Chapter 3

Aspen

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Introduction

Quaking aspen (*Populus tremuloides* Michx.) is the most widely distributed broadleaf tree in North American (Little 1971; Sargent 1890). Aspen forests occur from Labrador on the east coast to Alaska in the north to Mexico in the south. In its eastern range, aspen is relatively continuously distributed. In the western United States, however, it occurs on the more suitable sites on mountains and high plateaus (Jones 1985). On xeric sites, it is found primarily in riparian zones. Most western aspen occur on public lands and exist as pure clones, successional to conifer, or as small-scattered decadent groves. At least 75 percent of aspen in the western United States occurs in two states, Utah and Colorado. Almost 25 percent of Utah's forests are aspen (Mueggler 1988).

Aspen is portrayed as an excellent indicator of ecological integrity as well as landscape health (Kay 1991a,b; Woodley and Theberge 1992; Woodley 1993; Woodley and others 1993). Aspen ecosystems in the western United States yield numerous products and benefits, including forage for livestock, habitat for wildlife, water for downstream users, esthetics, sites for recreational opportunities, wood fiber, and landscape diversity (Hessl 2002). Loss and potential loss of aspen from these western landscapes is attributed primarily to the reduction of wildfire, long-term overuse by ungulates (both wildlife and domestic livestock), and successional processes. During 2006, considerable attention was given to aspen die-off. This die-off is described as the rapid death of entire clones, including the root system, with no potential for aspen survival (http://www.fs.fed.us/rmrs/research/highlights/aspen-restoration.php).

Aspen is an early successional, shade intolerant species with a short life span—usually less than 200 years in the western United States (Perala 1991). Aspen exist primarily as one of three types: 1) stable, 2) successional to conifers or seral, and 3) decadent and falling apart (Bartos 2001). Unlike other western forest trees, aspen are unique because they reproduce almost exclusively by suckering from the parent root system. Generally, a disturbance or die-back is necessary to stimulate regeneration of the aspen stands. It is speculated that these self-regenerating stands have existed for thousands of years. Unlike other tree species, if aspen stands are lost from the landscape they usually will not return through the natural process of seed dispersal.

Most aspen stands will eventually be replaced by conifers, sagebrush (*Artemisia* spp.), or possibly other shrub communities if the current conditions that have prevailed for the past 100 to 150 years (for example, lack of fire, excessive wildlife, and livestock browsing) continue. Current unpublished Forest Inventory and Analysis (FIA) data for the state of Utah show that there has been approximately a 60 percent decrease in aspen-dominated public lands since European settlement (Bartos and Campbell 1998a). This decrease is generally uniform across the entire state. When aspen lands convert to conifer dominated landscapes, substantially less water is available for streamflow, undergrowth biomass production is greatly reduced, and there is a marked decline in the diversity of plants and animals.

Development of useful management recommendations concerning the aspen, aspen/conifer, and aspen/sagebrush ecosystems requires a good understanding of the structure and function of these systems. Aspen literature for the western United States has been summarized through the early 1980s (DeByle and Winokur 1985). The published proceedings of a more recent aspen symposium held in June 2000 was a supplement to this earlier work (Shepperd and others 2001). Shepperd and others (2006) recently synthesized the existing information on the ecology and management of aspen in the Sierra Nevada of California.

The purpose of this paper is to discuss the historical and current conditions of aspen, with an emphasis on aspen in Utah. This paper also identifies clones at risk, provides a method of treatment prioritization, and offers management recommendations of aspen stands. Landscapes that need priority management are mixed-conifer/aspen, particularly where subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.) dominates, and aspen/sagebrush transition zones. Recommendations to regenerate aspen include the need for immediate, large-scale (200 to 400+ ha [500 to 1000+ acres]) treatments that mimic natural mosaic patterns.

An aspen stand typically contains numerous genetically identical stems (ramets) that propagated vegetatively from seedlings at some time in the past when conditions were favorable. Aspen stems originate from root suckers, some of which still may be interconnected via a shared root system. Shepperd and Smith (1993) reported that aspen stems establish independent root systems by approximately 25 years of age, with few stems still connected to the original root system. Barnes (1975) speculated that these self-regenerating clones have existed for thousands of years, being perpetuated over time by disturbance – primarily fire. Some clones occupy large areas. One such clone, referred to as Pando (Latin for *I spread*), occurs on the Fishlake National Forest in central Utah (Mitton and Grant 1996; Grant 1993). The authors claim that this 43 ha (106 acres) clone is the world's largest living organism. Kemperman and Barnes (1976) originally identified the Pando clone and speculated that clones as large as 81 ha (200 acres) may exist.

Successful sexual reproduction of western aspen is extremely rare (Mitton and Grant 1996). Jelinski and Cheliak (1992) describe a "window of opportunity" that may allow seedling establishment at intervals of 200 to 400 years. Abnormally cool, moist years may allow sexual reproduction of aspen seedlings (Elliott and Baker 2004). There are some documented cases of successful aspen seedling establishment in the west. Romme and others (2005) documented the establishment and persistence of sexually produced aspen seedlings after the 1988 fires in Yellowstone National Park. Elliott and Baker (2004) hypothesized that expanding aspen clones at tree line in the San Juan Mountains of Colorado were established by seed. However, unlike other western tree species, aspen seedling establishment is rare and undependable (Barnes 1966). Therefore, once aspen is lost from the landscape, it will seldom establish from seed in the Intermountain West. Vegetative reproduction by suckers generally requires a disturbance (fig. 1) or die-back that alters the hormonal balance of a clone (Bancroft 1989; Schier and others 1985). Frey and others (2003) recently reviewed the process of aspen sucker regeneration. Bancroft (1989) and Bartos (2001) discuss what happens when there is a disruption in the flow of two hormones (cytokinin and auxin) within an aspen tree. Basically, when the tree is killed or stressed, the flow of sucker-suppressing auxin from the crown down to the root system is disrupted, which allows a second hormone, cytokinin, to stimulate suckering.

Figure 1—One-year old aspen (*Populus tremuloides*) sprouts after the Sanford Fire, UT.

To abide by the words of Aldo Leopold: "…if we are serious about restoring or maintaining ecosystem health and ecological integrity, then we must first know what the land was like to begin with" (quote in Covington and Moore 1994). For millennia, wildfire dominated the natural functioning of the aspen ecosystem (Baker 1925; Parfit 1996; Pyne 1995). Baker (1918) and Schier (1976) describe western aspen as a fire dependent species. Jones (1985) stated "…fire is responsible for the abundance of aspen in the west and for the even-aged structure of so many stands." Conversely, reduction of fire in the Interior West is probably one of the main causes of the decline of aspen from so much of our landscapes because it permits the encroachment of shade tolerant conifers that reproduce from seed (Bartos 2001; Schier 1975).

Historical Conditions

Fire has been a major component of the vegetation landscape prior to humans arriving on the scene, and later became a critical tool for man. Pyne (1995) suggests that fire and humans co-evolved. The native inhabitants of the North American continent have long been considered natural conservationists living in harmony with the environment. However, it is now becoming clear that the American Indians had a profound influence on the landscape, particularly with respect to the use of fire. Ongoing fire history studies in southern Utah indicate that during the approximately 400 years prior to settlement, fire-free intervals varied from 20 to 60 years. These fire free intervals were longer with increases in elevation (Chappell 1997). When European man arrived in the western United States around 1850, they found a mosaic of vegetation profoundly influenced by Native Americans over a period of 12,000+ years (Kay 1997). These "pristine" landscapes were in a continual state of flux as a result of both natural and human caused burning.

Early settlers further impacted the land through removal of fine fuels by grazing domestic livestock, thus limiting the spread of fire. In the early part of the 20th century, land managers instigated a vigorous campaign of fire control. These two practices (grazing and fire control) have had a profound effect on the vegetation we now see on western landscapes. As a result of this lack of fire, aspen stands on the landscape have been greatly reduced. Because western aspen reproduces primarily by suckering, the elimination of fire combined with excessive browsing of the asexual reproduction by ungulates has caused many aspen dominated sites to convert to other vegetation types. Barnes (1975) speculates that many of the existing clones have been present since the last ice age, while Knight (1994) states some clones may have persisted since Pleistocene times. Therefore, clones that have persisted for thousands, if not hundreds of thousands, of years are in danger of being eliminated in as little as a couple of centuries.

Brown and Simmerman (1986) detailed the difficulty of burning pure aspen, while DeByle and others (1987) describe the aspen type as "asbestos" in nature and discuss its use as natural firebreaks. Earlier, Fechner and Barrows (1976) discussed the use of pure aspen as firebreaks. They reported that fire ignition rates in quaking aspen stands are less than half those for all other cover types in Colorado. Crown fires often drop to the ground when they reach pure aspen stands and therefore, fire spreads only a short distance into the stands. As early as 1925, Baker (1925) cited grazing as the primary reason many aspen stands no longer burn. He stated that intense grazing pressure reduces the undergrowth in aspen clones such that fire spread is inhibited. Fires before grazing became heavy in the mountains of Utah were as frequent as 7 to 10 years, but stopped about the time grazing began (Baker 1925). DeByle and others (1987) gave four possible reasons why fires are less frequent in aspen today than historically: 1) effective direct control of wildfires, 2) grazing, 3) removal of American Indians from historic range, and 4) succession. This phenomenon of aspen no longer burning needs to be further evaluated, especially now that there is so much concern with respect to the wildland/urban interface.

Current Conditions

Aspen is a disturbance-dependent species that flourished when these western lands burned periodically. With suppression of this natural force, many of these lands have converted, or are in the process of converting, to other vegetation types, such as conifer (for example, subalpine fir) or sagebrush. Three of the most critical products being lost from the aspen system as a result of this conversion are water, undergrowth vegetation, and biodiversity.

DeByle and Winokur (1985) reported that there were more than 2.8 million ha (7 million acres) of aspen in the Interior West and, Mueggler and Campbell (1986) reported over 648,000 ha (1.6 million acres) of aspen-dominated forests in Utah. The Rocky Mountain Research Station's FIA Project has compiled data representing current and historical acreage of aspen in the Interior West. The historical data result from summing all acres that currently contain at least one aspen, either living or dead. This assumes that this acreage is, or once was, occupied by aspen. For the state of Utah, FIA data show that there has been almost a 60 percent decrease in aspen dominated lands since the arrival of European man (table 1). This loss is fairly uniform across the entire state as seen from the data for individual forests. Similar trends are seen elsewhere throughout the western United States. Analysis of FIA plots in Idaho, Colorado, and Wyoming show regional aspen cover loss (Rogers 2002). Rogers and others (1998) report that 31 percent of the FIA plots in Colorado aspen forest types are "unstable" and are possibly in transition to other types. However, Manier and Laven (2002) and Kaye and others (2003) found no evidence of aspen decline in some areas of Colorado. Lachowski and others (1996) and Wirth and others (1996) used remote sensing and geographic information systems (GIS) to evaluate the loss of aspen in the Gravelly Mountains in southwestern Montana. They found that aspen decreased by approximately 47 percent over a 45-year period (1947 to 1992) and attributed this decrease to succession to conifer species. Brown (1995), in a review article, states that the deterioration of aspen in Oregon and Washington appears similar to the decline of aspen in other parts of the west. In the South Warner mountains in northern California, Di Orio and others (2005) found a 24 percent decline in aspen cover between 1946 and 1994 and a decrease in the mean size of aspen stands.

Ungulates

Ungulates have a major impact on aspen regeneration (Bartos and Mueggler 1981; Bartos and others 1994; Kay 1985, 1990, 1995; Kay and others 1994; McCain and others 2003; U.S. Department of Agriculture 1994). Excessive browsing on aspen suckers by wildlife (elk [*Cervis canadensis*], deer [*Odocoileus hemionus*], or moose [*Alces alces*]) or by livestock (cattle or sheep) can suppress or eliminate regeneration in aspen stands. Excessive livestock use of young aspen suckers can threaten stand maintenance in climax aspen communities, especially if the use is in August or September (McCartney 1993). Early in the past century, Sampson (1919) noted that cattle were not a detriment to the establishment of aspen suckers "unless the range is stocked with cattle beyond its natural carrying capacity." He further noted that sheep would eliminate all aspen reproduction where sites were grazed 3 consecutive years. Smith and others (1972) concluded that proper livestock management is essential to regeneration of aspen following removal of older stands. A recent model-based assessment of aspen and elk herbivory in Rocky Mountain National Park suggests that a reduction of elk populations up to 80 percent is necessary to allow regeneration of 90 percent of the park's declining aspen stands (Weisberg and Coughenour 2003).

The Pando clone (Kemperman and Barnes 1976) mentioned earlier is a good example of the problems associated with the decline of aspen. Managers observed that this single-story clone was not regenerating successfully. A cutting treatment was imposed on a portion of the

Table 1—Current and historical acres of aspen found in Utah (Unpublished data provided by the USDA Forest Service, Rocky Mountain Research Station's Forest Inventory and Analysis Project.).

	Current aspen		Historical aspen		Decline
National Forest	**hectares**	**(acres)**	**hectares**	**(acres)**	**(percent)**
Ashley	41,036	(101,358)	130580	(322,532)	69
Uinta	70,645	(174,492)	115527	(285,351)	39
Wasatch-Cache	52,071	(128,615)	151351	(373,837)	66
Dixie	61,965	(153,053)	177213	(437,715)	65
Fishlake	57,469	(141,948)	127014	(313,724)	55
Manti-LaSal	64,318	(158,866)	136845	(338,008)	53
Southern Utah	183,752	(453,867)	441072	(1,089,447)	58
Northern Utah	163,751	(404,465)	397457	(981,720)	59
Total	347,503	(858,332)	838529	(2,071,167)	59

clone in the early 1990s. The initial treated area produced suckers; however, none of them survived. Subsequently, an additional portion was cut and fenced with a 2-m (7-ft) high woven wire fence to exclude large animals. This resulted in successful regeneration in the cut and fenced area that should be stand-replacing. Apparently, deer and cattle browsing effectively suppressed sucker reproduction in the unprotected portion of the stand. The Coconino National Forest in Arizona installed two large elk exclosures after a wildfire to facilitate aspen regeneration. Elk browsed 85 percent of aspen shoots in high severity burned areas outside the exclosures and 34 percent of the shoots on the intermediate burned areas (Bailey and Whitman 2002). This suggests elk pressure is greatest in areas where recent high severity burns produce large numbers of aspen shoots.

This potentially severe ungulate influence on successful aspen regeneration adds a critical component to management considerations in certain parts of the west (Shepperd and Fairweather 1994). Burned clones subjected to repeated browsing only hastens their demise (Bartos and others 1994; Kay 1990). Therefore, treatments to induce suckering must not be initiated before relief from excessive browsing is obtained (U.S. Department of Agriculture 1994). Kay (1997) speculates that the historic abundance of aspen in the Intermountain area suggests that these ecosystems developed with relatively low levels of ungulate pressure. Larsen and Ripple (2003, 2005) found that aspen recruitment into the overstory was significantly higher on the National Forests bordering the northern portion of Yellowstone National Park than inside the park where elk pressure is greater. They also found that overstory aspen recruitment inside the park was only occurring in areas protected from ungulate browsing, such as fenced exclosures and scree slopes. This further supports the need to control or modify browsing intensity, where necessary, if these western aspen communities are to survive by vegetative reproduction.

Insects and Disease

Quaking aspen plays host to a wide array of both insects and diseases. Hinds (1985) states that many diseases attack aspen, however, very few kill or seriously injure living trees. For example, leaf spot (*Marssonina populi*) is a common leaf disease of aspen in the west. If this disease prevails for 2 or more years, it can cause twig and branch mortality, but rarely kills mature trees (Harniss and Nelson 1984). Many of the prevalent diseases in aspen affect both regeneration and mature trees (Hinds 1985). In both cases, trees damaged and weakened by animals are easy targets for introduction of disease. Disease plays a part in thinning excessive regeneration of aspen in areas where stand replacing episodes have occurred. Hart and Hart (2001) working in western Wyoming found that larger aspen stems (>15 cm [5.9 inches]) had less mortality due to disease/animal wounds than smaller stems. This observation substantiates the fact that disease tends to reduce aspen regeneration and smaller trees. Aspen die-off, mentioned earlier, is occurring from northern Arizona to Canada and currently, is most prevalent in southern Utah and western Colorado. Some speculate that this die-off phenomenon is a result of prolonged drought conditions coupled with either opportunistic diseases and/or insects. More detailed studies are needed to define this problem in order to make management recommendations that address the issue (Aspen Summit 2006).

More than 30 insect species are listed by Furniss and Carolin (1977) as utilizing aspen. Some of these species cause minimal damage, some just attack and impact weakened or dying trees, and some may severely damage or kill otherwise healthy trees. Insect impacts coupled with other factors, such as ungulate use, can have major impacts on regeneration. Jones (1985) gives a thorough discussion of insects and their impact on western aspen.

Fire Exclusion

The 7- to 10-year fire return interval suggested by Baker (1925) indicates these lands were burning often prior to European settlement. Because of the disruption in the fire regime and natural succession, we are finding aspen that are considerably older than the normal pathological rotation age of 70 to 80 years (Personal communication, John Guyon, Pathologist, Forest Health Protection, Ogden, UT). This aging of aspen on the landscape gives rise to the potential for more catastrophic disease and insect events. Additionally, this excessive mortality will add to the fuel loads.

Ecosystem Impacts Caused by Aspen Loss

Loss of aspen from western landscapes translates to a loss of water, forage, and biodiversity, as well as other benefits (Bartos and Campbell 1998b). Harper and others (1981) suggest that water yields can be expected to decrease 5 percent as conifers replace aspen. Gifford and others (1984) concluded from modeling studies that a net water loss of 18.6 cm (7.3 inches) occurs when spruce (*Picea* spp.) replaces aspen, and a loss of 7.2 cm

(2.8 inches) of water when subalpine fir forests replace aspen. Therefore, aspen-to-conifer succession appears to have the potential to markedly reduce water yields in the western United States (Bartos and Campbell 1998b). This loss of water means that it is not available to produce undergrowth vegetation, recharge soil profiles, or increase streamflow. In dry climates, such as the Great Basin, this loss of water is substantial and should be of great concern to the public.

Aspen forests traditionally have been considered prime grazing lands in the Intermountain West. Forty to 70 percent of the undergrowth production found in association with major aspen community types in the Intermountain Region consists of palatable forage (fig. 2) (Mueggler 1988). Mueggler found that the most productive aspen communities produced as much as 3,200 kg/ha (2,900 lbs/acre) air-dry undergrowth material and averaged over 1,350 kg/ha (1,200 lbs/acre). Mueggler (1985) found when conifers comprised as little as 15 percent of the total tree basal area on the Wasatch Plateau of Utah, undergrowth production was reduced approximately 50 percent (fig. 3a). Mueggler (1988) reported an even greater reduction on the Fishlake and Dixie National Forests of southern Utah, with undergrowth production reduced two-thirds when conifers comprised as little as 15 percent of the total tree basal area (fig. 3b). Similar reductions in undergrowth forage production occur in other forest types as conifer basal area increases (Jameson 1967). Once conifer invasion approaches 50 percent of the total tree basal area in aspen stands, undergrowth production is likely to be only a small fraction of what it once was on these formerly prime grazing lands.

When we consider that more than half of our aspen dominated stands in the Interior west have converted to conifer during the past 150 years, we obviously can conclude we have experienced a large loss of forage production. If this dramatic decrease in vegetation is not considered when determining stocking rates for grazing allotments, an overgrazing problem is compounded. For example, if a 50 percent decline of aspen stands on an allotment occurred without a corresponding adjustment of allowable stocking, the remaining forage producing areas will be subjected to much heavier utilization than previously. This excessive pressure will adversely impact aspen regeneration as well as the other forage resources.

Figure 2—Aspen (*Populus tremuloides*) with productive herbaceous understory, including cowparsnip (*Heracleum lanatum*), snowberry (*Symphoricarpos oreophilus*), and numerous other forbs and grasses. In addition, subalpine fire (*Abies lasiocarpa*) and other conifers are establishing in the understory.

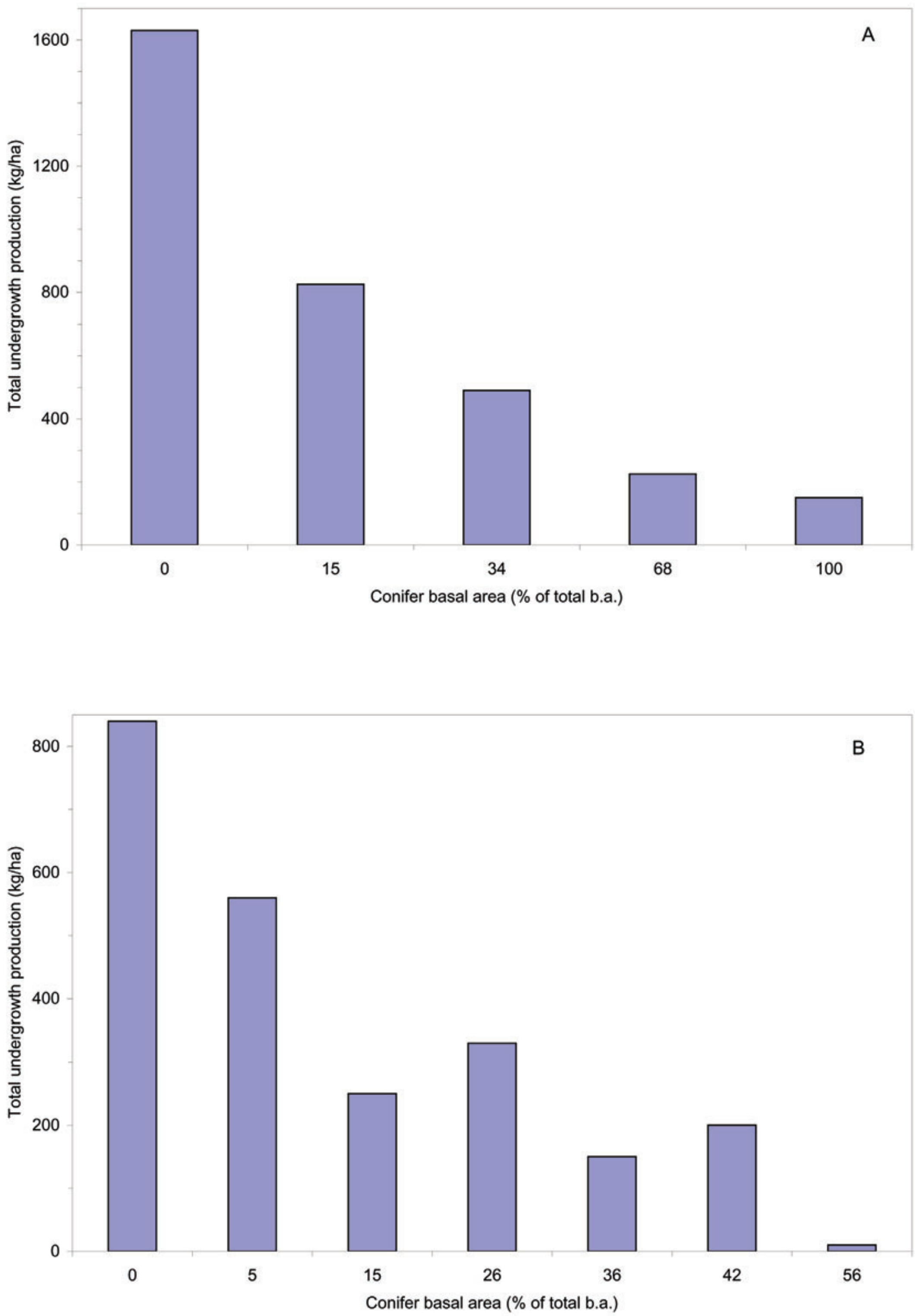

Figure 3—Effect of conifer invasion on annual undergrowth production: A. The Wasatch Plateau of central Utah. B. The Fishlake and Dixie National Forests of southern Utah. Re-analysis of data reported in Mueggler (1985).

Aspen communities have exceedingly high biodiversity–second only to riparian areas on western ranges (Kay 1997). When aspen lands become dominated by either conifer or sagebrush, marked changes in both flora and fauna occur and biodiversity is compromised. Not only is there a loss of forage production, but there is also a substantial decrease in plant species richness associated with the loss of aspen.

Winternitz (1980) reported that the density and diversity of birds was greater in aspen than conifer stands, and McGraw/Bergstrom (1986) found that mature stands of aspen had the most bird species compared to younger stands or those being invaded by conifers. Bird species diversity also increases with the size of aspen stands (Johns 1993). Linder (1995) found that bird community structure is positively correlated with the size of aspen patches.

Other aspects of biodiversity loss are not so readily apparent. For example lichens, bats, and snails are also affected by the decline of aspen. McCune (1997) indicates that a loss of aspen would adversely affect the epiphytic lichen communities that depend on mature to old aspen. Vonhof (1996) found in Southern British Columbia that both big brown bats (*Eptesicus fuscus*) and silver-haired bats (*Lasionycteris noctivagans*) prefer trembling aspen for roost sites. In the Cypress Hills, Saskatchewan, all little brown bats (*Myotis lucifugus)* roost in aspen trees (Kalcounis and Hecker 1996). Crampton and Barclay (1996) monitored two species of bats (little brown bats and silver-haired bats) in aspen stands in Lac La Biche, Alberta, and found that the bats preferred newly dead aspen with heart rot and trees with low leaf cover. Beetle (1997) determined that 21 species of land snails occurred in aspen stands in Yellowstone National Park and that these stands had a greater abundance and variety of snails than coniferous forests on more acidic soils.

Fire Regime Condition Classes

Fire Regime Condition Class 1 (FRCC 1)

Fire Regime Condition Class 1 (FRCC 1) indicates that the fire regimes are within a historical range and there is little risk of losing aspen from the ecosystem. This class includes the stable and seral stands that are reproducing and have little conifer encroachment. Mueggler (1989) speculated that as much as a third of the aspen in the west could be classified as climax or stable. Smith and Smith (2005) reported 16 percent of the aspen stands on the Uncompahgre Plateau, CO, were stable. Therefore, assuming a 60 percent decline in the historical range of aspen, only a portion of aspen dominated land (10 to 15 percent) would be considered functioning normally, in other words, regenerating naturally without major interventions. Stable clones are not subject to succession, primarily because there is a lack of conifer seed source and most natural regeneration is not consumed by ungulates. This leaves only a few aspen in Utah that are able to survive and perpetuate without naturally occurring wildfire or other disturbance. These are approximate figures for Utah and may be more or less depending on various situations. Caution should be used when extrapolating to aspen outside of Utah.

FRCC 1 aspen stands contain no conifers, are multistoried with numerous age classes, and usually have a fairy ring or skirt of regeneration around the outside of the clones, which makes them hard to see through or into (fig. 4) (Bartos 2001). These stands may also have present a large shrub component of species such as chokecherry (*Prunus virginiana* L.) or snowberry (*Symphoricarpos albus* (L.) Blake). Aspen stands in FRCC 1 regenerate without many external forces, such as disease or abiotic factors, acting upon them. The fire return interval in these stands is characterized by either frequent, very low intensity fires that cause some overstory mortality and subsequent regeneration, or more infrequent higher intensity fires every 100 to 200 years that result in new even-aged stands (Brown and Simmerman 1986). These high intensity fires in pure stands of aspen result when there are a lot of fine fuels, and/or a considerable shrub component coupled with extremely dry conditions. The low intensity fire return interval could be as frequent as 7 to 10 years (Baker 1925; DeByle and others 1987). In such cases, FRCC 1 seral stands should be able to regenerate enough sprouts to successfully restock the stand after fire. Baker (1925) states that 6,000 aspen sprouts per hectare (2,500 sprouts/acre) by the end of year 3 are successful post-fire stocking levels. However, ungulate pressure must also be low enough to prevent excessive browsing of the new aspen shoots.

Fire Regime Condition Class 2 (FRCC 2)

Fire Regime Condition Class 2 (FRCC 2) indicates that the fire regimes have been altered and there is a moderate risk of losing key ecosystem components. In Utah, approximately 25 to 30 percent of the aspen are in FRCC 2. Seral FRCC 2 stands have a large conifer component that over time will convert the stand to a mixed conifer type if no disturbance occurs (fig. 5).

Figure 4—Healthy aspen (*Populus tremuloides*) stand with a younger age class at the outer edge.

Figure 5—Aspen (*Populus tremuloides*) stand with a co-dominant overstory of subalpine fir (*Abies lasiocarpa*) and Engelmann spruce (*Picea engelmannii*). Dominant grass and shrub species are slender wheatgrass (*Elymus trachycaulus*) and gooseberry currant (*Ribes montigenum*).

Stable FRCC 2 stands have limited sprouting and some of the mature stems are dead and dying. Sprouting may be limited because of low light levels in the case of conifer encroachment, or if ungulate browsing pressure is heavy, the clone may be sprouting but the sprouts are summarily eaten.

Fire Regime Condition Class 3 (FRCC 3)

Fire Regime Condition Class 3 (FRCC 3) contains all aspen that are not included in the previous two classes. Fire return intervals have been drastically altered in FRCC 3 stands and there is a high risk of losing components of the ecosystem. Approximately 60 percent of the aspen component in Utah is classified as FRCC 3. Because of the lack of burning or other disturbance, these once aspen dominated lands are being replaced primarily by conifers. In FRCC 3 stands, conifer canopy cover exceeds aspen canopy cover and they are now considered mixed-conifer rather than aspen cover types. Without treatment or disturbance, these decadent aspen stands will soon be completely replaced by conifers and lost from the landscape (fig. 6). As these aspen stands age and are replaced by conifers, the aspen begin to die (around 120 years) and fall to the ground and increase fuel loadings. Higher fuel loadings can cause fires to burn more severely, creating significant soil heating. Most aspen suckers regenerate within 6 cm (2.4 inches) of the surface and virtually all regenerate within the upper 12 cm (4.7 inches) of soil (Perala 1991). Therefore, high severity fires could reduce suckering potential by killing the roots, and subsequent stocking levels may not be high enough to successfully regenerate the stand. Aging aspen stands also may not be able to regenerate successfully due to a reduced capacity of the root system to sprout after disturbance. Browsing by both wild and domestic ungulates further complicates the situation. In areas where ungulate pressure is intense, disturbance events are not successful because all suckers are browsed and some aspen clones are ultimately lost. Excessive utilization needs to be addressed before fire or other treatments are returned to the system (U.S. Department of Agriculture 1994). If one goal of management is to restore aspen, then animal use needs to be monitored, evaluated, and adjusted. Otherwise, animals utilizing the aspen regeneration can slow or defeat restoration efforts.

Recommended Treatments

Restoring aspen to a level near its natural range of variability requires that land managers take an aggressive management approach in the very near future. Land managers in the western United States are currently very interested in actively treating their deteriorating aspen stands. Five major risk factors have been identified

Figure 6—Former aspen (*Populus tremuloides*) stand now dominated by ponderosa pine (*Pinus ponderosa*), Douglas-fir (*Pseudotsuga menziesii*), and white fir (*Abies concolor*).

to assist managers in evaluating their aspen resource (Bartos and Campbell 1998a). These risk factors are: (1) conifer cover >25 percent, including reproduction in the understory; (2) sagebrush cover >10 percent; (3) aspen canopy cover <40 percent; (4) dominant aspen trees >100 years of age; (5) aspen regeneration 1.5 to 4.5 m (5 to 15 ft) tall and <1,200 stems per ha (<500 stems/acre).

A prioritized key (table 2) was developed by Campbell and Bartos (2001) that incorporates the above risk factors and is applicable to most aspen stands found in southern Utah. In part, this key was developed with knowledge and expertise of aspen systems that occur in this region. The stands at greatest risk and highest priority are those having conifer canopy cover that exceeds aspen canopy cover. These are currently mixed-conifer rather than aspen cover types. However, with proper treatments, the aspen cover type can usually be restored and sustained. Mueggler (1989) developed a general management decision model for maintaining aspen stands in the west (fig. 7). From work in Ontario, Peterson and Peterson (1992) suggested that stands targeted for treatment need at least 40 parent aspen stems per hectare (16 stems/acre) to produce the minimal acceptable stocking and about 125 parent aspen stems per hectare (50 stems/acre) to fully stock a stand. However, Shepperd (2004) reported successful regeneration of an aspen clone in Arizona with only two remaining parent stems when fencing was used to protect suckers.

Numerous techniques are available for restoring these decadent aspen clones or late successional clones dominated by conifers. These include prescribed burning, clear cutting, cutting and burning, fencing, spraying, ripping, and chaining (Bartos and Harniss 1990; Bartos and Lester 1984). Such treatments should give rise to abundant aspen suckers if the three essential elements to the aspen regeneration triangle are met: 1) hormonal stimulation, 2) proper environment, and 3) protection of new suckers (Shepperd 2001, 2004). Shepperd (2001) lists seven aspen regeneration alternatives: 1) doing nothing, 2) commercial harvest, 3) prescribed fire, 4) mechanical root stimulation, 5) removal of vegetative competition, 6) protection of regeneration from herbivory, and 7) regenerating from seed.

Table 2—Key to the risk factors used to prioritize areas with aspen for restoration and conservation actions in the Intermountain West. Assumption: Aspen are present with a density of at least 50 mature trees per hectare (20 mature trees per acre). Note: Couplet 1 refers to relative cover; couplets 2 to 5 use absolute cover (Campbell and Bartos 2001).

Risk Factor			Priority
1.	a.	Conifer species comprise at least half of the canopy cover.	Highest priority
	b.	Aspen comprises more than half of the total canopy cover.	2
2.	a.	Aspen canopy cover is less than 40 percent; *and* sagebrush usually a dominant understory species exceeds 15 percent cover.	High priority
	b.	Not as above.	3
3.	a.	Conifer cover (including overstory and understory) exceeds 25 percent.	Moderate to high priority
	b.	Conifer cover is less than 25 percent.	4
4.	a.	Aspen regeneration (1.5 to 4.5 m [5 to 15 feet tall]) is less than 1,200 stems/ha (500 stems/acre).	Moderate priority
	b.	Aspen regeneration exceeds 1,200 stems/ha (500 stems/acre).	5
5.	a.	Any two of the following three risk factors are represented: 1 – Aspen canopy cover is less than 40 percent. 2 – Dominant aspen trees are greater than 100 years old. 3 – Sagebrush cover exceeds 10 percent.	Low to moderate priority
	b.	Two of the three risk factors in 5a are not represented.	6
6.	a.	One of the three risk factors in 5a is represented.	Low priority
	b.	None of the risk factors above are represented.	Candidate for properly functioning condition

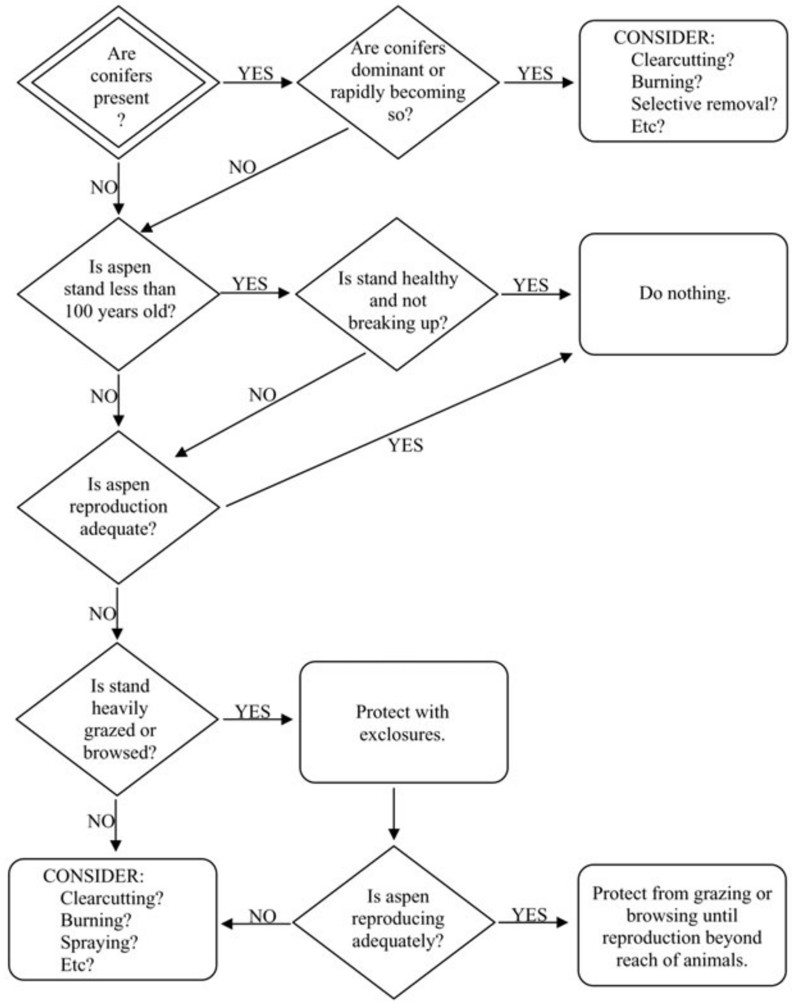

Figure 7—General management decision model for maintaining aspen (*Populus tremuloides*) stands in the western United States. Adapted from Mueggler (1989).

Doing Nothing

This action is suitable for stands in FRCC 1. Management intervention is likely unnecessary if the stand is "showing little sign of decline, disease, or distress from competition, contains multiple age classes, or is successfully suckering" (Shepperd 2001).

Commercial Harvest

Where aspen markets exist, clearfelling is the preferred harvest method to regenerate a declining stand. Partial harvests result in fewer suckers and expose the residual mature stems to breakage, windthrow, and sunscald (Shepperd 2001). Dormant season logging promotes more suckering than spring or summer harvests (Perala 1991; Frey and others 2003). Ohms (2003) found clearfelling aspen clones produced significantly more suckers and increased sucker height compared to the control clones. Treatment response was even more successful in the areas of the clearcuts where browsing was excluded. The successful regeneration of the aspen stand after clearfelling is dependent upon the size of the area treated, unless browsing is controlled (Mueggler and Bartos 1977). If ungulate use is not controlled, treated areas should be greater than 5 ha (12 acres).

Fire

In climax aspen stands, suckering following the death of decadent stems is usually sufficient to maintain the stands. However, in many areas where aspen is successional to conifers, fire can promote sufficient suckering to maintain aspen stands and remove encroaching conifers. Working in central Utah, Baker (1925) stated that a 50-year fire return interval would keep most aspen stands free of conifers. A comprehensive fire guide for treating aspen in the western United States was developed by Brown and Simmerman (1986). In this guide, they evaluated fuels and flammability for five specific types: aspen/shrub, aspen/tall forb, aspen/low forb, mixed/shrub, and mixed/forb. Various factors were evaluated and a probability for a successful burn was given. This information should be useful to the resource manager for developing prescribed fire burn plans.

Bradley and others (1992) prepared a detailed report that discusses fire ecology of forests and woodlands in Utah. This paper identifies various fire groups, some of which include an aspen component. Habitat types developed by Mauk and Henderson (1984), Youngblood and Mauk (1985), and some of the community types discussed by Padgett and others (1989) and Mueggler (1989) are assigned to 13 "Fire Groups" (Bradley and others 1992). Five of these fire groups contain aspen and would be relevant in planning any prescribed burn. These five specific fire groups are: (a) 7-community types where aspen appears to be climax or a long-term seral dominant; (b) 3-Ponderosa Pine Habitat Type (*Pinus ponderosa* P. & C. Lawson); (c) 5-Douglas-Fir Habitat Type (*Pseudotsuga menziesii* (Mirbel) Franco); (d) 6-White Fir and Blue Spruce Habitat Types (*Abies concolor* (Gord. & Glend.) Lindl. Ex Hildebr. and *Picea pungens* Engelm.); and (e) 10-Subalpine Fir Habitat Types. Items b through e contain aspen that are successional to various coniferous species indicated in the habitat type.

Fire that kills most of the aspen overstory produces more shoots than less intense fires that result in lower overstory mortality. Keyser and others (2005) reported high severity fires (>75 percent aspen mortality) in aspen stands produced more shoots, killed encroaching conifers, and had greater shoot height growth than low severity fires (25 to 75 percent aspen mortality) or unburned aspen stands. Aspen sprout aboveground biomass was 10 times greater on high severity sites (100 percent aspen mortality) than intermediate severity burned sites (~50 percent aspen mortality) after a wildfire in Arizona (Bailey and Whitman 2002).

Mechanical Root Stimulation

Severing aspen roots from parent stems can stimulate suckering. Shepperd (1996) found that tipping over mature aspen stems resulted in significantly more sprouts than stems that had been chainsaw felled. Ripping around the perimeter of a decadent aspen clone can also stimulate suckering. Variations of ripping around aspen clones have proved successful on the Coconino National Forest in Arizona (Shepperd 2004). This method can expand the size of the clone and introduce new age classes without cutting existing mature stems.

Removal of Vegetative Competition

Removing competing vegetation increases light to the understory (Shepperd 2001). Vegetation removal can be done alone or in conjunction with other treatments to increase aspen regeneration. In northern California, conifers up to 66 cm (26 inches) diameter at breast height (DBH) were removed around existing aspen stems to encourage suckering (Jones and others 2005). Aspen stem density was significantly higher in the treated stands 4 years post-treatment when compared to controls. In a separate study to re-invigorate the clone, all conifers were removed and an elk-proof fence was installed around a declining aspen clone consisting of only two remaining live aspen stems (Shepperd 2004). Four years post-treatment, the stand had produced numerous sprouts and the treatment was considered successful.

Protection of Regeneration from Herbivory

In some areas, confounding factors exist that will challenge the manager to be successful. Treatment to induce aspen suckering is not enough in areas where there is extensive ungulate pressure (both domestic and wildlife) because uncontrolled ungulate use could cause some aspen treatments to be unsuccessful. Such actions to induce suckering must not be initiated before relief from excessive browsing is obtained (U.S. Department of Agriculture 1994; Brown 1995). Treatments to stimulate aspen regeneration that are less than 2 ha (5 acres) may concentrate deer use and cause excessive browsing (Mueggler and Bartos 1977). In Colorado, harvesting several adjacent large (6 to 8 ha [15 to 20 acres]) units at one time resulted in successful aspen regeneration, even with large ungulate populations (Crouch 1983). Additional protection, such as temporary fencing, may be required to permit the growth of the aspen suckers beyond the reach of browsing animals if ungulate populations are high. Excessive browsing of sucker reproduction is perhaps the most detrimental influence on the successful regeneration of stands burned or otherwise treated to simulate suckering, as well as in stable aspen communities which usually produce ample suckers for regeneration as the aspen overstory gradually thins with age. Obtaining ample suckers to replace aspen stands is usually not a problem. Allowing these suckers to grow beyond the reach of ungulates is the issue (Shepperd and Fairweather 1994; Rolf 2001).

Various barriers to protect aspen regeneration under these conditions were evaluated by Kota (2005) in the Black Hills of South Dakota. He found that a barrier of hinged trees was useful in protecting aspen regeneration. Hinging is accomplished by felling the tree (live or dead) and keeping the bole on the stump, thus creating a barrier around patches of aspen regeneration. Standard fencing practices can also be used but are sometimes cost prohibitive, especially if large areas need to be protected. Kay and Bartos (2000) surveyed all known aspen exclosures on the Dixie and Fishlake National Forests in Utah in 1995 and 1996. The exclosures were originally established between 1930 and 1970 and were designed to exclude either all ungulates (livestock and wildlife) or just livestock. Aspen in the total exclusion exclosures successfully regenerated without other treatments and developed multi-sized, multi-aged stems. Aspen in the livestock-exclusion exclosures either failed to produce new stems greater than 2 m (6.5 ft) tall, or regenerated at lower densities than the total-exclusion plots. Outside the exclosures, aspen either failed to regenerate successfully or regenerated at lower densities than the livestock-exclusion plots. Aspen regenerated successfully on the livestock excluded plots and outside the exclosures only when mule deer populations were low.

Regenerating from Seed

The feasibility of seeding aspen in areas of abundant soil moisture is currently being tested (Shepperd 2001). If successful, this technique could be used to restore aspen where it has been lost from the system.

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