black Hills National Forest falls within three seed zones: 621, 622, and 623. A seed production area, Buskala Seed Production Area, has been established on the Spearfish/Nemo Ranger District near the boundary of seed zones 621 and 622. This area will have source-identified seeds from better-than-average trees to provide a local seed source for artificial regeneration in zones 621 and 622. Seed production areas for other zones within the Black Hills National Forest are currently in the process of being established.

In the late 1960s, Van Deusen and Beagle located and described cone and seed characteristics of several superior trees (better than average appearing) among six regions of the Black Hills (Van Deusen and Beagle 1970). The collection regions included the northern hogback, northern limestone, metamorphic, granitic, southern limestone, and southern hogback. Seedlings produced from the seed sources were planted in a common plantation on the Black Hills Experimental Forest. Ten-year progeny growth results were reported by Shepperd and McElderry (1986). Six parent seed sources for the western and northern Black Hills exhibited exceptional survival and growth. The progeny plantation still exists on the Black Hills Experimental Forest, so it is possible that seeds could be collected from well-performing progeny. The Black Hills National Forest staff is now in the process of relocating the original parent trees to collect cones and map their locations. If any of the trees are dead, missing, or have undesirable traits, new ones will be selected to achieve a target of 30 plus trees per district on the National Forest. Locating and collecting seed from the identified superior ponderosa pine trees will allow for a greater genetic diversity for artificial regeneration.

Site Preparation

Proper site preparation can increase the success of both direct seeding and planting. Site preparation should be initiated as soon as possible after the disturbance that eliminated the seed source for natural seedlings. Grasses, forbs, and shrubs can inhibit seedling establishment. Competition from other plants may reduce survival of planted seedlings in burned or logged areas by 30 percent or more for each year planting is delayed (Heidmann 1988; Stewart 1978). If a hardwood overstory is present and the goal is to manage for ponderosa pine, the hardwood overstory should be reduced or eliminated to reduce competition for resources (Boldt and Van Deusen 1974).

There are four primary methods of site preparation for ponderosa pine: mechanical, chemical, prescribed

burning, and a combination of the three (Heidmann 1988). Hallman (1993) lists numerous types of specialized equipment for site preparation along with their function, description, advantages, disadvantages, and places to obtain them. The purposes of mechanical site preparation are to remove plant competition and slash, loosen compacted soil, and scarify the forest floor (Hallman 1993). Scalping 18 to 24 inches in an area to be planted to reduce grass competition is a common practice in the Black Hills. Some important things to consider while using heavy equipment for mechanical site preparation include soil compaction, loss of topsoil, and cost effectiveness.

Herbicides are often used to control vegetation that would otherwise compete with pine seedlings for moisture, nutrients, and sunlight. Herbicides are often used in concert with other site preparation techniques such as prescribed burning. The use of herbicides has been limited, however, due to concerns about the unknown and often controversial toxicity of the chemicals (Hallman 1993). Although the potential exists, herbicides have not been used extensively to control competing vegetation when planting or seeding ponderosa pine in the Black Hills.

Prescribed fire is another way to remove plant competition and slash (Hallman 1993). The fire temperature must be hot enough to kill the vegetation (Heidmann 1988), but cool enough to lessen the loss of volatile soil nutrients. Sites should be burned when the fuel is dry but the soil is moist enough to avoid oxidation of the nutrients and soil organic matter. An exposed mineral soil seedbed is conducive to ponderosa pine seed germination. To minimize soil erosion on slopes after a prescribed burn, sites should be burned when conditions are beneficial for germination and establishment of pine seedlings. Typically, burning in spring prior to the growing season will best meet these requirements. Other mechanical treatments can be used to reduce soil erosion following prescribed burns. Monitoring of the area for invasion of noxious weeds is warranted, especially in areas that were sodbound. Hallman (1993) lists numerous types of specialized equipment for treating prescribed burns, along with their function, description, advantages, disadvantages, and places to obtain them.

Establishing pine regeneration by seeding areas that were subjected to intense wildfire that eliminated the overstory might be difficult due to increased soil temperature. Surface temperatures during the first growing season are often too high for seedling establishment due to the blackened soil, especially on southern and western aspects.

Seeding

Seed collection

Ponderosa pines in the Black Hills usually produce a good to excellent seed crop every 2 to 5 years (Boldt and others 1983). Seeds should be collected from trees that display desirable traits of growth, form and resistance to pests. Favorable tree characteristics identified by Boldt and Van Deusen (1974) include: growth rate, age, stem, branches, bark, and health. Trees that equal or exceed the average site index height for their age and have an average diameter growth of 2 inches (5 cm) or more per decade indicate a superior growth rate. Vigorous growth is indicated by dark bark and wide fissures that show a bright orange color. Trees that are 75 to 150 years old should be chosen. These trees should have a straight stem without any forks or excessive taper for good form. Branches should be small in diameter, short in length, average four or less branches per whorl, nonpersistent after death, and be at a right angle to the bole. Live crown should be 40 to 60 percent of the total tree height. Look for trees without any evidence of successful attack by insects, diseases, or porcupines (*Erethizon dorsatum*). Needles should be green, and there should be evidence of good cone production in past years. To ensure that the seed collected have the desirable characteristics, seeds should come directly from cones on the trees, rather than those from squirrel caches (Boldt and Van Deusen 1974). If the objective of the seed collection is to promote genetic diversity, however, then collection from squirrel caches is fine.

Van Deusen and Beagle (1970) described the cone and seed characteristics of several superior ponderosa pine trees among six regions of the Black Hills. Over the entire area, the average cone length was 2.65 inches (6.7 cm) with a range of 0.44 inch (1.1 cm). The number of seeds per pound was variable within a collection area as well as between regions. One could expect to find from 8,000 to 23,000 seeds per pound of cones collected. There was a trend toward an increase in average number of seeds per pound from south to north. Viable seeds per cone averaged around 57.

Seeds are extracted from cones by shaking or tumbling. It is estimated that 6 to 19 seeds may be lost per cone using this method. Because there are year-to-year variations in cone and seed characteristics, a seed collector should first sample the current cone crop before making a mass collection. Cones are ripe when the color turns from green to purplish brown (Krugman and Jenkinson 1974). In addition, the specific gravity of cones containing ripe seeds in the Black Hills is 1.00 (Oliver and Ryker 1990).

Seeds are typically dispersed from September to January.

Seed storage

Viable ponderosa pine seeds must be dried to a moisture content of 4 to 8 percent, placed in an airtight container, and stored at 32 °F (0 °C) or less. Storage at 0 °F (-18 °C) is preferred (Schubert 1974). Seeds should be viable for up to 15 years (Krugman and Jenkinson 1974).

Germination treatment

Stored seeds germinate better if they are subjected to cold stratification (Krugman and Jenkinson 1974). To stratify seeds, first place them in water for 1 or 2 days, then in a moist medium and maintain temperature around 33 to 41 °F (0.6 to 5 °C) for 30 to 60 days. To test for seed viability, germinate a sample of the seeds under near-optimum conditions of aeration, moisture, temperature, and light. Published germination tests for ponderosa pine have used a variety of growth media, temperature, and light regimes (Krugman and Jenkinson 1974). Germinating stratified seed in Petrie dishes on moistened blotter paper under a diurnal regime of 8 hours of light at 86 °F (30 °C) and 16 hours of darkness at 68 °F (20 °C) should give consistent results. Tests should be run for 30 days to ensure complete germination. To avoid mold, seeds should be washed in a 3 percent chlorine bleach solution and rinsed in distilled water before being placed into the Petrie dishes, and removed as soon as they germinate.

Sowing techniques

Several devices and techniques for sowing seed have been used or considered in the Black Hills, but none have been evaluated experimentally (Boldt and Van Deusen 1974). Hallman (1993) discusses several pieces of equipment to carry out direct seeding and their advantages and disadvantages.

Direct seeding has been used successfully since 1905 for reforestation in the Black Hills (Boldt and Van Deusen 1974; Harmon 1954; Wagar and Meyers 1958). This technique can be fast and economical if properly done. Direct seeding typically costs less than hand planting, but survival is usually higher for hand-planted seedlings.

Broadcast seeding is one of the simplest methods of direct seeding; seeds are scattered uniformly over the area to be revegetated. Broadcast seeding has been successful in the Black Hills (Boldt and Van Deusen 1974), but one problem is the potential for multiple seeds to be placed in one spot. When optimum climatic factors allow germination and survival

of too many seeds, the resultant clusters of young pines will require thinning (Wagar and Meyers 1958).

Factors for successful direct seeding were identified by Wagar and Meyers (1958) for the Black Hills, with precipitation being one of the most important. Stocking was highest when seeds were sown during periods of abundant precipitation. For instance, seedings that resulted in at least 50-percent stocking received on average 8 inches (20 cm) of precipitation from October to April and about 15 inches (38.1 cm) from May to September during each of the first 3 years after sowing. In contrast, stocking averaged less than 10 percent when precipitation was 6 inches (15 cm) from October to April and 11 inches (28 cm) from May to September. Distribution of precipitation is also important. Seedling mortality can be high if there are hot, dry periods during late summer and early fall within the first 3 years of sowing. Greenhouse studies have shown that ponderosa pine seedling survival and growth are best when seedlings receive 1 inch (2.54 cm) or more of water evenly distributed throughout the month (Noble and others 1979).

Topography and soil influence soil moisture availability and thus seedling success. Wagar and Meyers (1958) reported survival to be highest on lower, gentle, north and east slopes where soil moisture was most favorable. Southern and western aspects typically had lower survival, especially on slopes greater than 30 degrees, which often have thin soils. Recently burned areas are likely to have higher erosion rates. Even limestone soils with 12 percent slopes and metamorphic soils with slopes of 20 percent have shown severe erosion and loss of seeds (Wagar and Meyer 1958).

Competition with grasses and forbs does not seem to be a problem in years of abundant precipitation (Wagar and Meyers 1958). In dry years, however, survival was low and the survivors averaged smaller than seedlings on adjacent bare ground. Seeded pines do not compete well with hardwoods such as aspen, birch, or bur oak.

Broadcast seeding, spot seeding, and sowing with a corn planter give similar survival results. The amount of seed varies depending on the sowing method. For spot seeding, 0.2 to 2.5 pounds of seed (12,000 ponderosa pine seeds per pound) per acre is sufficient. With spot seeding, seeds can be situated in superior microsites and covered with soil to improve germination and lessen losses to rodents (Schubert 1974). Broadcast seeding requires more seeds with at least 4 to 6 pounds per acre when conditions are favorable (Wagar and Meyer 1958). Broadcast seeding can be performed from the air or the ground. Germination

success can be improved by preparing a loose seedbed or by disking after sowing (Schubert 1974).

Seed protection

Because ponderosa pine seeds are eaten by many mammals and birds in the Black Hills, predation on seeds can influence the outcome of reforestation attempts. Early attempts to control seed predation by protecting seeds with chemicals and by trapping animals have been shown to be costly and ineffective (Keyes 2000). Recent efforts are focusing on creating strategies that utilize our knowledge of feeding habits and life cycles of seed predators.

Seed predators occur in all phases of forest development, with the greatest population densities in the early seral stages (West 1992). Small mammals, such as chipmunks, squirrels, and deer mice (*Peromyscus maniculatus*) cause the most seed loss in ponderosa pine direct seeding programs (Evans 1988). Small mammals have predictable cyclic population patterns, with high densities in the fall and low densities in the spring (West 1992). Birds such as Merriam's turkey rely heavily on ponderosa pine seed for winter forage. Thus, direct seeding should occur in the spring when small mammal populations are lower and other forage is available.

Several methods can be used to determine the probability of seed predation. Direct sampling of small mammal populations with Sherman live traps will provide an estimate of small mammal density (Keyes 2000). A relatively inexpensive way to quantify seed consumption is to use small, mesh-covered exclosures that will allow seeds to fall through, but keep animals from reaching them. Comparing the number of seeds in the exclosure to an uncovered area next to it can indicate consumption levels (Gashwiler 1967).

If seed predation is causing a regeneration problem, there are several management options for mitigation. Options include reducing per-capita seed consumption, avoiding large populations of seed predators, and reducing seed predator populations (Keyes 2000). A manager can reduce the per-capita seed consumption of ponderosa pine seed by providing alternative food sources, such as sunflower seeds. This technique has been shown to increase lodgepole pine seed survival by 30 to 60 percent (Sullivan and Sullivan 1982). If this method is used, alternative food sources must be continued until an acceptable number of seedlings are established (Keyes 2000). Any seed to be used as an alternate food source should first be rendered nonviable to avoid introduction of invasive nonnative plants.

Reducing large population levels of small mammals can also reduce seed consumption. Poisoning small mammals is a short-lived approach because recolonization is rapid. Furthermore, the use of poison can also kill nontarget animals. A better way of reducing small mammal population levels is to reduce the habitat they rely on. Reducing woody debris and shrub cover will decrease the habitat quality and may increase predation by larger predators (West 1992). Per-capita mammal seed consumption has been shown to increase during good seed years, but with more of available seeds, the consumption is compensated (West 1992). Therefore, the success of seeding operations to regenerate ponderosa pine can be enhanced if they are scheduled during good seed years.

Lastly, seed predation should be seen as another natural process that occurs in an ecosystem. The influence of seed predators in the ponderosa pine forest is not all negative. In fact, ponderosa pine seed is an important food source for many animals and birds. Accommodating seed predation is often the most realistic approach.

Planting

Raising seedlings at a nursery and subsequently planting them on the site to be reforested can avoid many of the problems associated with artificial seeding of ponderosa pine (Hallman 1993). Seedlings grown in nurseries are selected from seed sources that exhibit good form, growth, and disease resistance. The controlled conditions in nurseries eliminate many of the uncertain growing conditions in the field and associated early seedling mortality. Unhealthy and poorly formed seedlings are culled before seedlings are shipped from the nursery. Although planting seedlings is much more expensive than seeding, the rate of reforestation is quicker because healthy, vigorous seedlings with established root systems are planted.

Successfully planting seedlings is dependent on the ability of the planter to do the job correctly. There are many steps in getting a seedling from the nursery into the ground at the planting site, and careful handling of the nursery stock is of utmost importance. It is also imperative that seedlings to be planted in the Black Hills should come from a seed source within the Black Hills, ideally from as close to the planting site as possible.

The absence of dwarf mistletoe (*Arceuthobium campylopodum*) in the Black Hills is another reason to be extremely careful about the source of planting stock. Dwarf mistletoe infects ponderosa pine forests all over the Western United States, but the Black Hills so far

has not been infected. The potential for dwarf mistletoe to be introduced to the Black Hills is greatly increased when seedlings are grown among other seed sources in nurseries outside the Black Hills. Any seedlings grown at an outside nursery and transported to the Black Hills should be carefully inspected for dwarf mistletoe.

Bare-root seedlings

Bare-root seedlings are grown in exposed seedbeds, lifted from the soil, and shipped to the planting site with the roots separated from the soil (Tinus and McDonald 1979). Sometimes seedlings are transplanted into another seedbed to make the root systems more compact and to create more fine lateral roots. Seedlings are classified based on age and treatment designated by two numbers. The first is the number of years the plants grew as a seedling within the same seedbed; the second is the number of years grown as transplants. Therefore, a bare-root seedling designated a “2-1” means it has grown for 2 years as a seedling plus 1 year as a transplant. A bare-root seedling designated a “2-0” is a 2-year-old seedling grown for 2 years in the same bed without transplanting (Smith and others 1997).

To date, only one study (Boldt 1968) has investigated which class of ponderosa pine planting stock should be used for Black Hills plantings. Boldt examined the survival and height growth of 1-0, 2-0, 1-1, and 2-1 planting stock on two burned sites in the Black Hills. One site represented an area marginal for planting with only 18 inches (45.7 cm) of annual precipitation, while the other site had high potential for timber production with 24 inches (61 cm) of annual precipitation. After the third growing season, survival was greater than 70 percent for all planting stock classes on both sites. The trees showed substantial differences in average height at the end of the fifth growing season, however. On the marginal site, the average heights were 10.9, 15.4, 18.8, and 17.1 inches (27.7, 39.1, 47.8, and 43.4 cm) for the 1-0, 2-0, 1-1, and 2-1 stock classes, respectively. The favorable site had average heights of 14.7, 17.7, 20.1, and 23.0 inches (37.3, 45, 51.1, and 58.4 cm) for the stock classes 1-0, 2-0, 1-1, and 2-1 respectively. Boldt recommended that, because 2-0 planting stock appears to provide the best combination of economy and performance it should be used as the standard for Black Hills plantings. However, the author did suggest further research to determine if 2-0 stock will perform as well during dry years, and if not, whether 2-1 stock might be better.

Increasing survival and growth of bare-root seedlings—Several studies have investigated techniques to

increase survival and growth of bare-root ponderosa pine, but none were conducted with Black Hills seed sources or sites. Until these issues can be studied for the Black Hills, planters should utilize the information provided from other studies.

The use of bare-root seedlings for reforestation involves several steps that can stress the seedling and result in lower survival rates. These steps include lifting, processing, storing, shipping, field handling, field storing, and planting (Rietveld 1989a). With proper procedures and care, the loss of growth potential can be minimized.

Root volume—One of the most advantageous aspects of nursery-raised seedlings is that they have already passed the critical early stage of development; the seedling already has a stem, shoots, buds, and a root system. For a seedling to survive, the root system must be viable and able to grow fast enough to maintain contact with the soil moisture supply. Seedlings with larger root systems have better survival, growth, and drought resistance. One study in Oregon showed that, after two growing seasons, ponderosa pine seedling survival and growth were significantly greater for seedlings that were planted with initial root volumes of greater than 0.43 inch³ (>7 cm³) compared to root volumes less than 0.28 inch³ (<4.5 cm³) (Rose and others 1991).

Shoot-root ratios — Don't choose a seedling based solely on its height. Instead, choosing a seedling with a large root system and a relatively small top appears to be advantageous (Heidmann 1988). Shoot-root ratio is a good indicator of internal water stress and potential survival of an outplanted seedling because shoot and root sizes directly affect water loss and uptake. Lopushinsky and Beebe (1976) reported that low shoot-root ratios increased survival and height growth of outplanted ponderosa pine seedlings on a dry site in north-central Washington. The authors suggest that the seedlings with lower shoot-root ratios survived better because (1) desiccation and moisture stress immediately after planting were minimized, and (2) later internal water stress was low enough to permit some root growth, which improved the absorptive ability of the root system. Seedlings with a high shoot-root ratio will have a greater transpiring surface in relation to the root's surface area, which could be detrimental during a drought (Heidmann 1988).

Root growth potential—The ability of seedlings to grow new roots is referred to as root growth potential (RGP) (Ritchie and others 1985). RGP is used to estimate planting stock quality because it incorporates several morphological and physiological features into a biologically meaningful estimate of the ability of a

plant to grow new roots (Rietveld 1989a). High RGP values in ponderosa pine are strongly correlated with high field survival (Heidmann 1988). Primary techniques for estimating RGP include soil culture, hydroponic culture, aeroponic culture (Rietveld 1989b), and volume displacement (Harrington and others 1994).

Lifting and storage of bare-root seedlings—Seedlings are often lifted at nurseries and stored over winter in freezers so that they are readily available to customers in the spring. In nature, seedlings undergo vegetative dormancy as temperatures lower and photoperiod shortens (Hocker 1979). To break from this dormancy, seedlings must experience some exposure to low temperatures. Stone (1970) reported that ponderosa pine seedlings require exposure to night temperatures at or below 43 °F (6 °C) for at least 60 days to achieve maximum root growth after dormancy. Fall lifting and long-term storage have several operational and management benefits (Hocking and Nyland 1971 cited in Omi and others 1991), but they can be detrimental to plant growth and development.

Seedlings should be lifted at the peak of their dormancy to best survive long-term storage (Sloan 1991). Long-term storage can reduce seedling quality because seedlings are no longer exposed to photoperiods (Lavender and Wareing 1972), temperatures may be too cold to satisfy chilling requirement, seedlings may become desiccated, and carbohydrate reserves decline (Falconer 1988; Ritchie 1982, 1986). Loss of carbohydrate reserves may lead to reduced root initiation, thus lowering seedling survival and growth when outplanted. Furthermore, some beneficial mycorrhizae fungi, such as *Pisolithus tinctorius*, do not survive cold storage (Alvarez and Linderman 1983).

In general, ponderosa pine seedlings lifted in November have good survival rates (Omi and others 1991, 1994; Stone and Schubert 1959). A preliminary study at Lucky Peak Nursery in Idaho indicated that although ponderosa pine seedling survival after two growing seasons was similar for seedlings lifted in October, November, December, March, and April, growth for the same period was greatest for seedlings lifted in November, December, and April (Sloan 1991). The seedlings lifted in mid-April had the greatest growth, which the authors attributed to the short 2-week storage time before planting. Another study of an Oregon seed source indicated that lifting ponderosa pine seedlings in September followed by long-term storage lowered survival after two growing seasons, while lifting in November was as good as or better than spring lifting (Omi and others 1994).

Root dipping—Roots are commonly dipped in a variety of substances to prevent bare-root seedlings

from drying out in storage, desiccating at the planting site, and to improve seedling survival and growth after planting (Sloan 1994a). Substances used for root dips include: soil slurries, vermiculite or ground sphagnum moss, hydrophilic gels, and other materials such as organic compounds, pesticides, or other chemicals (Sloan 1994a). These substances are coated on the root system. Studies investigating the survival and growth of bare-root ponderosa pine seedlings have been limited to root dips consisting of vermiculite (Ryker 1981; Sloan 1994b), vermiculite slurry (Sloan 1994b), peat moss (Ryker 1981), hydrophilic gels (Ryker 1981; Sloan 1994b; Tuskan and Ellis 1991), and hygroscopic gels mixed with auxin (Tuskan and Ellis 1991).

Vermiculite is an expanded mica substance that can hold water tightly and resist drying when coated on a root system (Sloan 1994a). Hydrophilic gels can absorb between 40 and 500 times their weights in water depending on the type of solution. Thus far, studies using vermiculite or the hydrophilic gels, Aquagel™ (Sloan 1994b), Viterra™ (Ryker 1981) and WaterLock Superabsorbant Polymer B-204 gel (Tuskan and Ellis 1991), have not shown an increase in ponderosa pine seedling survival or growth. Tuskan and Ellis (1991) did show an increase in outplanted seedling growth and survival using a hydrophilic gel fortified with auxins at concentrations of 100 and 300 ppm indole-3-butyric acid (IBA). Field survival of the outplanted seedlings with the fortified gel was 15 percent higher than the seedlings dipped in the gel without auxin, and 28 percent higher than for the nondipped seedlings.

Transporting and handling bare-root planting stock—Planting stock should be packaged so that roots remain moist and cool while allowing oxygen to enter the package and carbon dioxide and other gases to escape (Smith and others 1997). The seedlings should be refrigerated at temperatures slightly above freezing. Proper precautions should be taken to prevent desiccation of bare roots due to freezing temperatures. Seedlings should be kept under refrigeration until ready to plant. Under no circumstances should bare-root seedlings be transported in open trucks, or allowed in the sun in their unopened transport bags. Lethal temperatures can occur inside the bags after just a few minutes exposure to full sunlight. Physical damage while handling seedlings can hinder seedling growth and survival after planting. The most common damage is the loss of fine roots, which are important for water absorption and carbohydrate storage (Rietveld 1989a).

Containerized seedlings

Containerized seedlings are grown in small soil-filled containers, usually inside a controlled-environment

greenhouse, and are delivered to the planting site still in the containers in which they were grown. Seedlings are removed from the containers as they are planted. Containerized seedlings undergo less transplanting shock because there is minimum root exposure and disturbance (Kinghorn 1974). Survival and growth are usually increased due to the constant contact of the roots with soil, and the nutrients and water-holding capacity of the container media. Although production of containerized seedlings is more expensive and transport of seedlings is more cumbersome, increased survival and growth can make up for the disadvantages. This is especially relevant on dry, harsh sites that are difficult to regenerate. The container tree nursery manual published by the USDA Forest Service lists several types of containers (Landis 1990a) and growth media (Landis 1990b).

Controlling root growth and form—Narrow, deep containers are most compatible for ponderosa pine because they produce taproot systems (Tinus and McDonald 1979). However, there is concern that the roots in such a constricted space will start growing in a spiral and become “pot-bound” when they hit the walls of the container. While root spiraling is not detrimental to seedling growth in the nursery, it can seriously degrade seedling quality after outplanting (Landis 1990a).

Several features in newer containers minimize root spiraling. Some containers contain vertical internal ribs to reduce root spiraling and possible root strangulation (Smith and others 1997; Tinus and McDonald 1979). The ribs help direct the roots to the bottom of the container, which has a hole in the bottom. The low humidity of the air beneath the hole stops the roots from growing any further, but the roots retain the ability to grow when the seedling is planted (Landis 1990a). Although this technique prevents root spiraling, the distribution of the root system in the container does not correspond to that of naturally grown seedlings.

Ponderosa pine seedlings found in nature produce long lateral roots within the top 12 inches (30.5 cm) of the soil (Stein 1978), which provide a foundation to aid in windfirmness. In contrast, seedlings grown in containers have lateral roots that are forced to grow downward, producing a dense, fibrous root mass bunched around the taproot in the upper soil layers. When the seedling is planted, new lateral roots are produced by growth of new root tips from the cut ends of the existing laterals initiated higher up the stem (Stein 1978).

McDonald and others (1984a) reported that coating the inside of containers with a mixture of acrylic latex paint and CuCO₃ (concentration 50 to 100 grams per

liter of paint) halted lateral root growth at the container wall. Inhibition of the initial lateral root growth leads to production of secondary lateral roots, which are once again halted when they reach the container wall. The resulting roots systems are more fibrous and evenly distributed throughout the container (Landis 1990a). Containers treated with CuCO_3 promote better seedling stability after outplanting due to the production of roots in the middle and upper zones of the container (Wenny 1988). Normal root growth resumes when the seedling is removed from the container and a more natural, branched root system is produced after outplanting (McDonald and others 1984a). It should be noted that the acrylic latex paint was phytotoxic to the seedlings, but the paint's effects were overridden by the CuCO_3 in the tests reported by McDonald and others (1984a). In another study, McDonald and others (1984b) reported that a combination of CuCO_3 (concentration 50 grams per liter of acrylic latex paint) coated on the side of the container wall and inoculation with ectomycorrhizal fungi (*Suillus granulatus* and *Pisolithus tinctorius*) increased the formation of ectomycorrhizae on the roots of ponderosa pine seedlings.

Growth conditions—Growing containerized seedlings in the greenhouse allows greater control of environmental and nutrient conditions. Larson (1967) investigated the effect of air and soil temperature on initial development of ponderosa pine seedlings from the Black Hills. Root growth responded more to soil temperature, while top growth responded more to air temperature. The optimum air and soil temperatures for initial root growth of ponderosa pine were near 59 °F (15 °C) air and 73.4 °F (23 °C) soil, but a constant temperature of 73.4 °F (23 °C) was more favorable for entire seedling growth than any combination of alternating day and night temperatures. Based on these results, Tinus and McDonald (1979) recommend a constant greenhouse temperature setting of 73.4 °F (23 °C) with an allowable range of 68 to 75 °F (20 to 24 °C).

Optimal greenhouse humidity for germination is 80 percent, but once the radicle has penetrated the soil surface, optimum humidity is 60 percent (Tinus and McDonald 1979). Maximum photosynthetic rates are obtained when light levels are between 775 to 1550 $\mu\text{mol m}^2/\text{s}^{-1}$ (Hadley 1969).

Handling and transporting containerized seedlings—Transporting containerized seedlings depends on the type of container in which the seedlings are grown. If the seedlings are grown in tubes or Styrofoam blocks, the entire container is shipped. Shipping containers require more shipping space and the containers must be returned to the nursery, which increases the

cost. The roots are highly protected, although the shoots must still be carefully handled. If the seedlings are grown as plugs, the seedlings can be transported with or without the containers. If the plugs are transported without the containers, less shipping space is required, but the stock quality can be seriously impaired during cold storage and handling if seedlings are not completely dormant (Brissette and others 1991).

Seedlings should be transported in either cold storage (33.8 to 37.4 °F; 1 to 3 °C) or in a covered vehicle that protects them from direct sunlight and wind. The number of seedlings transported and stored at a planting site should be limited to the number that can be planted in one day. Handling of seedlings should be limited to loading them into planting bags during the actual planting (Carlson 1991).

Planting techniques

Once the seedlings reach the planting site, they should be promptly planted. If bad weather or changes in planting schedules do occur, temporary storage in a cold storage (33.8 to 37.4 °F; 1 to 3 °C) facility is recommended. If no cold storage is available, seedlings should be stored in a cool, shaded spot with a water supply and good surface drainage (Carlson 1991). Bare-root seedlings should never be stored in the open at ambient temperatures.

Hallman (1993) describes several types of planting equipment to be used when planting both bare-root and containerized seedlings. Of utmost importance is the reestablishment of the root-soil interface. If hand tools are used to plant seedlings, care must be taken to excavate and backfill a hole or trench that will not have air pockets, or glazed surfaces that roots cannot penetrate (Ronco 1972). If power augers are used to excavate holes, the auger operator must work closely with the planter, so that excavated soil does not dry out before the seedlings are planted. Ronco (1972) also recommends hand scalping an area 18 inches (50 cm) wide around each planting spot to remove competing vegetation, providing a clean area to pile excavated soil, and removing debris that might fall into the hole and prevent the roots from making intimate contact with the soil.

Heidmann (1988) provides several guidelines for planting ponderosa pine seedlings. First, select a planting spot carefully and take advantage of favorable soil conditions. Second, scrape away surface litter and dry soil before making the hole. Third, the hole should be deeper than the length of the seedling's roots. When planting the seedling in the hole, care should be taken to maintain a natural vertical position and prevent "J" or "L" rooting (Hallman 1993). Fourth, once the seedling

is in the hole, fill it loosely with soil. Fifth, pull the seedling up toward the soil surface so that the root collar is almost level with the soil surface and pack the soil around the seedling with the heel of your foot. Containerized seedlings should be planted deep enough to cover the root plug to avoid desiccation. Replacement of forest litter around the seedling can help retain moisture (Hallman 1993).

Hand planting seedlings can be labor intensive, but usually is the only option for steep slopes, shallow soils, and rough terrain. Because the process of hand planting usually takes longer to get the seedling from the onsite storage facility into the ground, considerable care must be taken to maintain root moisture. If bare root seedlings are being planted, make sure to dip the roots in water as soon as they are removed from the cold storage and placed in the planting bag to be carried around the site. Make sure the planting bag has some form of wet sawdust, peat moss, or burlap to ensure that roots stay moist. Planters should only carry as many seedlings as can be planted in 2 hours to ensure that roots stay moist. Furthermore, the planters should not expose the seedlings to wind or sun for more than 5 minutes (preferably less than 1 minute) because the roots will desiccate. When separating the seedlings, the planters should be quick but careful and attempt to reduce breakage of the roots (Long 1991).

Mycorrhizal inoculation of seedlings

Trees form symbiotic associations between their roots and soil fungi called mycorrhizae. Mycorrhizae benefit seedlings and mature trees because they increase the surface area of the roots, thus increasing the absorption of water and nutrients. Castellano and Molina (1989) identified several types of ectomycorrhizal fungi that associate with ponderosa pine including: *Hebeloma crustuliniforme*, *Laccaria laccata*, *Thelephora* spp., *Rhizopogon fuscorubens*, *Rhizopogon subgelatinosus*, *Rhizopogon ochraceorubens*, and *Rhizopogon evadens*. Riffle and Tinus (1982) reported associations with *Rhizopogon roseolus* and *Suillus granulatus* in ponderosa pines grown in North Dakota. A common mycorrhizal fungus used in artificial inoculation of ponderosa pines in nursery operations is *Pisolithus tinctorius* (Castellano and Molina 1989; France and Cline 1987).

Mycorrhizae can be used to increase the survival and growth of outplanted seedlings. Seedlings with ectomycorrhizae on soils of low fertility typically have a higher mineral nutrient content per unit seedling dry weight than nonmycorrhizal seedlings (Kropp and Langlois 1990). Uptake of nitrogen was significantly higher in spruce inoculated with ectomycorrhizal fungus than in noninoculated plants (Alexander and Fairley

1986). Most importantly, ectomycorrhizal seedlings are usually more resistant to water stress (Kropp and Langlois 1990). Ectomycorrhizae also produce plant growth regulators and other secondary metabolites that aid in the expansion of the root surface area through increases in branching and rootlet diameter (Kropp and Langlois 1990). Ectomycorrhizal fungi are also known to protect the host tree from root decay by species of *Rhizoctonia*, *Fusarium*, *Pythium*, and *Phytophthora* (Kropp and Langlois 1990).

Several factors influence the formation of ectomycorrhizae in nurseries (Kropp and Langlois 1990; Marx and others 1989). For instance, *Pisolithus tinctorius* grows best at soil pH between 4.5 and 5.5, but is inhibited at pH greater than 6.0 (Marx and Cordell 1988). Because a large pathogen population can inhibit ectomycorrhizal development, soil fumigation of bare-root facilities before sowing has been recommended to reduce pathogens (Castellano and Molina 1989; Kropp and Langlois 1990). Pesticides such as triamephon, hexazone, and trichlopyr can inhibit ectomycorrhizal development (Kropp and Langlois 1990). In addition, fertilizers with high concentrations of phosphorus and nitrogen decrease the formation of ectomycorrhizae (Castellano and Molina 1989; Cornett 1982; Kropp and Langlois 1990).

Increased survival and growth of mycorrhizal seedlings—Inoculation of seedlings with mycorrhizae during the production of bare-root or containerized planting stock can help improve the quality of the seedling root system and therefore the survival and growth after outplanting (Cornett 1982; Heidmann and Cornett 1986; Kropp and Langlois 1990; Riffle and Tinus 1982). Inoculated seedlings have a net advantage over noninoculated seedlings in soils with low fertility, but are often equal in size or smaller than noninoculated seedlings in soils with high fertility (Kropp and Langlois 1990). Although seedlings in nurseries with highly fertile soils can grow rapidly without mycorrhizal inoculations, when these seedlings are outplanted in soil less fertile than the nursery soil, they will lack the benefit of the mycorrhizal association. Outplanting stress can be lessened with mycorrhizal inoculations because the fungus can aid in establishing root contact with the soil.

Baer and Otta (1981) reported an 18 percent increase in survival for bare-root ponderosa pine seedlings inoculated with *Pisolithus tinctorius* outplanted on prairie soil after two growing seasons. Riffle and Tinus (1982) investigated the influence of six different ectomycorrhizal fungi on survival and growth of ponderosa pine seedlings planted on a North Dakota site with nonmycorrhizal prairie soil. After five growing

seasons, the authors reported greater survival for the seedlings inoculated with *Rhizopogon roseolus* (62 percent), *Suillus granulatus* (56 percent), *Pisolithus tinctorius* (52 percent), *Cenococcum geophilum* (53 percent), and pine duff (75 percent). Seedlings inoculated with *Thelephora terrestris* did not benefit from the inoculation and had similar survival (30 percent) and growth as noninoculated seedlings. The pine duff inoculant was obtained from the A1 horizon of a 38-year old ponderosa pine plantation in North Dakota near where the experimental seedlings were planted. Seedlings inoculated with the pine duff had 1058 percent greater biomass than noninoculated control seedlings. The authors suggest the greater success was probably a result of a mixture of native ectomycorrhizal fungi, but do not recommend using pine duff for inoculation because of the risk of transmitting soil-borne plant pathogens.

Artificial mycorrhizal inoculation—Soil, spores, and vegetative mycelium are the three primary sources for artificial inoculation used in the nursery industry (Castellano and Molina 1989). Marx and Cordell (1988) suggest that over 50 percent ectomycorrhizal formation on the roots may be needed to guarantee a positive response to inoculation. The area in which nursery seedlings will be outplanted will most likely contain native mycorrhizae if it is or was recently forested. One method of inoculating nursery seedlings is to take soil from the potential reforested area and incorporate it into the seedbed of a bare-root operation or into the growth media for container seedlings. In bare-root seedling operations, the inoculum soil and seeds can be applied together in the nursery soil (Cordell and others 1987). Typically, up to 10 percent by volume of soil inoculum is incorporated into the top 4 inches (10 cm) of the seedbed before sowing (Castellano and Molina 1989). The use of soil inoculum can have several disadvantages, including the requirement of large quantities of soil, introduction of pathogens, weed seeds, and rhizomes into the nursery soil, and the variation of inoculum quality (Castellano and Molina 1989). The advantage of inoculating with the endemic mycorrhizal fungi is appealing, although some studies have shown that the native mycorrhizae usually replace the nursery inoculant within a year of outplanting (Castellano and Molina 1989).

Spore inoculum is a popular method for both bare-root and containerized seedling operations due to its low cost and ease of dispersion (Kropp and Langlois 1990). Spores can be added to irrigations systems, fertilizer spreaders, or applied as a seed coating before sowing (Castellano and Molina 1989; Kropp and Langlois 1990). One disadvantage of this method is

that spore inoculum requires large quantities of spores and is less effective than solid inoculum. However, *Pisolithus tinctorius* and species of *Rhizopogon*, both of which have been shown to associate with ponderosa pine seedlings (Alvarez and Linderman 1983; Castellano and Molina 1989; McDonald and others 1984b; Riffle and Tinus 1982), produce large quantities of spores (Kropp and Langlois 1990).

Vegetative mycelium inoculum is usually grown in culture on a special medium (Castellano and Molina 1989). The inoculum is placed in a peat-vermiculite carrier and mixed into container growing media. Vegetative inoculation is expensive and labor intensive. In addition, this type of inoculum has not shown better survival or growth in seedlings, so it is not recommended as an option.

Management of Associated Species

White Spruce

The silvical and ecological characteristics of white spruce in the Black Hills are similar to those of Engelmann spruce (*Picea engelmannii*) as far as seed size, regeneration requirements, shade tolerance, and so forth. However, there are some important ecological relations to consider when managing white spruce in the Black Hills. First, white spruce does not occur in association with any true firs (*Abies* spp.) in the Black Hills. Instead, white spruce grows in association with ponderosa pine, aspen, and occasionally, paper birch and lodgepole pine. These ecologic associations have important management implications for white spruce. Because a shade-tolerant true fir cohort is missing from these stands, managers can expect somewhat different regeneration responses when cutting white spruce in the Black Hills.

White spruce is the most shade-tolerant species in the mixed forests of the Black Hills, and therefore is going to be the species that regenerates best under overstory shade. It is also light seeded and can be expected to disperse seed farther into openings and through stands than ponderosa pine. White spruce seedbed and germination requirements are similar to those of ponderosa pine. Presence of a mineral soil seedbed is critical, especially when attempting to naturally regenerate white spruce in vegetation associations that have sod-forming grasses in the understory. It is important to ensure that harvest activities in these cases create sufficient soil disturbance to provide scarified seedbeds. Otherwise, very little of the seed can be expected

to germinate. Where overstory canopies are open and allow more light to reach the forest floor, less shade-tolerant species such as ponderosa pine, lodgepole pine, and aspen can be expected to regenerate more abundantly than white spruce. Aspen especially will sprout prolifically if overstory shade is removed in mixed spruce-aspen stands.

White spruce theoretically can be managed under both even-aged and uneven-aged silvicultural systems. White spruce is a shallow-rooted species with large crown volumes, and can be expected to be more susceptible to windthrow than ponderosa pine. Care must be taken when harvesting spruce stands to avoid windthrow, especially in forests with single-storied canopies.

Marking guides presented in Alexander (1987b) should be followed when marking white spruce for harvesting. As with ponderosa pine, white spruce should normally be regenerated naturally. In cases where artificial reforestation is necessary, however, white spruce seedlings must be planted so they are protected from too much sunlight. Although elevations in the Black Hills are considerably less than those of spruce forests in the Rocky Mountains, similar measures must be taken in handling and planting spruce stock. Ronco's (1972) planting guidelines for Engelmann spruce are appropriate when planting white spruce in the Black Hills.

White spruce seedlings can be expected to establish and grow somewhat more slowly than ponderosa pine seedlings in the Black Hills. Nevertheless, intermediate treatments will probably be needed to maintain desirable levels of growing stock for multiple resource uses. As with ponderosa pine, precommercial thinning will probably be the most cost effective way of removing excess growing stock, while minimizing slash accumulations and adverse impacts to wildlife habitat and forage resources.

Because spruce grows at higher elevations in the Black Hills and predominantly on northern slopes, its precipitation regime is more driven by snow, similar to conditions in the Central Rocky Mountains. Watershed management implications for the spruce type can therefore be expected to be quite similar to those reported for Engelmann spruce in the Central Rockies (Shepperd and others 1992).

Aspen

The aspen/paper birch forest type currently covers about 50,000 acres in the Black Hills (USDA Forest Service 1996a). Aspen (*Populus tremuloides*) occurs in small isolated clones in the Black Hills between

elevations of 4,000 and 7,000 feet (1,200 to 2,100 m), predominantly on north-facing slopes with favorable moisture conditions (Severson and Thilenius 1976). Most aspen stands occur in the northern half of the Black Hills in areas receiving greater than 18 inches (>46 cm) of precipitation per year (Severson and Thilenius 1976). Isolated aspen clones are often surrounded by ponderosa pine stands (fig. 50) and provide nesting and forage habitat for several bird, big game, and livestock animals. In addition, aspen plays an important role in riparian ecosystems by providing dam building material for beavers (*Castor canadensis*) (Parrish and others 1996).

Factors that could affect the continued existence of aspen in the Black Hills include: fire exclusion during most of the 20th century, competition from more shade-tolerant conifers, lack of younger age classes in existing clones, browsing and barking by animals, susceptibility to diseases, and global climate change. These factors, along with the lack of extensive mixed conifer/aspen forests, indicate that growing conditions for aspen in the Black Hills are marginal, at best. Therefore, special actions may be necessary to expand and conserve existing aspen clones.

Retaining aspen in Black Hills landscapes will require intensive management. Because aspen regenerates primarily by root suckering, opportunities to initiate new aspen stands (or replace clones that are lost) are limited. Planting aspen is not recommended. If attempted, planting would require a moist site and fencing to protect aspen seedlings from browsing animals. Even with fencing, normal mortality rates of natural aspen suckers (Shepperd 1993) suggest that very few planted seedlings would survive. Therefore, many more seedlings would have to be planted than would be desired in an established aspen stand. As with all planting efforts, seed or vegetatively reproduced planting stock should be collected from local sources.

Recommended techniques for regenerating or expanding existing clones include removal of competing conifers, protection of existing new suckers, and stimulation of new suckers (Shepperd 2001). Removing conifers growing in and around an aspen clone can reduce competition for moisture and light, which will favor the aspen. Aspen roots extend a considerable distance away from existing stems. Clearing surrounding conifer forest back 1 to 1.5 tree heights away from a declining aspen clone will allow new suckers to establish and expand the area occupied by the clone.

Given the high deer and elk populations and extensive livestock grazing throughout the Black Hills,



Figure 50: An aspen stand adjacent to a ponderosa pine stand.

protection of new sprouts from browsing will be essential in most situations. Fencing aspen stands to exclude animals has proven effective in intensively browsed areas (Rolf 2001), but must be retained until aspen stems reach at least 1.5 inches (3.8 cm) in diameter at breast height (Shepperd 2001). Retaining logging slash on the ground has been found to be effective in deterring browsing damage in the Black Hills (Rumble and others 1996), but Shepperd (1996) found that felling all aspen and leaving them onsite was detrimental to suckering. Some aspen genotypes are more desirable to browsing animals, so some clones may be damaged more severely than others (Shepperd and Fairweather 1994).

If removal of competition and protection do not result in new sprouting, it may be necessary to stimulate suckering. Three things are necessary to initiate and optimize sucker growth in aspen: interrupting the flow of auxin to roots (Shier 1973), providing full sunlight to the forest floor, and achieving soil temperatures of 54 to 68 °F (12 to 20 °C) (Hungerford 1988; Landhausser and Lieffers 1998). Cutting mature aspen stems can interrupt the flow of auxin to roots and (if sufficient numbers of trees are removed), provide sunlight and warm the soil. Clearfelling is the best

means of harvesting aspen to meet these requirements and avoid damage to residual stems. However, other techniques can be effective to stimulate suckering along the edges of small isolated aspen clones without killing all of the existing stems. Ripping along the edge of the clone with a tractor or plow can sever lateral roots and initiate suckering (Shepperd 2001). Similarly, a light prescribed fire can initiate a new cohort of suckers in an aspen clone without killing all of the existing stems. Suckers will typically be densest around the edges of a clone where tree mortality from the fire is highest and where soil temperatures are raised because of blackened soil. To maximize the suckering response, treatments to stimulate aspen suckering should be done during the dormant season. This minimizes the time roots have to exist on stored carbohydrate reserves prior to the next spring when suckering will begin (Shepperd 2001).

A rule of thumb to assess the need to treat aspen clones in the Black Hills would be to intervene where:

- Clones are deteriorating as indicated by numerous dead or diseased overstory stems, multiple aspen logs on the ground, and lack of younger age classes (as indicated by lack of multiple canopy layers and stem sizes).

- Clones have numerous conifers in the overstory canopy and adjoining the edge of the clone.
- Clones are attempting to sucker, but the suckers are being consumed by animals (suckers have a “hedged” appearance with multiple leaders, none of which have grown above the reach of animals).

Multistoried aspen clones, those with intact overstory canopies, and isolated clones with a round globular shape with green branches near the ground on the exterior edges of the clone are probably in good shape and not in need of treatment.

Bur Oak

Bur oak (*Quercus macrocarpa*) is a widely distributed eastern hardwood that achieves the westernmost limit of its distribution in the Black Hills (Little 1971). It occurs in the Black Hills either as an understory species associated with ponderosa pine, as a dominant shrub in the northern Black Hills, or as emergent individuals in ravines and riparian areas along the edges of the Black Hills (Sieg 1991). Sieg found that bur oak in the Black Hills could be associated with soil iron levels, slope, and sedge (*Carex* spp.) cover.

The age class distribution of bur oak varies from site to site in the Black Hills (Sieg 1991), but a bimodal age distribution with peaks in the 1900s and 1940s is typical. Bur oak is relatively intolerant of shade, and therefore may be displaced by more shade-tolerant associates (Sieg 1991). Fire exclusion, livestock grazing, invading exotic competitors, and increasing deer and elk populations all have probably influenced the distribution and abundance of bur oak in the Black Hills. Oak may benefit from prescribed fire, but one attempt to improve oak regeneration with a prescribed burn proved inconclusive (Sieg 1991). Herbivory and mast consumption by animals can also affect the ability of oak to persist in the Black Hills.

Where oak is associated with ponderosa pine, management to favor oak may include removal of competing pines or other vegetation, regulation or exclusion of grazing, and use of prescribed fire to improve seedbeds and stimulate vegetative sprouting. Where management goals are to favor pine, frequent prescribed fires and increased fall grazing may be desirable to reduce oak’s presence on the site so pine seedlings can ultimately be established.

Watershed Management

Water yield is a concern for communities in and around the Black Hills (USDA Forest Service 1996a).

The water provided by streams that drain out of the Black Hills is an important resource utilized for water supplies, irrigation, and recreation of the surrounding area (Addison 1991). Forested landscapes intercept rainfall and lower the quantity of water that reaches the stream. The decrease depends upon the density or coverage of the forest. Other factors that influence water yield and streamflow include: climate, geology, slope and soil conditions, channel conditions, vegetation, riparian area conditions, and instream structures (Orr 1975).

Historic accounts by the Custer and Dodge Expeditions of 1874 and 1875 noted that streamflows were more often perennial, riparian zones were more abundant and more lush, and wet meadows were more plentiful in low-gradient valley bottoms than conditions today. Subsequent postsettlement timber harvesting, mining, livestock grazing, and beaver trapping changed the water yield and streamflow of the Black Hills. These changes damaged the riparian zones and led to an increase in peak flows and the intensity of flash floods, and a decrease in the perennial flow of some streams (Parrish and others 1996). Since the creation of the Black Hills National Forest and implementation of fire suppression in the early 1900s, forests in the Black Hills have become dense and water yields have decreased.

Black Hills Hydrology

Precipitation follows several potential pathways as it travels through the ecosystem. First, precipitation can be intercepted and can evaporate back into the atmosphere before it reaches the ground. Second, it can be absorbed into the soil and later taken up by the roots of a plant and transpired back into the atmosphere. Third, as the soil becomes saturated with moisture, subsurface flow can transport water very slowly to a stream. Or, once the soil becomes completely saturated and no more can soak into the ground, water runs into streams as overland flow.

The type of rock that precipitation falls on influences the pathway it will take. For instance, the central crystalline core of the Black Hills is largely impervious to moisture, and water that falls upon it runs off as surface flow, or remains in shallow soils near the surface. In contrast, the sedimentary rock that surrounds the central core is porous and plays a significant role in the ground-water hydrology of the area. Water traveling over porous rock types such as limestone and sandstone can disappear into sinkholes and underground streams. The water goes underground to recharge the numerous aquifers in the Black Hills. At times, the

underground water reappears as springs elsewhere (Orr 1975). This underground movement of water has major implications for management activities intended to increase water yield. The only way to increase water yield to areas downstream would be to increase the flow so much as to exceed the amount that can physically be transported underground. Orr (1975) suggests that management of relatively small amounts of water at many locations upstream from the “loss” zone could increase water yields.

Addison (1991) developed monthly rainfall/runoff relationships for 11 Black Hills drainage basins: Fall River at Hot Springs, Beaver Creek near Buffalo Gap, Battle Creek near Keystone, Grace Coolidge Creek near Game Lodge, Battle Creek at Hermosa, Spring Creek near Hermosa, Castle Creek above Deerfield Reservoir, Rapid Creek above Pactola Reservoir, Boxelder Creek near Nemo, Elk Creek near Rapid City, and Spearfish Creek at Spearfish. In most basins, the previous month’s discharge was shown to be the best predictive variable to determine rainfall/runoff relationships. Current month’s precipitation was most important during the months of April, May, and June.

Timber/Water Yield Relationships

The influence of forested landscapes on water yield has been investigated in the Black Hills in several studies (Orr 1968, 1972, 1975; Orr and VanderHeide 1973; Smith 1983). In general, water discharge increases after timber harvest or wildfire due to the decreased interception by the vegetation and a decrease in the evapotranspiration rate. Anderson (1980) reported an average increase of 2.24 inches (5.7 cm) after 25 percent of the timber volume was cut.

Orr (1968) investigated the soil-moisture patterns and trends after thinning and patch clearcutting in a dense 70-year-old, second-growth pine stand on limestone soil. Basal area was 193 ft² per acre (44.3 m² per hectare) for the control plot, 80 ft² per acre (18.4 m² per hectare) for the thinned plot, and 0 ft² per acre (0 m² per hectare) for the clearcut. Evapotranspiration was significantly lower on the thinned plot than on the densely stocked control plot, indicating more moisture was held in soil storage on the thinned plot. This was especially important during years of drought. During drought years, evapotranspiration was greater on the thinned plots, but there was more soil moisture available as well as higher seepage potential. The clearcut treatment had the highest water seepage potential since evapotranspiration from trees was eliminated. However, this increase in moisture availability cannot be

sustained because soil moisture storage is lowered as the new forest is established.

Orr and VanderHeide (1973) described the water yield characteristics of three small watersheds in the northeast Black Hills near Sturgis. Average basal areas were 120 ft² per acre (27.6 m² per hectare), 130 ft² per acre (29.8 m² per hectare), and 166 ft² per acre (38.1 m² per hectare) for watersheds 1, 2, and 3, respectively. Annual water yields over a period of 6 years were 25, 27, and 23 percent of annual precipitation for watersheds 1, 2, and 3, respectively, which marginally supports the plot study results discussed in Anderson (1980) and Orr (1968).

Precipitation patterns among the different regions where ponderosa pine forests can be found differ significantly. For instance, in a study of climatic regimes of four ponderosa pine regions, both northern Arizona and the Black Hills were found to have approximately the same amount of annual precipitation (Pearson 1951). In northern Arizona, however, the majority of precipitation falls in the winter months as snow, with some rain during July and August due to monsoon weather patterns. Baker (1986) reported that, in the Beaver Creek watershed of Arizona, over 97 percent of the annual water yield is generated in the winter months because of snowmelt, and that by early May, the streambeds are dry. In the Black Hills, the majority of precipitation falls between April and July, with 12.4 percent of the precipitation from November to February (Driscoll and others 2000). Therefore, results of water yield studies from Beaver Creek should not be used for Black Hills watershed management. Studies from the Colorado Front Range ponderosa pine zone might be more appropriate since the precipitation patterns are somewhat similar to those in the Black Hills (Gary 1975).

Seasonal water yields are influenced by the amount and timing of spring and early summer precipitation. In the Black Hills, general thaw is common from the middle to the end of March. By the end of March, flows start to increase in the streams. In April, a sharp increase in flow is attributed to the combination of precipitation and melting of accumulated snow. Flows reach their maximum during May or June. Although precipitation is highest in June, streamflow does not increase proportionally due to the increased evapotranspiration of the forested stands. Most of the July to September precipitation is lost to evapotranspiration, and little contributes to streamflow (Orr and VanderHeide 1973).

In the Black Hills, snowmelt contributes primarily to soil recharge and only affects streamflow indirectly (Orr 1975). Creating openings in the forest canopy

was thought to increase water yield by providing aerodynamics that favored snow accumulation (Baker 1986). Cutting in patches, strips, or blocks had been shown to increase snow accumulation and streamflow in the Colorado Front Range and Rocky Mountain subalpine zone (Gary 1975; Leaf 1975). However, later work at Fraser Experimental Forest has shown that the lack of interception losses back to the atmosphere when trees are cut is the primary contributor to increased streamflow in snow-dominated systems (Troendle and King 1987). Removal of trees in any fashion will therefore increase runoff to some degree (Shepperd and others 1992).

If openings are created, evaporation of snow in them will be twice as high as the rate found in the understory of a forest (Gary 1975). The snow in openings will also melt sooner in the spring while transpiration is lower, and more of the runoff will reach the stream. Gary suggests that the pattern of the cuts determines whether runoff will be increased. For instance, large clearcut blocks or selective cuts of individual trees will increase water yields far less than if the same volume was harvested in a system of small, dispersed forest openings.

Furthermore, snow will be scoured from sites exposed to wind, and they will also have greater evaporation losses. Slope aspect also influences water yield (Cline and others 1977). Clearcut north slopes in Idaho were reported to show a substantial gain in water yield versus south slopes. The difference in water yield between slope aspects was a result of quicker establishment of vegetation on the south-facing slopes (Cline and others 1977).

In conclusion, any silvicultural activity or natural disturbance that removes ponderosa pine trees from a forest will potentially increase water yield. Conversely, the introduction of new trees, or growth of existing trees, will potentially decrease water yield. The increase or decrease is proportional to the amount of stocking removed, but usually 25 to 30 percent of the growing stock in a forested watershed must be removed before there is a measurable increase in water in the stream leaving the watershed.

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Appendix A: Common and scientific names of species listed

Common name	Scientific name
Alders	<i>Alnus</i> spp.
American elm	<i>Ulmus americana</i>
American robin	<i>Turdus migratorius</i>
American sweetvetch	<i>Hedysarum alpinum</i>
Armillaria root disease	<i>Armillaria ostoyae</i>
Arrowleaf balsamroot	<i>Balsamorhiza sagittata</i>
Aspen	<i>Populus tremoides</i>
Baneberry	<i>Actaea rubra</i>
Beaked hazelnut	<i>Corylus cornuta</i>
Bearberry	<i>Arctostaphylos uva-ursi</i>
Beaver	<i>Castor canadensis</i>
Black-backed woodpecker	<i>Picoides arcticus</i>
Blackbellied clerid	<i>Enoclerus lecontei</i>
Black-capped chickadee	<i>Parus atricapillus</i>
Black-headed grosbeak	<i>Pheucticus melanocephalus</i>
Blue staining fungus	<i>Ceratocystis montia</i>
Bluebell = Harebell	<i>Campanula rotundifolia</i>
Bracken fern	<i>Pteridium aquilinum</i>
Brown creeper	<i>Certhia americana</i>
Brown-headed cowbird	<i>Molothrus ater</i>
Bur oak	<i>Quercus macrocarpa</i>
Bushy-tailed woodrat	<i>Neotoma cineria</i>
Canada violet	<i>Viola canadensis</i> = <i>Viola rugulosa</i>
Chipping sparrow	<i>Spizella passerina</i>
Chokecherry	<i>Prunus virginiana</i>
Common juniper	<i>Juniperus communis</i>
Cream peavine = Yellow vetchling	<i>Lathyrus ochroleucus</i>
Currants	<i>Ribes</i> spp.
Dark-eyed junco	<i>Junco hyemalis</i>
Deer mice	<i>Peromyscus maniculatus</i>
Diploida tip blight	<i>Sphaeropsis sapinea</i>
Downy woodpecker	<i>Picoides pubescens</i>
Dryspike sedge	<i>Carex foenea</i> = <i>Carex siccata</i>
Dusky flycatcher	<i>Empidonax oberholseri</i>
Dwarf mistletoe	<i>Arceuthobium campylopodum</i>
Engelmann spruce	<i>Picea engelmannii</i>
Evening grosbeak	<i>Coccothraustes vespertinus</i>
Fairybell	<i>Disporum trachycarpum</i>
Firs	<i>Abies</i> spp.
Fringed sagewort	<i>Artemisia frigida</i>
Gray jay	<i>Perisoreus canadensis</i>
Green ash	<i>Fraxinus pennsylvanica</i>
Grouseberry	<i>Vaccinium scoparium</i>
Hairy woodpecker	<i>Picoides villosus</i>
Ironwood	<i>Ostrya virginiana</i>
Kentucky bluegrass	<i>Poa pratensis</i>
Leafy phlox	<i>Phlox alyssifolia</i>
Least chipmunk	<i>Tamias minimus</i>
Lewis' woodpecker	<i>Melanerpes lewis</i>
Lindley's aster	<i>Aster ciliolatus</i> = <i>Aster lindleyanus</i>
Little bluestem	<i>Andropogon scoparius</i> = <i>Schizachyrium scoparium</i>

Common name**Scientific name**

Littleseed ricegrass	<i>Oryzopsis micrantha</i>
Lodgepole pine	<i>Pinus contorta</i>
Longspur violet	<i>Viola adunca</i>
Low northern sedge	<i>Carex concinna</i>
MacGillivray's warbler	<i>Oporornis tolmiei</i>
Manitoban elk	<i>Cervus elaphus manitobensis</i>
Maryland sanicle	<i>Sanicula marilandica</i>
Merriam's turkey	<i>Meleagris gallopavo merriami</i>
Mountain bluebird	<i>Sialia currucoides</i>
Mountain cliff fern	<i>Woodsia scopulina</i>
Mountain pine beetle	<i>Dendroctonus ponderosae</i>
Mule deer	<i>Odocoileus hemionus</i>
Needle cast	<i>Elytroderma deformans</i>
Ninebark	<i>Physocarpus monogynus</i>
Northern bedstraw	<i>Galium boreale</i>
Northern flicker	<i>Colaptes auratus</i>
Northern flying squirrel	<i>Glaucomys sabrinus bangsi</i>
Northern goshawk	<i>Accipiter gentiles atricapillus</i>
Nuttall's cottontail	<i>Sylvilagus nuttallii</i>
Oregon grape	<i>Berberis repens</i>
Ovenbird	<i>Seiurus aurocapillus</i>
Paper birch	<i>Betula papyrifera</i>
Pasqueflower	<i>Anemone patens</i> = <i>Anemone nuttalliana</i> = <i>Pulsatilla patens</i>
Pine engraver	<i>Ips pini</i>
Pine siskin	<i>Carduelis pinus</i>
Pink shinleaf	<i>Pyrola asarifolia</i>
Poison ivy	<i>Toxicodendron rydbergii</i>
Ponderosa pine	<i>Pinus ponderosa</i>
Porcupine	<i>Erethizon dorsatum</i>
Poverty oatgrass	<i>Danthonia spicata</i>
Prickly rose	<i>Rosa acicularis</i>
Purple meadowrue	<i>Thalictrum dasycarpum</i>
Pussytoes	<i>Antennaria</i> spp.
Pygmy nuthatch	<i>Sitta pygmaea</i>
Red crossbill	<i>Loxia curvirostra</i>
Red raspberry	<i>Rubus idaeus</i>
Red rot	<i>Dichomitus squalens</i>
Red squirrel	<i>Tamiasciurus hudsonicus</i>
Red turpentine beetle	<i>Dendroctonus valens</i>
Redbellied clerid	<i>Enoclerus sphaeus</i>
Red-breasted nuthatch	<i>Sitta canadensis</i>
Red-naped sapsucker	<i>Sphyrapicus nuchalis</i>
Rocky Mountain elk	<i>Cervus elaphus nelsoni</i>
Rocky Mountain juniper	<i>Juniperus scopulorum</i>
Rough-leaved ricegrass	<i>Oryzopsis asperifolia</i>
Ruby-crowned kinglet	<i>Regulus calendula</i>
Ruffed grouse	<i>Bonasa umbellus</i>
Russet buffaloberry	<i>Shepherdia canadensis</i>
Sedges	<i>Carex</i> spp.
Shoestring root rot	<i>Armillaria mellea</i>
Sideoats grama	<i>Bouteloua curtipendula</i>
Silvery lupine	<i>Lupinus argenteus</i>
Slender wheatgrass	<i>Agropyron caninum</i> = <i>Elymus trachycaulus</i>
Smooth brome	<i>Bromus inermis</i>

Common name**Scientific name**

Southern pine tip moth	Rhyacionia neomexicana
Spreading dogbane	Apocynum androsaemifolium
Spurred gentian	Halenia deflexa
Star lily	Leucocrinum montanum
Starry false Solomon's seal	Smilacina stellata = Maianthemum stellatum
Sumacs	Rhus spp.
Sun sedge	Carex heliophila
Swainson's thrush	Catharus ustulatus
Sweetroots	Osmorhiza spp.
Sweet-scented bedstraw	Galium triflorum
Thirteen-lined ground squirrel	Spermophilus tridecemlineatus
Three-toed woodpecker	Picoides tridactylus
Timber oatgrass	Danthonia intermedia
Timothy	Phleum pratense
Townsend's solitaire	Myadestes townsendi
Trogositid beetle	Temnochila chlorodia
True mountain mahogany	Cercocarpus montanus
Twinflower	Linnaea borealis
Virginia's warbler	Vermivora virginiae
Vole	Microtus spp.
Warbling vireo	Vireo gilvus
Western gall rust	Peridermium harknessii
Western pine tip moth	Rhyacionia bushnelli
Western serviceberry	Amelanchier alnifolia
Western snowberry	Symphoricarpos occidentalis
Western tanager	Piranga ludoviciana
Western wheatgrass	Agropyron smithii
White coralberry	Symphoricarpos albus
White spruce	Picea glauca
White-breasted nuthatch	Sitta carolinensis
White-tailed deer	Odocoileus virginianus dakotensis
White-tailed jackrabbit	Lepus townsendii
Wild honeysuckles	Lonicera spp.
Wild lily-of-the-valley	Maianthemum canadense
Wild sarsaparilla	Aralia nudicaulis
Wild spirea	Spiraea betulifolia
Wild strawberry	Fragaria virginiana
Willows	Salix spp.
Woodland strawberry	Fragaria vesca
Woods rose	Rosa woodsii
Yarrow	Achillea millefolium
Yellow-rumped warbler	Dendroica coronata



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