Chapter 4

Pinyon/Juniper Woodlands

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Introduction

Pinyon-juniper woodlands occur in 10 states and cover large areas in many of them. These woodlands can be dominated by several species of pinyon pine (*Pinus spp*. L.) and juniper (*Juniperus spp*. L.) (Lanner 1975; Mitchell and Roberts 1999; West 1999a). A considerable amount of information is available on the expansion of the woodlands that has occurred over large parts of the geographic ranges of the tree species involved (Miller and Tausch 2001). In southern Utah, the woodlands contain Utah juniper (*Juniperus osteosperma*), singleleaf pinyon (*Pinus monophylla*), and two-needled pinyon (*Pinus edulis*). Singleleaf pinyon is present in the western, and two-needled, or Colorado pinyon, in the eastern portions of the region. Each species can occur alone, or in a mix of one of the pinyon species with Utah juniper. Overall, information on woodland ecology for the southern Utah study area is limited. For this reason, the available literature for the woodlands in general, but particularly for the Great Basin, will be summarized and possible implications for southern Utah indicated.

Wherever they are found, pinyon and juniper woodlands are a landscape scale phenomenon (fig. 1). These trees have large ecological amplitudes and are capable of invading into, and dominating, a wide range of communities (Miller and Tausch 2001; West and others 1978a,b; West and others 1998). Their range can extend

Figure 1—Pinyon-juniper landscape with an old chaining treatment.

from the upper edge of salt desert shrub communities at the lowest elevations to the lower fringes of subalpine communities at the higher elevations (West and others 1998). The dynamics of these woodlands and their associated vegetation processes need to be understood at landscape scales. In the Great Basin, most of the communities where the trees have established were dominated by sagebrush (*Artemisia* spp.) (West and others 1978b). The woodland tree species are found associated with a range of sagebrush species and subspecies, and each taxon can represent a range of plant communities. Sagebrush presence usually extends in elevation both below and above the range of potential tree distribution, and their associated ecosystems are the matrix within which the trees occur. As a result, the dynamics of the woodlands are linked with, and in many ways dependent on, the dynamics of the shrub-dominated ecosystems involved.

Where they co-occur, sagebrush and woodland communities can have different states or levels of co-dominance within the overall successional dynamics of the sagebrush/woodland ecosystem complex of a particular landscape area. It is not possible to really understand or manage pinyon and juniper at these landscape scales without understanding the entire topography, soils, and vegetation complex for each landscape area of interest. Because this ecosystem complex is dynamic and highly variable across the landscape, identification of community type is determined from the species composition of the associated, usually sagebrush-dominated, communities (West and others 1978b; 1998). At any woodland location on the landscape, its successional status and associated ecosystems are the result of complex interactions of topography, soils, environmental conditions, past patterns of disturbance, and how successional processes have operated through time. In some locations in southern Utah, Gambel oak (*Quercus gambellii*) can be a part of the community (Thompson 1999; West 1999b) and can influence community dynamics.

Historical Conditions

How the patterns of disturbance were spatially distributed across the landscape and the subsequent successional changes through time following those disturbances were much different prior to Euro-American settlement than afterward. Prior to settlement, the primary disturbance was fire (Gruell 1999; Miller and Tausch 2001). The pattern and behavior of fire was closely related to the unique interactions of topography, soils, environmental conditions, and vegetation composition present at that time on each landscape area of interest. These complex community types contained an equally complex mix of fuel types and levels that determined fire pattern and behavior. Across the Great Basin region, trees were present on less than one-third of the area they currently occupy. For areas where trees were present, their pre-Euro-American settlement densities were on average about one-fourth to one-tenth or less of the density present at the beginning of the twenty-first century (Bauer 2006; Miller and others, in press). Similar densities appear to have been present on the eastern Colorado Plateau portion (Floyd and others 2000; Romme and others 2003) of the southern Utah study area.

Vegetation cover prior to pre-European settlement varied widely, depending on local conditions, from less than 20 percent to over 80 percent on the most productive sites (West and others 1998). The majority of sites were below 50 percent total cover (Miller and Tausch 2001). Similar averages and variability in cover appear to have been present in the southern Utah area. Total vegetation cover appears to have always been relatively the same for similar sites, whether they were sagebrush-dominated or tree-dominated. Total biomass, however, varied from about seven to nearly 20 times greater when a site was tree-dominated versus sagebrush-dominated (Tausch and Tueller 1990).

The mix of sagebrush- and tree-dominated sites over the pre-Euro-American settlement landscape and the distribution of size and age classes within tree-dominated sites depended on the interactions between disturbance patterns and post-disturbance successional development. The primary control on these differences appears to be landscape variation in the pattern and frequency of fire. The heterogeneity of the landscape, combined with variation in successional processes, associated heterogeneous mix of community types, and associated fuel types and fire regimes, resulted in the maintenance of vegetation that varied widely across the landscape.

Pre-settlement old growth woodlands were commonly found on relatively fire safe sites with limited surface fuel loads (fig. 2) (Burkhardt and Tisdale 1969; Burwell 1998, 1999; Holmes and others 1986; Miller and Rose 1995, 1999; West and others 1998). The high level of landscape and associated vegetation heterogeneity present prior to European settlement resulted in a high degree of edge between sagebrush and tree-dominated communities (Tausch and Nowak 1999). These heterogeneous conditions often represented optimum habitat for many species of wildlife (Miller and Tausch 2001).

Figure 2—The presence of very old Utah juniper (*Juniperus osteosperma*) suggests that this rocky site would rarely, if ever, develop a grassy understory capable of carrying a surface fire.

Then, as now, larger fires tended to occur during periods of drought (Betancourt and others 1993; Swetnam and Betancourt 1998). Insects, diseases, and native ungulates appear to have played a widespread but relatively minor role. Information is more limited for the Colorado Plateau than the Great Basin, but it indicates fire may have been less frequent in many areas compared to the Great Basin (Floyd and others 2000; Romme and others 2003). Overall, there was a dynamic balance between disturbance and succession resulting in a complex shifting distribution of the woodland and sagebrush dominance throughout the landscape.

It is the interaction between topography, vegetation, and fire that influenced both the patterns of disturbance and the kinds of communities that were found on a particular position on the landscape at a particular point in time. The deeper soils in the canyon bottoms and swales are generally more productive, particularly for the herbaceous species. These locations appear to have had the highest fire frequencies (Bauer 2006; Burwell 1998; Gruell 1999; Swetnam and Basian 1996). As soils become shallower, generally as the topography becomes steeper, the abundance of perennial herbaceous vegetation is limited to years with above average precipitation. On these locations, fires appear to have been less frequent, increasing the probability of dominance by trees. The most fire-safe sites, generally on the steepest slopes or shallowest soils, were generally the locations of woodlands that were often several centuries old (Miller and others 1999; Waichler and others 2001). These sites also have generally low levels of productivity of perennial herbaceous vegetation. A few pre-settlement aged woodlands appear to be present from nothing more than the off-chance of not having burned for over 200 years (Miller and others, in press).

Current Conditions

Euro-American settlement activities have caused major changes to the composition of vegetation within the Great Basin (Miller and Tausch 2001; Rowland and Wisdom 2005). The rapid woodland expansion observed during the late 1800s and early 1900s resulted from a combination of conditions (Miller and Tausch 2001; Miller and others, in press): (1) heavy livestock grazing that removed the herbaceous vegetation (fine fuels), (2) the associated reduction in the presence of fire (Heyerdahl and others 2001; Savage and Swetnam 1990; Swetnam and Betancourt 1998), and (3) wet conditions that created an ideal situation for tree establishment (Antevs 1938). The resulting expansion and increasing dominance of the trees in the Great Basin has continued to the present (Burkhardt and Tisdale 1976; Cottam and Stewart 1940; Miller and Rose 1995, 1999; Miller and others, in press; Tausch and others 1981).

Livestock grazing, particularly in the late 1800s and early 1900s (Young and Evans 1989), generally had the largest impact on the vegetation composition. Grazing reduced the herbaceous vegetation cover, which resulted in a reduction in fire frequency (Burkhardt and Tisdale 1969, 1976; Campbell 1954; Ellison 1960; Miller and Rose 1999; Miller and Tausch 2001). The reduction of herbaceous species by grazing also promoted an increase in shrub cover. The shrubs acted a nurse crop and promoted tree seedling establishment (Burwell 1998, 1999; Chambers and Vander Wall 1999; Chambers and others 1999; Chambers 2001; Cottam and Stewart 1940; Eddleman 1987; Madany and West 1983; Miller and Rose 1995, 1999). With the reduction in fire frequency, the new tree seedlings were able to survive and the areas of woodlands expanded. As with pre-settlement woodlands, total vegetation cover of expansion woodlands remains relatively similar to the shrub cover that preceded tree dominance (Tausch and Tueller 1990; Tausch and West 1995). Therefore, when the shrub layer was absent, the establishment of the trees was more limited (Erdman 1970; Everett and Ward 1984).

Much of the woodland expansion has been into the more productive sites (for example, canyon bottoms and swales). In the absence of fire, the trees are well adapted and competitive in these more productive locations. Prior to tree expansion these areas represented some of the more diverse and productive sagebrush ecosystems in the region and currently support, or will support, some of the highest levels of tree dominance and fuel loads. Pre-Euro-American settlement woodlands have had up to a 10-fold increase in tree densities during this period (Bauer 2006; Miller and others, in press). Density increases may be less on the eastern Colorado Plateau portion (Floyd and others 2000; Romme and others 2003) of the southern Utah study area. As the area of tree dominance continues to increase in the Great Basin, the heterogeneous sagebrush-dominated ecosystems are being replaced by homogenous woodlands (fig. 3) (Miller and Tausch 2001; Milne and others 1996; Tausch 1999a).

Before about 1870, woodlands occurred on less than 10 percent of their currently occupied area in the northwest Great Basin (Miller and others 1999) and on less than 30 percent in the central and southern Great Basin (Creque and others 1999; Miller and others, in press; Miller and Tausch 2001; O'Brien and Woodenberg 1999; Rogers 1982; Tausch and others 1981). Little information is available on the pre-settlement woodlands of the Colorado Plateau. Expansion woodlands now cover an average of three to four times the pre-Euro-American settlement area (Chambers and others 2000a, b; Miller and others, in press). This woodland expansion has proceeded at a nearly continuous rate across the Great Basin over the last 100 years (Chambers and others 2000a, b; Miller and others, in press) and possibly equals or exceeds previous woodland expansions of the Holocene (Miller and Wigand 1994). Consequently, sagebrush communities will continue to decline as this tree dominance continues to increase (Despain and Mosley

Figure 3—Initial establishment of Utah juniper (*Juniperus osteosperma*) in a stand of big sagebrush (*Artemisia tridentata*).

1990; Miller and others 1994; Miller and Tausch 2001; Suring and others 2005; Tress and Klopatek 1987; West 1984).

Because of the generally slow growth of the trees, it has taken all of the approximately 100 years since Euro-American settlement for a doubling of the fuel loads to take place. Trees in the extensive areas of woodlands that have established over the last century are now rapidly maturing, and as they do, the fuel loads are increasing at an accelerated rate on these sites. On the majority of these areas, the density needed for trees to dominate is now in place (Miller and others, in press). While it took fuel loads in the expansion woodlands the past 100 years to double (Chambers and others 2000a, b), they will double again in the next 40 to 50 years (Miller and Tausch 2001). The expansion of tree distribution into new sagebrush areas is continuing (Betancourt 1987; Knapp and Soule 1998; Miller and others 2000; Miller and Tausch 2001; Suring and others 2005; West and Van Pelt 1986), and with it continues the increase in the level and continuity of tree-dominated fuel loads. Similar patterns appear to exist in southern Utah.

The rate of the transition from sagebrush ecosystem to tree-dominated woodland is variable and depends on the site potential, sagebrush species and subspecies present, and rate of tree establishment (Miller and Tausch 2001; Miller and others, in press). In general, a minimum of 60 to 90 years is required for trees to dominate a site (Barney and Frischknecht 1974; Huber and others 1999; Miller and others 1999; Miller and Rose 1995; Miller and Tausch 2001; Tausch and Tueller 1990; Tausch and West 1995). The decline in sagebrush biomass is not proportional to the increase in tree biomass (fig. 4). When the trees have reached about one-half their potential biomass, sagebrush biomass has declined to about one-third, sometimes one-fourth, of its former level (fig. 5) (Miller and Tausch 2001; Tausch and West 1995). The pattern of the decline is relatively consistent across the sagebrush species and subspecies, although the rate involved is not (Miller and others, in press; Miller and Tausch 2001). This expansion may be facilitated by the increasing CO_2 in the atmosphere (Johnson and others 1993). Similar changes in the landscape level patterns are present in the woodland changes of southern Utah.

As the dominance of the trees continues to increase beyond Phase I or Phase II (fig. 5), not only will fuel loads double from current levels over the next few decades, but the continuity of those fuels across the landscape will rapidly increase. Because of the young age of the trees, the ongoing increases in fuel loads are primarily in the highly flammable fine fuels. Once the trees dominate a site, these fine fuels can reach 9,000 kg/ha (10 tons/acre) on more productive sites (Chambers and others 2000a, b). Overall, woodland sites in the Great Basin vary widely, but probably have average fuel levels of about two-thirds of those sampled on the more

Figure 4—Increasing dominance and water use by Utah juniper (*Juniperus osteosperma*) are the likely cause of the die-back of big sagebrush (*Artemisia tridentata*) on this site.

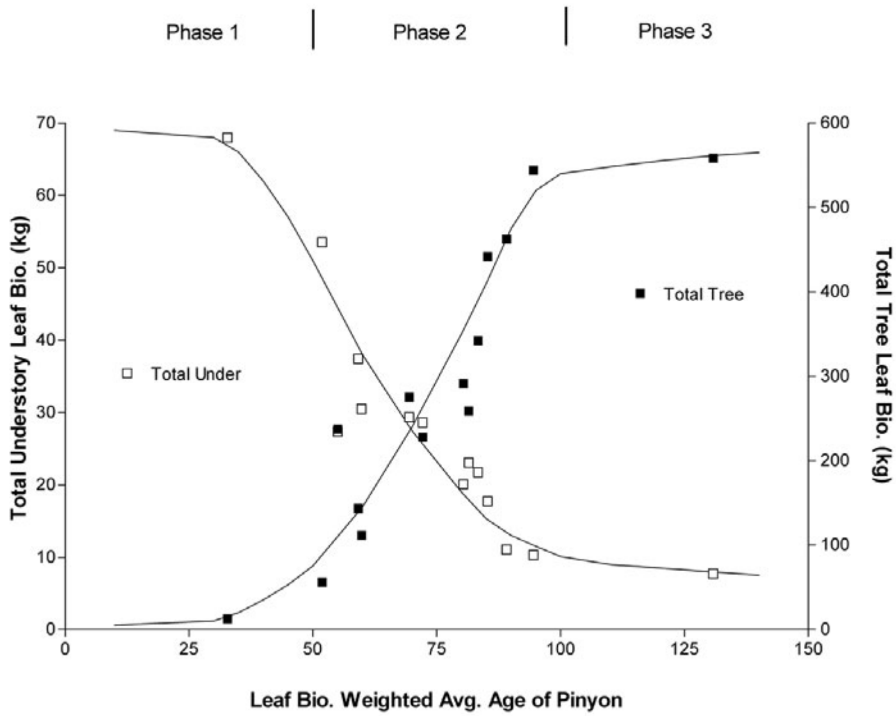

Figure 5—Comparison of both the total tree leaf biomass (closed boxes) and total understory leaf biomass (open boxes) over time as indexed by the range in leaf biomass weighted average age of pinyon for 14 plots in southwestern Utah (Tausch and West 1995; Miller and Tausch 2001). X-axis = (sum of [tree age * leaf biomass] over all trees) / total stand leaf biomass. Phase I is the early tree establishment phase, Phase II is the period of tree growth and increasing tree dominance, and Phase III is tree dominance in expansion woodland sites.

productive central Nevada sites. The increasing crown closure of post-settlement woodlands is increasing the occurrence of crown fire (Miller and Tausch 2001; Tausch 1999a,b; West 1999a,b). Similar patterns appear to be taking place in southern Utah.

Post-fire vegetation response depends on the composition of the shrub-dominated community and the level of tree dominance (Barney and Frischknecht 1974; Dhaemers 2006; Erdman 1970; Everett and Ward 1984; Pickett 1976; West and VanPelt 1986). As the trees dominate a site, there is a decrease in the herbaceous species (Dhaemers 2006; Erdman 1970; Koniak and Everett 1982; West and others 1978a; West and others 1998), an increase in soil erosion (Wilcox and Breshers 1994), changes in soil fertility (Rau 2005), losses in forage production, and changes in wildlife habitat (Miller and Tausch 2001). The more dominant the trees are at the time of disturbance, the more the plant species composition of the communities that follow the disturbance can change. The intense crown fires more frequently occurring on tree-dominated sites further reduces understory plant species survival (Tausch 1999a).

Exotic species are changing the outcome of post-fire response (D’Antonio and Chambers 2006). The higher the level of tree dominance, the higher the probability that a crown fire will leave an open site. These open sites are increasingly being dominated by exotic plant species, many of which are annuals, such as cheatgrass (*Bromus tectorum*) (Chambers and others 2007; Roundy and others, in press). The presence of cheatgrass can cause increases in fire size and frequency (D’Antonio and Chambers 2006; Swetnam and others 1999; Tausch 1999b; Whisenant 1990; Young and Evans 1973) and homogeneity of those communities across the landscape (Young 1991; Young and Evans 1973). More recently, exotic perennials have begun establishing in these areas (D’Antonio and Chambers 2006). Once this conversion occurs, any return to the original sagebrush ecosystem, or even eventually to woodland, is often no longer possible without extreme restoration efforts (Miller and Tausch 2001). Our ability to assess the susceptibility of the communities associated with different sagebrush species and subspecies to conversion to cheatgrass is improving (Chambers and others 2007; Roundy and others, in press).

The problems associated with the exotic grasses are increasing through time in proportion to the level of CO_2 in the atmosphere. Three ecotypes of cheatgrass from three elevations in the Great Basin have been investigated for the effects of increasing levels of atmospheric CO_2. Four levels of CO_2 were used, ranging from pre-settlement to an estimate of 2020 levels. From pre-industrial levels to the estimated 2020 level, the productivity of the upper elevation ecotype doubles, the mid-elevation increases 2.5 times, and the low-elevation triples (Ziska and others 2005). Flammability also increases (Blank and others 2006). Cheatgrass will have an increasingly negative impact over time on any woodland site where it becomes established following fire.

Prior to European settlement, ecosystems in the Great Basin and southern Utah were, on a landscape basis, more resilient to disturbance by fire. Fire regimes differed between the different sagebrush species and subspecies-dominated ecosystems and their associated communities, but were relatively consistent within each community. With tree expansion and increasing dominance there has been an increase in homogeneity and a loss of resiliency. Even old growth woodland areas that were relatively protected from fire prior to settlement are increasingly susceptible to fire damage from the adjacent tree-dominated areas (Miller and Tausch 2001) that increasingly dominate fire behavior (Hann and others 2004). Developing effective restoration procedures requires more study on the ecology, structure, and long-term dynamics of the woodlands and their interaction with the associated sagebrush ecosystems (Chambers 2005; Dhaemers 2006; Miller and others, in press).

Fire Regimes

Five fire regime classes have been defined for assessing landscape dynamics for historic or past fire patterns and frequency (table 1) (Hann and others 2004; Hann and Strohm 2003; Romme and others 2003; Schmidt and others 2002; Waichler and others 2001). Class I was very rare in the southern Utah area prior to Euro-American settlement and is not covered here. Prior to settlement, a heterogeneous mix of fire regime classes II through V often existed within relatively

Table 1—Natural (historical) fire regime classes from Hann and others (2004) as interpreted by the authors for this analysis.

Fire regime class	Frequency (mean fire return interval, in years)	Severity	Community structure description
I	0 to 35 frequent	Surface mixed	Open woodland or savannah structures maintained by frequent fire; also includes frequent mixed severity fires that create a mosaic of different age post-fire open woodland, early to mid seral woodland structural stages, and shrub or perennial grass dominated patches.
II	0 to 35 frequent	Replacement	Shrub or shrub/perennial grass maintained or cycled by frequent fire: fires kill non-sprouting shrubs, such as sagebrush, which typically regenerate and become dominant within 10 to 20 years; fires remove tops of sprouting shrubs such as rabbitbrush, which typically resprout and can dominate after several years; fires typically kill most tree regeneration.
III	35 to 200 infrequent	Mixed surface	Mosaic of different age post-fire woodland, early to mid-seral (Phase I and Phase II, fig. 6) woodland structural stages, and shrub or shrub/perennial grass dominated patches maintained or cycled by infrequent fire.
IV	35 to 200 less frequent	Replacement	Large patches of post-fire shrub or shrub/perennial grass dominated structures, or early to sometimes late seral (Phase I to Phase III, fig. 6) woodland cycled by infrequent replacement fire.
V	> 200 rare	All types	May have large patches of similar post-fire shrub or shrub/perennial grass dominated structures, or early to usually late seral (Phase I to usually Phase III, fig. 6) woodland cycled by rare replacement fire. In systems with little fire or only localized torching effects of lightning strikes effects the composition and structure may be complex.

small areas of the landscape. Because of the vegetation heterogeneity that existed prior to settlement, the abundance and distribution of the various types were first controlled by topographic heterogeneity and secondly, were both determined and controlled by the vegetation heterogeneity. Some separation of, or differences in, fire regime probably existed between the sagebrush species and subspecies present prior to settlement reflecting differences in their specific site conditions. Even with the abundance and widespread distribution of areas with the fire regimes classes II, III, and IV, large areas representing fire regime V, often old growth woodlands, were still present and widely distributed within, and often surrounded by the other, fire regimes.

Since Euro-American settlement, increasing homogeneity of the vegetation has resulted in increased fuel loads and continuity. Areas that were in fire regime V are now in fire regime IV, or sometimes even III. This trend will continue as the surrounding vegetation changes. The vegetation heterogeneity that resulted from differences between sagebrush species and subspecies is generally disappearing (Miller and Tausch 2001). Many areas are increasingly at risk of fire and increased fire size and frequency (Hann and Strohm 2003; Swetnam and others 1999). Despite the increasing appearance of homogeneity with woodland expansion, the vegetation response following a crown fire can still be driven as much by the differences between the sites identified by the original sagebrush community as by the level of tree dominance.

Fire Regime Condition Classes

Three Fire Regime Condition Classes (FRCC) have been defined for assessing the departure of current vegetation communities from historical vegetation structure and fire patterns (chapter 1, table 2, this volume) (Hann and Strohm 2003). Overall, the appearance of a particular woodland site and its associated area of the landscape determine the effective FRCC. This is because the context of the surrounding landscape, particularly where it is represented by expansion woodlands, can drive fire behavior and severity independent of the conditions of a particular old growth woodland site (Hann and others 2004). Effectively restoring a mix of sagebrush and woodland dominance at the landscape level also requires the restoration of the former landscape heterogeneity that makes a dynamic stability with fire possible.

Prior to settlement, most of the sagebrush/woodland areas of the Great Basin, and apparently southern Utah as well, were in FRCC 1. Woodlands and sagebrush ecosystems were in a dynamic balance from areas that burned two to three times a century, to areas that burned about once a century, to areas where fire occurred at intervals greater than 200 years. Areas that burned more frequently were sagebrush and bunchgrass dominated at the time of the fire, although some trees may have established after a previous fire. Because of the ongoing changes, most of the Great Basin is at least in FRCC 2. Many sites may already be in FRCC 3 or are rapidly approaching that condition.

For many woodlands, FRCC sometimes does not depend so much on what an individual pinyon-juniper stand looks like, but on the probable pre-settlement community composition and the current landscape context within which it is located. For example, an area that was old growth woodland prior to settlement could be in FRCC 1 when surrounded by sagebrush-dominated communities. The surrounding areas remained in sagebrush because those communities supported more frequent fire of lower intensity. Under these conditions fires, usually did not crown into the adjacent old growth woodlands and they appear to have remained relatively fire safe (fig. 6). Most of the expansion woodlands, however, are occurring in areas that were usually sagebrush-dominated prior to settlement, changing the pattern and behavior of fire compared to what occurred prior to tree dominance. These tree-dominated expansion woodlands often have continuous canopy cover, which can support high intensity crown fires under high wind conditions. These adjacent, expansion woodland sites can now drive fire behavior (Hann and others 2004). As a result, the old growth woodland stand becomes FRCC 3, even where little change to the woodlands has occurred. However, many of these pre-Euro-American settlement woodlands have also experienced increases in tree density, sometimes up to 10 times or more over the last century (Bauer 2006; Miller and others, in press), which is also directly changing the Fire Regime Condition Class of these sites. Combined, these vegetation changes are resulting in changes in the size, intensity, and frequency of fires for all parts of the landscape. This includes the increasing risk for conversion to cheatgrass.

For the Great Basin as a whole, little of the current woodland area is in FRCC 1. These are the pre-settlement woodlands that have not seen a significant increase in tree density and may represent less than 10 percent of today's total (Bauer 2006; Miller and Tausch 2001). Future loss of some amount of current sagebrush-dominated ecosystems to tree encroachment is still possible (Suring and others 2005). The situation in southern Utah appears

Figure 6—An old growth Utah juniper (*Juniperus osteosperma*) growing with pinyon pine (*Pinus* spp.). The tree likely reached this age because of inadequate surface fuels to carry high intensity fire and stand density was too low to support crown fires. The increased tree density is increasing the risk of lethal fire.

to be similar. Depending on location and community, current vegetation attributes put one-half to two-thirds of the current woodland/sagebrush ecosystem complex in the Great Basin in FRCC 2. Most of these woodlands that are currently in FRCC 2 will be rapidly transitioning to FRCC 3 over the next 40 to 50 years. This has serious implications for habitat for multiple species (Wisdom and others 2005). Similar changes appear to be occurring in southern Utah with similar consequences.

Recommended Treatments

The goals of treating woodlands include fuel load reductions, restoration of sagebrush communities, increasing the heterogeneity of the landscape and associated disturbance processes, improving watershed protection, enhancing wildlife habitat, and increasing forage production (Miller and Tausch 2001). The locations of the treatment sites or patches should be based on topographic features and areas that tended to have a higher fire frequency and thus, historically were more likely dominated by sagebrush communities. These are areas with deeper soils and higher herbaceous vegetation productivity that can carry fire. Retaining pre-settlement woodland sites requires as much or more effort to restore the surrounding communities as it does to restore the pre-settlement site.

Many treatment procedures have been attempted, but they have often been unsuccessful over the long term because of the lack of information about treatments (Chambers 2005; Tausch and Tueller 1977, 1995). A focus on landscape scales, rather than on just individual project scales, can improve treatment effectiveness (Hann and Bunnell 2001). Central to this has been the general lack of recognition of the variability of the communities that the trees are capable of dominating, and the range of disturbance histories represented by the previous communities. Because there is even less direct information available for the southern Utah woodlands, the distribution and extent of similar conditions and patterns of change in the area need to be determined on a site-specific basis.

Tree Removal

Tree removal is the primary management option for restoring areas affected by the ongoing woodland expansion. However, additional treatments have been proposed, many of them using new techniques. First and second order effects, and the success and longevity of the outcomes of any treatment, are highly specific to the site and the method used, how the treatment is used and its timing. For a detailed description of common treatments in pinyon-juniper communities, refer to the restoration chapter in Monsen and Stevens (1999) and Monsen and others (2004). However, some general guidelines are becoming apparent as the results from past and current studies improve our understanding of how these treatments interact with the vegetation dynamics in the woodland zone (Chambers 2005; Miller and Tausch 2001; Monsen and others 2004). Applied in the right way, at the right place, and at the right time with the proper follow up, if needed, any of the existing or proposed treatments can have positive outcomes.

Prescribed Fire

Prescribed fire may be used to remove trees and restore sagebrush communities before tree dominance is so high it reduces surface fuels to a low enough level that they cannot carry fire. Once tree dominance is at the high levels of late Phase II or Phase III (fig. 5) for an extended period of time, susceptibility to the establishment of exotics such as cheatgrass increases. Once these levels of dominance are reached, some form of mechanical treatment followed by seeding is necessary to reduce the level of tree dominance (Chambers 2005). This allows recovery of sagebrush and herbaceous vegetation before the use of prescribed fire can more fully restore a sagebrush ecosystem.

Prescribed fire in pinyon-juniper has been used to control the establishment of trees, increase forb productivity, increase habitat diversity, control invasion of other conifers, alter herbivore distribution, enhance forage palatability and nutritive quality, and prepare sites for reseeding (Bunting 1990). While prescribed fire can be beneficial, many limitations exist. Vegetation response following fire depends on the composition of the shrub community on a site and the level of tree dominance (Barney and Frischknecht 1974; Dhaemers 2006; Erdman 1970; Everett and Ward 1984; Monsen and others 2004; Pickett 1976; West and VanPelt 1986). As trees increasingly dominate a site, the associated sagebrush ecosystems are greatly reduced (Chambers 2005; Erdman 1970; Koniak and Everett 1982; West and others 1998). This reduction in fine fuels often makes it difficult for a fire to carry through a mid-successional stand. If fire does occur, increasing tree dominance increases the recovery time of herbaceous plants and increases the potential for invasion of exotic plants and erosion (Bunting 1990; D'Antonio and Chambers 2006). Bruner and Klebenow (1979) developed an index to predict when fire will carry through mid-successional pinyon and juniper based on wind speed, shrub and tree cover, and air temperature. Dangerous burning conditions exist when the index is greater than 130. Optimal prescribed burning conditions are an index between 125 and 130. This can be modified by fuel moisture levels. Tree-dominated woodlands can be easier to burn than the mid-successional woodlands and are increasingly carrying large crown fires (Miller and Tausch 2001).

Mechanical Thinning

Chaining and thinning are the most commonly used mechanical methods to reduce tree cover. This may be necessary prior to prescribed burning in order to reduce crown fuels and stimulate understory vegetation. In Spanish Fork Canyon, UT, chaining increased total ground cover from 47 to 80 percent and forage production from < 22 kg/ha (<20 lbs/acres) to 1,120 kg/ha (1,000 lbs/acres) 7 years after treatment (Chadwick and others 1999). Similar increases were seen between 4 and 7 years after chaining in eastern Nevada (Tausch and Tueller 1977, 1995) This initial increase in ground cover resulted in significantly less runoff and soil erosion than the control area (Farmer and others 1999). The size, type, and arrangement of the chain can be varied to accomplish different objectives and control the size and amount of trees removed. Stevens and Monsen (2004) provide basic guidelines for chaining in pinyon-juniper. Double chaining in opposite directions removes additional trees missed in the first pass and covers the seed after the area has been broadcast seeded prior to the second pass (Stevens 1999a). A once over chaining is appropriate if sufficient understory remains, trees are sparse and mature, and seeding is not required (Stevens 1999b).

Chaining for tree control increases herbaceous biomass, but can be short-lived. Often after the 4- to 7-year increase there can be a rapid decline to pre-chaining levels in 25 years as a result of accelerated growth of surviving trees (Tausch and Tueller 1977, 1995). Although usually a stand alone procedure, chaining should generally be used only as an effective first treatment followed by a second treatment, such as prescribed fire, which would remove the surviving trees.

Thinning overstory trees with handsaws reduces tree cover and causes less soil disturbance than chaining (Loftin 1999). In a case study in New Mexico, Loftin (1999) reported 2.5 times greater herbaceous cover two growing seasons after hand felling juniper trees without seeding. This method can also be marketed as a fuelwood sale to offset costs. In an economic comparison of chaining versus thinning using chainsaws, Chadwick and others (1999) found thinning cost 44 percent more than chaining. In the same study, thinning did not create an effective seedbed, and subsequent forage production was low compared to the chaining treatment. The different responses between the two studies are most likely due to differences in pre-treatment site conditions. This underscores the importance of choosing appropriate site-specific treatments. Hand thinning can be as equally short-lived as chaining and should also be considered as either a pre-treatment procedure before prescribed fire, or a regularly repeated treatment.

Mastication is another increasingly popular mechanical thinning method. This method grinds and chips trees to reduce tree cover and compact fuel beds. Over 13,360 ha (33,000 acres) have been masticated in Utah alone (Bruce Roundy, personal communication). Because mastication is such a new treatment, the ecological effects are largely unknown and warrant future research.

Seeding

Seeding may be required to prevent the establishment of exotic weeds if the understory is depauperate (Thompson and others 2006). After a tree removal treatment, seeding should occur prior to the next growing season to restrict the establishment of exotics (Stevens and Monsen 2004). Fall seeding is the most ideal time to seed in the Intermountain West, although in southern Utah, seeding just prior to mid-July monsoons has also been successful (Stevens 1999b). Fixed wing aircraft, helicopters, or rangeland drills are normally used for seeding. Aerial seeding treats large areas on steeper slopes or where tree densities are high. Drill seeding is used in open areas (Thompson and others 2006). Aerial seeding followed by chaining after fire significantly increased seeded grass cover and decreased cheatgrass cover compared to seeding alone (Ott and others 2003).

Historically, introduced species seed mixes were used to control soil erosion and forage production. In recent years, there has been more interest in using native seed mixes to increase species diversity and restore ecosystems (Richards and others 1998). Successful establishment of native grasses and forbs from different seed mixes has been demonstrated in several recent studies (Ott and others 2003; Thompson 2006; Waldron and others 2005).

Herbicides

Herbicides to control encroaching pinyon and juniper trees are another alternative to reduce tree cover. Basal spraying of herbicides allows for highly selective application with little effect on non-target species. Tebuthiuron (Spike® 80W) and picloram (Tordon® 22K) are commonly used herbicides in these systems. Parker and others (1995) tested the two chemicals' efficacies in controlling pinyon and juniper trees using basal application under different concentrations. Control was best for trees less than 1.8 m (6 ft) in height, with picloram killing over 90 percent of the sprouts and seedlings. Tebuthiuron, a slower acting herbicide, killed 30 to 60 percent of the sprouts and seedlings after 9 months, but results were expected to improve over time. Mortality of trees taller than 1.8 m (6 ft) was between 10 and 30 percent for picloram and 5 and 10 percent for tebuthiuron (Parker and others 1995). Johnsen (1986) states that individual tree application is best suited for newly invaded sites with fewer than 500 trees/ha (200 trees/acre) under 1.8 m (6 ft) tall. The longevity of these treatments will depend on the number and age class of the trees removed. Concentrating only on the older trees and leaving many of the younger trees will reduce the longevity of the treatment.

Broadcast application of tebuthiuron and picloram produce more variable results. One-seed juniper (*Juniperus monosperma*) and Rocky Mountain juniper (*J. scopulorum*) are often the most difficult to control (McDaniel and WhiteTrifaro 1986). Johnsen (1986) reported that herbicides readily killed trees on the ridges, but not on areas of deep soils or bottom land. Trees along the ridges are often old growth pockets of pinyon and juniper that generally should not be a target for removal. Areas invaded by pinyon and juniper, where herbicides are not as effective, are also places where fire would have historically limited their establishment. These are the areas often needing treatment. Concern over killing non-target species, with potentially limited mortality of pinyon and juniper, makes this treatment less desirable than individual tree application.

Summary

Management goals that deal with woodland expansion need to account for the landscape variability in community composition and disturbance regimes (Miller and Tausch 2001). Vegetation treatments should also focus on the source of woodland changes. In other words, they should focus on areas of the woodlands that represent expansion beyond the pre-settlement distributions. Within these areas, the focus should then be on woodland sites that have only recently transitioned away from FRCC 1. There are areas of recent tree expansion with vegetation attributes that indicate they may still be returned relatively easily to FRCC 1. These communities include the remaining sagebrush-dominated areas, sagebrush areas still in early Phase I (fig. 5), and the areas of old growth woodlands that are present within the landscape matrix. The sagebrush ecosystems and their associated communities, successional classes, and distributions need to be determined on a landscape basis for each management area.

There is, however, another management reality. With at least two-thirds of the woodland area in the Great Basin (and probably in southern Utah as well) representing expansion woodlands, millions of acres are now involved. Even under the best of conditions, only a minority of such a large area will be successfully treated before a wildfire occurs. For the remaining areas that are being increasingly dominated by trees that will burn before treatment is possible, we need to determine restoration/rehabilitation needs and possibilities following wildfire, particularly when cheatgrass or other exotics are present. Additional research is needed to help with the development of these procedures.

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