MECHANICAL TREATMENT OF PINYON-JUNIPER AND SAGEBRUSH SYSTEMS IN THE INTERMOUNTAIN WEST: A REVIEW OF THE LITERATURE

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Abstract

This literature review summarizes the ecological importance and management concerns for both piñon (*Pinus spp.*) and juniper (*Juniperus spp.*) systems, as well as big sagebrush (*Artemisia tridentata*) dominated systems, and reviews the literature detailing what have become accepted practices for treatment and management of these systems: mechanical treatments. The literature review is organized by the various goals that are often given as the reason for prescribed mechanical treatments. These include using mechanical treatments to enhance wildlife habitat, to use as a tool for fuels management, to restore expected age class and structure of these systems, to restore watershed productivity and provide for general ecological restoration, and to respond to encroachment of woody plants into previously non-wooded areas. Based on this review of the literature, there is a preponderance of evidence that while, in the short-term, forage production can often increase after mechanical treatments, over-all the negative ecological repercussions of mechanical treatment of both piñon-juniper and sagebrush often tend to outweigh positive ecological benefits, when looking at these systems from the standpoint of ecosystem health, function, and resiliency.

Introduction

This literature review summarizes the ecological importance and management concerns for both piñon (*Pinus spp.*) and juniper (*Juniperus spp.*) systems, as well as big sagebrush (*Artemisia tridentata*) dominated systems, and reviews the literature detailing what have become accepted practices for treatment and management of these systems: mechanical treatments. The term "mechanical treatments" used in this literature review refers to all treatments implemented in order to remove vegetation, flatten or "control" shrubs or trees or otherwise apply management which involves machines crossing the land. This includes drill seeding, and using tractors or other machinery to apply seeds or herbicides or other chemicals to the landscape (Miller et al 2005). As one of the goals for this review is to be relevant to Utah, we note whenever references to studies conducted in Utah are used.

Both single leaf piñon (*Pinus monophylla*) and two needle piñon (*pinus edulis*), along with Juniper, *(Juniperus osteosperma*), are found in the areas of Utah and intermountain West on which this literature review focuses. Typical sagebrush species in our focus area are likely to include basin big sagebrush (*Artemisia tridentata* subsp. *tridentata*), Wyoming big sagebrush (*Artemisia tridentata* subsp. *wyomingensis*), mountain big sagebrush (*Artemisia tridentata* subsp.*vaseyana*), and subalpine big sagebrush (*Artemisia tridentata* subsp. *spiciformis*).

Ecological importance of healthy, intact piñon-juniper and sagebrush stands

Contrary to what may be popular opinion, piñon-juniper systems can be extremely biodiverse communities. About 450 species of vascular plants occur in piñon-juniper woodland zones (Jacobs 1989).

Both piñon-juniper and western juniper woodlands have high diversities of vertebrate species. Piñon-juniper woodlands provide seasonal to year-long habitat for over 150 vertebrate species (Buckman and Wolters 1987), many of which decline in abundance with reductions in woodlands (Belsky 1996). Both elk and mule deer are year-round residents in piñon-juniper habitat, both of which consume both leaves and fruit of piñons and junipers (Martin et al. 1961) Maser and Gashwiler (1975) attributed the higher diversity of bird species in juniper woodlands to high structural diversity, large numbers of sites for perching, singing, nesting, and drumming, and plentiful berries and high insect diversity for food. They attributed high mammal diversity in the same communities to the presence of hollow trunks, shade, thermal cover, and foliage and berries for food.

Four species of corvid birds have mutually beneficial relationships with piñon pine and mixed piñon-juniper forests: Clark's nutcracker, Steller's jay, scrub jay and piñon jay. These nutcrackers and jays are the primary agents of dispersal of piñon pine seeds, and piñon pine seeds provide a large portion of their diet. Scrub jays and piñon jays cash seeds in these forests and are responsible for the regeneration of the species. However, the older the trees, the more valuable they are for these bird species. Piñon pine trees may produce cones when 25 years of age but can only produce significant quantities each season after reaching 75-100 years old. In general, good seed crops occur every 4 to seven years. Balda (1987), in a discussion of the diversity of birds in piñon- juniper woodlands, reported that he knew of “no other plant community in North America in which the dominant plant species have coevolved and [formed] mutualistic relationships with animals.”

The U.S. Dept. of Agriculture plant guide for two needle piñon pine states, “Pinyon nuts are a preferred food for turkeys, pinyon jays, woodrats, bears, and other wildlife, and they are a common food for deer, particularly during harsh winters with deep snows. Pinyon-juniper woodlands provide habitat for a varied wildlife population, including mule deer, white-tailed deer, elk, desert cottontail, mountain cottontail, and wild turkey" (Nesom 2002).

Similarly, sagebrush ecosystems support a variety of other species. For example, seventeen native mammals consume sagebrush (Welch and Criddle 2003). This includes the pygmy rabbit, which has been long-known to require dense stands of sagebrush (e.g. Burak 2006, Crawford 2008), and whose food habits indicate it relies up to 99% on sagebrush during the winter months (Green and Flinders 1980). Over 100 species of birds which forage and nest in sagebrush communities have been listed (Braun et al. 1976 and references therein), including the Brewers sparrow which nests off the ground in the foliage of big sagebrush plants (Best 1970). Twenty-four species of lichens are associated with sagebrush (Rosentreder 1990). A wide variety of native plants are associated with sagebrush including 16 species of Indian paintbrushes (*Castilleja spp.*). Seventy two species of spiders, 18 species of beetles, 13 species of grasshoppers or katydids, 54 aphid species, and 32 species of midges are associated with sagebrush (Welch 2005). This diversity of insects hosts a large list of bird, mammal and reptile species in healthy sagebrush habitat. Moreover, because sagebrush taxa are generally the dominant vegetation over the vast areas they occupy and are ecologically influential on all other organisms in the region (Braun et al. 1976; Knick et al. 2003; Connelly et al. 2011), they satisfy the criteria for keystone species (Khanina 1998; Smirnova 1998).

Sagebrush offers many valuable resources for a variety of wildlife. Sagebrush is recognized as providing valuable thermal and security cover (Connelly et al. 2000 and references therein, Beck et al. 2012). Sagebrush taxa also contain high levels of protein and other nutrients (Welch and McArthur 1979, Kelsey et al. 1982, Wambolt 2004), and are highly digestible (Welch and Pederson 1981, Striby et al. 1987). Thus, sagebrush provides forage for many wild ungulates (Welch 2005 and references therein, Beck et al. 2012), especially during the winter months when sagebrush has a higher crude protein level and digestibility than most other shrubs and grasses (Peterson 1995). In fact, some species like sage-grouse, elk and mule deer are known to be dependent on sagebrush for a portion of the year, if not year-round (McArthur et al. 1988, Peterson 1995; Welch 2005). Additionally, the crowns of sagebrush plants tend to break up and weaken hard crusted snow on winter ranges, making it easier for big game to access understory plants for foraging (Peterson 1995).

Sagebrush has also been found to possess important qualities that contribute to it’s community’s soil properties and hydrological function. For example, Welch (2005) explains how big sagebrush can help create “islands of fertility” across the landscape, as big sagebrush, with its particularly deep roots, can extract minerals far deeper in the soil profile than grasses or forbs can, and effectively bring these minerals and nutrients to the soil surface for use by other plants. In fact, a number of studies show that big sagebrush is a “soil builder” in this way (Fairchild and Brotherson 1980, Doescher et al. 1984, Chambers 2001). The nutrient content—such as nitrogen, phosphorus, potassium, calcium, etc. — directly under the canopy of big sagebrush is higher than the nutrient content in the interspaces (Welch 2005). Richards and Caldwell (1987) found that big sagebrush has the capacity to draw water from deep, moist soil layers and at night redistribute water into the drier upper layers of the soil, where other plants can use it (Caldwell and Richards 1989). Big sagebrush also helps to promote the uniform accumulation of snow, delays its melting, and retards the development of ice sheets, thus benefitting deep soil water storage in a system (Hutchison 1965). Moreover sagebrush, through shading the soil beneath its canopy, can help "extend" water near the soil surface by 2 weeks versus interspaces between plants (Wight et al.1992). This, plus the reduction in solar radiation from shading, can prolong the period favorable for seedling establishment for perhaps as long as 28 days (Pierson and Wight. 1991, Wight et al.1992, Chambers 2001).

We chose to organize our review of the literature on mechanical treatment of both sagebrush and P-J types by means of some of the chief reasons given to perform mechanical treatment of either sagebrush or P-J habitat:

Using Sagebrush and/or piñon-juniper mechanical treatment as a tool to enhance wildlife habitat

Hypothesized assertions used to support sagebrush reduction in wildlife habitat are common and include theories such as improving wildlife habitat by reducing "decadence"[1] of sagebrush communities, increasing habitat diversity, and creating edge (e.g. Winward 1991; Olson and Whitson 2002; Lutz et al. 2003). However, Beck et al. (2012) recently conducted a thorough review to ascertain whether the scientific literature supported these assertions, and concluded that the available information supporting this view is speculative and that empirical data are lacking.

Numerous studies show that sagebrush dependent wildlife prefer sagebrush cover at higher percentages than the 10-20% that some managers claim is expected in conditions that represent pre-settlement (Grinnel et al 1930; Rasmussen and Griner 1938; Feist 1968; Best 1972; Winter and Best 1984; Petersen and Best 1986; Knick and Rotenberry 1995, Welch and Criddle 2003). And this in turn implies that higher canopy cover amounts are the normal condition. For example, Brewer sparrow and sage sparrow prefer 20-35% canopy cover (Welch and Criddle 2003). Sage grouse in the Strawberry Valley, UT have their highest nesting success in 50% sagebrush canopy cover (Rasmussen and Griner 1938). And Wallestad's (1975) research with sage-grouse concluded that 80% of the locations of feeding and loafing sites for strutting males occurred in sagebrush with a canopy coverage of 20-50%. Reviewing the literature on this topic, Welch and Criddle (2003) concluded that the 10% and 20% sagebrush canopy cover figures often cited as natural is unsupported by data and studies.

One claim often made is that piñon-juniper and sagebrush treatment will "release" diminished amounts of native perennial grasses and forbs from competition with the more dominant sagebrush and piñon-juniper, and lead to an increase in cover, productivity and frequency of native grasses and forbs. It is true that treated sites often show increases in herbaceous vegetation compared to pre-treatment conditions, though long-term studies indicate that often

[1] This is a term principally used by the grazing management community. Wildlife scientists generally do not use this term (Welch 2005).

this benefit is short-lived (as summarized in Peterson 1995). The literature is replete with examples of studies that have shown that removals of sagebrush have no measurable effect on grass and/or forb abundance (i.e. Blaisdell 1953, Peek et al. 1979, Clary et al. 1985, Wamboldt et al. 2001, Summers 2005, Stringham 2010, Davies et al. 2011) or that sites with sagebrush removal experienced reduced productivity and/or diversity of native grasses and forbs (i.e. Pechanec and Stewart 1944, West and Hassan 1985, Cook et al. 1994, Watts and Wambolt 1996, Wambolt et al. 2001).

To discover to what degree the State of Utah's Watershed Restoration Initiative (UWRI) has been successful with meeting goals of more herbaceous cover following mechanical sagebrush treatments within Utah, we reviewed all post-treatment monitoring reports for mechanical treatments in sagebrush contained in the 2009 state-wide UWRI report (UDWR 2009). Most monitoring reports featured within this document monitored treatments that were conducted somewhere between 2004 and 2006, and all had data for at least 2 years post treatment. Most projects were harrow projects, with seeding. We found that while overall cover of both forbs and grasses tended to increase after these treatments (though cheatgrass was rarely separated out of the cover totals), forb frequency decreased as often as it increased post-treatment, and the frequency of grasses was equally likely to increase, have no observable change, or decrease post-treatment (UDWR 2009).

Many researchers have broadly challenged the long-standing claim that reduction of sagebrush will lead to increased herbaceous production (Peek et al. 1979, Anderson and Holte 1981, Wambolt and Payne 1986, Connelly and Braun 1997, Wambolt et al. 2001, Welch 2005, Beck et al. 2012). In cases where herbaceous production has increased after sagebrush treatments, the causal factors may be difficult to assess because post-treatment grazing deferment followed by changes in grazing management and other improvements routinely accompany sagebrush treatments (Beck et al. 2012). Livestock grazing is the principal management practice and influencing factor that affects grass cover and height (e.g. Rickard et al. 1975, Crawford et al. 1992).

Studies that have investigated changes in herbaceous understory following mechanical piñon-juniper treatment have also found that the treatments do not always deliver the promised result of increased cover of grasses and forbs (Evans 1988, Belsky 1996). This includes a study of a P-J cabling treatment in New Mexico that found that shrubs can respond more quickly than herbaceous species to the removal of trees and release of resources, further reducing grass and forb biomass and cover at the site to the point where grass and forb cover was greater on the control plots (Rippel et al. 1983); a study of a P-J removal treatment in which treatment plots had much lower grass cover than control plots (Wilcox 1994); a study in eastern Nevada that studied long-term changes in vegetation cover many decades after past P-J treatments which reported that at all treatment sites cover of herbaceous species decreased and cover of woody species increased (Bristow 2010); and a study in southern Utah that found that the percentage of forb/grass cover in P-J treatment areas was less than half of the percentage of cover in reference areas (Frey 2010).

Moreover, on both sagebrush and piñon-juniper sites, while the herbaceous community may tend to increase following treatment, this increase can be comprised not of desired native forbs and perennial grasses that were historically present at those sites, but of less desirable species such as rabbitbrush, snakeweed, horsebrush, halogeton, cheatgrass, and other weedy annual grasses and forbs (e.g. Pechanec et al. 1965, Graham and Sisk 2002, Welch and Criddle 2003, Ross et al. 2012). As summarized by Pechanec et al. (1965), "Care must be taken in sagebrush control work to avoid exchanging one problem for a more difficult one."

Where sagebrush communities are identified as habitat for sage grouse, currently a candidate species under consideration for federal listing under the Endangered Species Act, additional concerns have been raised by many sage-grouse biologists regarding any reduction in the amount of sagebrush canopy. 20-38% percent sagebrush canopy cover was cited for most nesting sites in the most widely used management guidelines for sage grouse (Connelly et al. 2000), so many sage-grouse biologists do not recommend sagebrush canopy reduction to cover less than 20% in or within two miles of breeding, nesting or brood areas. Good winter habitat for sage grouse often has higher canopy cover amounts (Connely et al. 2000). Thus, vegetation treatments for sagebrush cover less than 35% is not recommended for winter habitat for sage-grouse (Connelly et al. 2000).

Mechanical treatments have been suggested as a tool to improve habitat for greater sage-grouse. Yet the U.S Fish & Wildlife Service's recent decision that the species was warranted (but precluded) for listing identified habitat loss and fragmentation, including that caused by vegetation treatments, as a key reason for sage-grouse declines, and also indicated that treatments may not be beneficial to sage-grouse and that the rationale for conducting them needs further scrutiny (USFWS 2010).

Mowing is one of the mechanical treatments that some suggest to improve sage-grouse habitat, because by reducing cover of sagebrush, this may allow perennial grasses and forbs (very important for sage-grouse nesting and cover habitat) to increase in cover and density once there is more room for germinatation in shrub interspaces post treatment. However, studies by Hess (2011) and Hess and Beck (2010, 2012) in Wyoming (in addition to studies referenced above) indicate that mowing does not lead to an increase in critical sage-grouse early brood-rearing needs such as forb abundance, weights or abundance of beetles and grasshoppers, or perennial grass cover or height[2]. Similarly, forb nutritional content was not enhanced by the treatment (Hess and Beck, 2010, 2012). Hess's and Beck's work in Wyoming has mirrored others' work in other types of sagebrush communities. For example, Christiansen's (1988) study in mountain big sagebrush communities in Wyoming found that mowed sites had lower beetle species diversity and richness following treatment. Scoggan and Brusven (1973) also found that grasshopper species richness was reduced after big sagebrush control.

Most researchers who have investigated the impacts of big sagebrush control or removal projects on sage-grouse have concluded that these treatments can have serious negative impacts on sage grouse (Benson et al. 1991; Braun et al. 1977; Carr 1968; Connelly et al. 2000; Fischer et al. 1996; Klebenow 1970; Kufeld 1968; Martin 1970; Peterson 1995; Pyrah 1972; Swenson et al. 1987; Wallestad 1975). In particular, studies suggest that removal of big sagebrush in sage-

[2] Hess and Beck (2012), in all cases over both years, found that perennial grass canopy cover and height at burned sites, mowed sites and control sites surpassed minimum guidelines for sage-grouse breeding habitat. However, the lack of differences they found in perennial grass canopy cover and height between mowed and paired reference sites suggests that mowing is not effective in increasing perennial grass structure in Wyoming big sagebrush communities.

grouse winter or breeding habitats have a negative effect on sage-grouse (as summarized in Beck et al. 2012), though some studies suggest little measurable – but not positive - effects (Gates 1983, Martin 1990, Robertson 1991). On Parker Mountain in south-central Utah, Dahlgren et al. (2006) reported a decline in sage-grouse pellets 20 m from the edge and almost no pellets >40 m from the edge of intact patches of sagebrush within tebuthiuron, Dixie harrow, and Lawson aerator treatments. The same study also reported that sage-grouse use of mechanical sites was not different from non-treated reference sites (Dahlgren and Mesmer 2006). In North Park, Colorado, Braun and Beck (1996) found that after over 28% of the 3,500+ ha study area was plowed and chemically sprayed between 1965 and 1970, the mean 5-year average of attending males on leks dropped 25% from 765 (1961-1965) to 575 (1971-1975). And in a study of both mechanical and chemical treatments within a 0.5 km radius around four leks in central Montana, the resulting loss of 10 to 30% of suitable sage-grouse habitat within a 1.5 km radius around those leks led to a 65% drop in males attending those leks (Wallestead 1975). A study by Hollaran found that sage-grouse populations in Wyoming began declining with as little as 3.4% of the sagebrush removed (as reported in Molvar 2011). Based on much of the research summarized above, Autenrieth (1996) summarized the impacts of big sagebrush control on sage-grouse and recommended that sagebrush control work should never be conducted within 2 miles of a lek, nor in known sage-grouse wintering concentration areas, nor along streams, meadows, or secondary drainages, both dry and intermittent.

Based on many cases where sagebrush treatment has not shown evidence of benefiting sage-grouse, numerous sage grouse biologists have recommended against sagebrush treatment in areas important to sage-grouse. This includes Connelly et al. (2000) who recommended treating no more than 20% of breeding habitat in Wyoming big sagebrush every 30 years; Beck et al. (2012, summarized elsewhere in this review); Fischer et al. (1996) who noted that their findings did not support the idea that killing big sagebrush enhanced sage grouse brood-rearing habitat; Woodward (2006) who cautioned against removing sagebrush stands even if the herbaceous community is depleted and not ideal for sage-grouse, and Hess and Beck (2010), who stated that, "If sagebrush community characteristics in untreated communities do not meet the minimum Connelly et al. (2000) guidelines, managers should reconsider treatments in those areas, and instead consider other practices such as improved grazing management…"

There are also examples of piñon-juniper removal employed for the purpose of improving occupied sage-grouse habitat, yet post-treatment monitoring did not indicate that conditions improved substantially or that sage-grouse preferred the treated areas. For example, a population of sage-grouse outside the town of Alton in southern Utah was monitored both two and three years after 2005 P-J removal treatments in the area (the goal of which was to improve sage grouse habitat by emulating breeding habitat). Researchers (Frey and Heaton 2009) discovered that during the period of 2005 to 2007, sage-grouse preferred the intact sagebrush stands to the treated areas. They also found that the percentage of forbs and grass cover in treatment areas was less than half of the percentage of cover in breeding habitat reference areas (Frey 2010). In addition the average forb/grass height was also twice as high in breeding reference areas than in the treatment areas (Frey 2010). Frey's team also is monitoring sage-grouse in the Bald Hills in southern Utah. There they are finding that sage-grouse are actually nesting under juniper trees in some cases (personal communication Cheyenne Burnet**,** Utah State University) suggesting that sage-grouse in the extreme southern portion of the range may be utilizing less traditional nesting sites, and perhaps P-J treatment is not necessary to improve and expand sage-grouse habitat in these locales.

While sage-grouse responses to sagebrush treatment are some of the more common studies that are conducted in regards to wildlife and sagebrush control, other studies have investigated responses of other species to treatment. For example, during a study in Colorado in which over 120 flocks of birds (representing over 3,000 birds) were observed during two winters, only four of these flocks were found in altered (chemically sprayed, plowed, burned, or seeded) mountain big sagebrush habitats, although over 32% of the study area had been treated (Beck 1977). And in a study comparing mechanically treated and burned sagebrush sites to adjacent, non-treated controls at the Curlew National Grasslands in Idaho, Welch (2002) found that total numbers and species richness of birds was significantly greater in controls compared to the treated sites.

A number of studies have investigated pygmy rabbit responses to sagebrush treatment. In her study of pygmy rabbit response to sagebrush removal in northern Utah, Wilson (2010) found that rabbits that approached treatment edges were less likely to enter treatments than expected by chance. In a similar study conducted on Parker Mountain, south-central Utah, Lee (2008) found significantly greater counts of pygmy rabbit pellets in areas with continuous sagebrush compared to sagebrush strips and islands within treated areas. In a study investigating pygmy rabbit abundance and habitat use in mechanically treated sagebrush sites in Grass Valley, Utah, Pierce et al. (2011) found that the proportion of active burrows and the relative abundance of pygmy rabbits were reduced near habitat edge. This reduction was associated with an increase in terrestrial predators and competitors near habitat edge. Flinders et al. (2005), working in the same study area in Grass Valley, Utah, stated that, “active burrows encountered on re-walked surveys of BLM treatment surveys were only found in sagebrush treatment mosaics connected to remaining stands of sagebrush or areas where swaths of removal were much smaller and distances between one treatment to the next were minimal.” Due to their similar findings of negative consequences of sagebrush removal on their pygmy rabbit study populations, most of the researchers above reached the same conclusion, which is mechanical treatments should be avoided in occupied pygmy rabbit habitat. Flinders et al. (2006) summed it up in this way: “we caution against traditional habitat treatment aimed at reduction of sagebrush cover (e.g. dixie harrow, burning, application of herbicide, etc.) in areas where pygmy rabbit burrows are found.”

In closing their recent literature review on the impacts of big sagebrush treatment on wildlife habitat, Beck et al (2012) conclude that, “the preponderance of available literature indicates that habitat management programs for sagebrush steppe that emphasize treating sagebrush (i.e., sagebrush removal) are clearly not supported.” They go on to caution that recommendations by habitat managers to remove large swaths of sagebrush for the purported purpose of benefitting wildlife are recommendations that do “not appear to be supported by the literature and, given the reliance of so many species on sagebrush, could lead to declines of these species. Relying on dogmatic beliefs rather than the best available data to support management programs is premature at best for some species and irresponsible at worst for sage-grouse and possibly other species, especially given the stressors currently affecting sagebrush steppe habitats.” They close with: “Given the overall lack of evidence documenting positive population responses of sage-grouse, pronghorn, mule deer, or elk to treatments in Wyoming big sagebrush, we urge land managers to refrain from these treatments until information is available that clearly documents appropriate treatments and the conditions, including appropriate temporal and spatial scales, under which those treatments are expected to impact these wildlife species." Beck et al.'s (2012) recommendation echoes that of Welch and Criddle (2003), who concluded that as more and more acres of the sagebrush ecosystem are converted into human development, stands of annual weeds, and seeded perennial exotic grasses, the likelihood that treatments may place sagebrush obligates' survival at risk questions the advisability of any burning, mechanical, or chemical treatment of sagebrush habitat.

Similarly, studies looking at wildlife use of treated piñon-juniper have also shown that the treatments do not always deliver on the promises of improved wildlife habitat either. For example, Sedgwick and Ryder (1987) found that while the chaining of trees from a piñon - juniper woodland in Colorado increased herbaceous production, it significantly reduced site utilization by birds. As a result, avifauna diversity was higher in woodlands than in chained sites, with the foliage/timber searching guild, aerial foraging guild, and cavity nesting guild most affected by treatment. Moreover, woodland clearance has generally shown few effects on population sizes of big-game species such as deer and elk (Terrell and Spillett 1975, Skousen et al. 1989 – a study in central Utah, Belsky 1996). One reason it is theorized that deer will not tend to utilize cleared P-J sites above normal use levels for the area is because of their hesitancy to expose themselves in large open areas (Short et al. 1977, Lanner 1981).

Whether or not to use various treatment methods to encourage increased levels of vigorous and succulent spring forage is a matter of some debate. While treatments have been shown to increase levels of deer and elk browse and forage in the spring (Gottfried et al. 1995 and references therein, Brockway et al. 2002, Owen et al. 2009), the question is, is increased spring forage more important than the winter browse and thermal protection provided by the partial or intact overstory offered by mature piñon-juniper habitats? For this reason, the value of piñon-juniper conversion as a tool in big game habitat management still remains a debated topic (e.g. Gottfried et al. 1995).

Using Sagebrush and/or piñon-juniper mechanical treatment as a tool for fuels management

***Historical piñon-juniper fire cycles*.** The estimation of the natural, historic fire intervals of piñon-juniper communities in the Great Basin and Colorado Plateau are less of a published topic than that of sagebrush fire intervals; and in fact is a matter of some debate. A recent, comprehensive review by 15 researchers (Romme, et al 2009) identifies three types of piñon-juniper ecosystems: persistent piñon-juniper woodlands, piñon-juniper savannahs and wooded shrublands. They stress that it is important to consider these three P-J types separately when the topic of fire fires intervals are discussed and debated.

These three piñon-juniper types have distinct differences in understory composition and length of fire rotations. For instance, persistent piñon-juniper woodlands, which can be found throughout much of the Colorado Plateau and Great Basin, range from sparse stands of scattered, small trees growing on poor substrates to relatively dense stands of large trees on more productive sites, and while exhibiting variable cover of shrubs, sub-shrubs, forbs, and grasses, understory is often sparse, with significant areas of bare ground (Romme, et al 2009). Fire is inherently infrequent in persistent piñon-juniper woodlands, and thus favorable for tree growth within P-J patches and nearby sites (i.e. wooded shrublands). Romme et al (2009) describe how many piñon and juniper woodlands exhibit little to no evidence that they ever sustained widespread fires during the period that trees have been alive in the stand, and that high severity, or "crown" fire was likely the dominant type of fire in these woodlands in both historical and modern eras. Over time, dense woodland conditions become highly flammable (i.e., fuel accumulation over decades or centuries) regardless of fine fuel conditions; the probability of ignition and duration of the fire season define the actual fire return intervals in persistent P-J woodlands, in which fire is typically stand-replacing. Best estimates from Romme et al. (2009) on historical fire rotations for persistent woodlands vary from 400-600 years, based on best available fire scar data across the West.

Historically, persistent piñon-juniper woodlands likely had somewhat longer fire return intervals than piñon-juniper savannahs, which are often found further south and east than persistent P-J woodlands, in places such as New Mexico and Arizona. Romme et al. (2009) describe these forests as having low to moderate density and cover of piñon or juniper or both, with a well-developed understory of nearly continuous grass (with forb) cover. Shrubs may be present but are usually only a minor component. However, even though P-J savannahs may have (or historically had) somewhat shorter fire cycles than persistent woodlands, the fire interval is still quite long on these P-J types as well, and Romme et al. (2009) stress that spreading, low-intensity, surface fires had a very limited role in molding stand structure and dynamics of many or most piñon and juniper woodlands in the historical landscape. Historical fires in all P-J types generally did not "thin from below", i.e., they did not kill predominantly small trees. Instead, the dominant fire effect was to kill most or all trees and to top-kill most or all shrubs within the burned area, regardless of tree or shrub size. This statement is also true of most ecologically significant fires today (Romme et al. 2009). The fact that nearly all species of piñon and juniper are relatively fire intolerant also support the above statements of Romme et al. Romme et al. (2009) go on to explain that in many piñon-juniper woodlands, stand dynamics are driven more by climatic fluctuation, insects, and disease than by fire.

Prior to Romme et al. (2009), Baker and Shinneman (2004) also conducted a review of the piñon-juniper fire cycle literature, somewhat similar to that of Romme et al. (2009), but that was more systematic in its approach. Baker and Shinneman (2004) came to very similar conclusions as Romme et al. (2009), namely that, based on best available historical fire scar data, low-severity surface fires were likely not a common type of fire in P-J woodlands, other than moderately reliable evidence that spreading low-severity surface fires occur in higher elevation P-J ecotones with ponderosa pine. Baker and Shinneman (2004) also came to the same conclusion as Romme et al. (2009) that while it seems logical that fires did spread in juniper savannas, data to support this idea are surprisingly meager. Furthermore, Baker and Shinneman point out that no reliable data suggest that low-severity surface fires would have consistently lowered the density of trees in moderate-density woodlands, even with a sagebrush or grassy understory. Another area of agreement between the two reviews is that high severity fires historically were the norm for most P-J woodlands, and indeed could still be considered the natural course today. For example, Baker and Shinneman (2004) describe how, since EuroAmerican settlement, 126 observed or reconstructed wildfires in piñon–juniper woodlands in the West have been reported in the literature, and of these, two were low-severity surface fires, three were possibly mixed severity, and 121 were high severity fires. In addition, Baker and

Shinneman conclude that there is no basis for concluding that high-severity fires have or have not increased in piñon–juniper woodlands since EuroAmerican settlement. Therefore, arguments that piñon–juniper woodlands have become more dense due to fire suppression are non-sensical (Baker and Shinneman 2004, Romme et al. 2009). Based on their review, Baker and Shinneman (2004) caution land managers against falling back on theories and blanket statements that P-J woodlands are supposed to burn often (sometimes cited as frequently as every 13 – 35 years, e.g. Frost 1998, Brown 2000, and Hardy et al. 2000), and stress that these postulations of fire frequency are not supported by the scientific evidence. They go on to state that there is thus insufficient scientific basis for land managers to apply uniform fire and structural treatments in piñon–juniper woodlands.

***Historical sagebrush fire cycles*.** Sagebrush treatments are also sometimes justified as a means to reestablish the expected fire interval. Range managers often argue that sagebrush is dependent on regular thinning by fire in intervals from 10-40 years. Welch and Criddle (2003) comprehensively reviewed studies that discussed fire and sagebrush and whether fire is a required component of a healthy sagebrush community. A 31 year study demonstrated that a big sagebrush ecosystem can maintain itself without the occurrence of fire (Lommasson 1948). There are a number of studies and reports that indicate that the natural fire interval in sagebrush-grass communities and big sagebrush communities (*A.t. wyomingensis*, *A.t. tridentata*) are likely between 50 to 125 years (Houston 1973, Wright and Bailey 1982, Whisennant 1989, Welch and Criddle 2003, Welch 2005), yet Baker (2011) postulates that fire rotations in Wyoming big sagebrush range in frequency from 200 yr to 350 yr and are dependent on climate, topography, plant composition, and ecological site characteristics. The fire interval in mountain big sagebrush is somewhat less frequent than that of Wyoming big sagebrush (Welch 2005 and references therein).

Many biological and ecological characteristics of sagebrush suggest that it did not evolve in an environment with frequent fires: (1) sagebrush taxa have a life expectancy of 75 to over 200 years (Ferguson and Humphrey 1959; Ferguson 1964, Welch 2005), (2) sagebrush has highly flammable bark, (3) sagebrush has a low growth form that is susceptible to crown fires, (4) sagebrush is non-sprouting, and must germinate from seed (Pechanec et al. 1965, Tisdale and Hironaka 1981), (5) sagebrush lacks a strong seed bank in the soil, and (6) seeds lack fire resistance adaptations (all of above summarized and referenced in Welch and Criddle 2003, Welch 2005).

Another avenue of research that can help shed light on the length of sagebrush fire cycles is the length of time it takes for sagebrush to recover after fire, or a large control or removal event. The literature reports that post-burn recovery of Wyoming big sagebrush is anywhere from 25 to well over 100 years (Watts and Wambolt 1996, Connolly et al. 2000, Wambolt et al. 2001, Welch 2005, Baker 2006, Baker 2011).

Whether to conduct piñon-juniper and sagebrush fuels treatments, or not? Because of both actual and perceived increased P-J canopy densities in some locales in recent times, one of the stated goals of some piñon-juniper treatments is to reduce tree density so as to make it more unlikely that large, crown fires will occur[3], especially if those fires might carry to populated areas or structures.

One of the most pressing problems facing both piñon-juniper, and especially sagebrush rangelands, is cheatgrass encroachment. Cheatgrass has only limited grazing use by livestock and tends to quickly crowd out other, more desirable forage grasses which do not demonstrate the flammability, fire recovery, early germination and rapid growth of cheatgrass. Studies on the effects of fire on native and invasive grasses have shown that repeated burning will tend to deplete perennial native grasses and allow annual grasses, primarily cheatgrass, to increase its coverage dramatically. Once communities are depleted of their perennial grass cover, as in a wildfire, a secondary succession begins which eventually results not only in the dominance, and even monocultures, of cheatgrass within only a few years, but also speeds up the natural fire interval of rangelands (Nevada Division of Water Planning. 2000, Reisner 2010, Baker 2011).

There are many studies that indicate that when sagebrush is mechanically treated and if cheatgrass and/or other exotic annuals are present in the system before treatment, then cover of these species will increase post treatment, thus increasing the rate of invasion by cheatgrass in the area. These include; Wisdom and Chambers (2009, literature summary); Prevey et al. (2010) who found 3–4 times more exotic herbs in plots where sagebrush was removed than in undisturbed habitats; and Davies et al. (2011, 2012) who found increased cover, density and production of annual forbs and exotic annual grasses in mowed sites compared to unmowed Wyoming big sagebrush sites (this leading them to conclude that mowing as a stand-alone treatment will not restore herbaceous understories in degraded Wyoming big sagebrush communities). In their recent literature review of the impacts of big sagebrush treatment on wildlife habitat, Beck et al. (2012) stated that "the recognition that that sagebrush is essential to maintaining native plants and limiting the invasion of exotic plants in sagebrush communities…suggests that future treatments should be limited to those that do not eliminate or greatly reduce sagebrush."

There are likewise many studies that report when piñon-juniper is mechanically treated and if cheatgrass and/or other exotic annuals are present in the system before treatment, then cover of these species will increase post treatment, posing similar problems with future fire regimes in these systems. These include: Davis et al. (1990) who reported significant increases in weedy annuals on a chained piñon-juniper site in central Utah; Evans and Young (1985, 1987) who found that a site in California from which western juniper had been cleared was colonized and eventually dominated by cheatgrass and medusahead; Miller et al. (in press) who found that attempted restoration of piñon-juniper habitat after a fire (using seeding then chaining) resulted in increased cover of exotic annual plants rather than seeded perennials; Owen et al. (2009) who observed increases in cheatgrass following both lop & scatter/pile burn and mastication treatments in P-J on their study sites on the San Juan N.F in southwestern Colorado; Ross et al. (2012) who found on their study site on Shay mesa, southeastern Utah, that cheatgrass was not present on control sites but comprised more than 18% of total understory cover on the lop and scatter/pile burn treatment in the PJ woodland and between 11% and 18% of total understory cover on the PJ mastication treatment; and Vaitkus and Eddleman (1987) who found that although herbaceous production doubled after the removal of western juniper in eastern Oregon,

[3] This, despite the point that these sort of fires, though very infrequent, were likely to have been the normal condition, especially in persistent P-J woodlands (Baker and Shinneman 2004, Romme et al. 2009).

much of this increase came from annual forbs such as fireweed and led the authors to conclude that “an increase in herbage production after tree removal does not necessarily result in an improvement in range condition.”

Using Sagebrush and/or piñon-juniper mechanical treatment to restore the expected age classes and stand structure of woody plants

Another theory used to justify mechanical treatments, especially in sagebrush tracts, is that removing large swaths of sagebrush and seeding with a diverse seed mix that also includes sagebrush will eventually lead to more diversity of age classes. This is another issue that Beck et al. (2012) addressed in their review of literature on treatments in Wyoming big sagebrush, and they reported that large-scale treatments, including mechanical treatments, are likely to result in stands of sagebrush that are more even-aged than sagebrush in undisturbed stands. Beck et al. go on to report that, in the absence of disturbance, sagebrush communities are characterized by a shrub stratum composed of diverse age classes (citing Passey and Hugie 1962, Daubenmire 1975). Hess and Beck’s (2010) studies in the Bighorn Basin in Wyoming echoed the sentiment of Beck et al; they found that, both 9 and 19 years after mowing treatments in big sagebrush habitat, these treatments do not result in improved habitat conditions for sage grouse nesting and early brood rearing habitat, and this included no positive changes in structural aspects of the treatment areas for sage grouse.

The review by Beck et al. (2012) explores how long it takes sagebrush to recover after large-scale, stand-replacing disturbances. The authors report that it will be decades after disturbance before a mature community reestablishes a natural turnover with maximum structural and compositional diversity and value as habitat (citing Lommasson 1948; Ferguson 1964; Nelle et al. 2000; Beck et al. 2009), and that aging and death of individual sagebrush plants are natural processes and offer an opportunity for the community to achieve maximum diversity through an optimal vegetative pattern for wildlife habitat (citing Lommasson 1948; Passey and Hugie 1962).

One argument for sagebrush treatments in habitat preferred by deer and pronghorn is that as plants age they lose their nutritional value. Yet in their review of the big sagebrush treatment literature, Beck et al. (2012) reported that terpene levels (which in high quantities can significantly degrade the forage value of the plant) in basin, mountain, and Wyoming big sagebrush are not affected by plant age (citing Kelsey et al. 1983), suggesting that treatments designed to manipulate the age structure of Wyoming big sagebrush will not reduce terpene levels. In addition, both Peterson in his review of the literature (1995) and Wamboldt (2004) reported that there is no relationship between crude protein content and age of mountain big sagebrush (both authors), or basin or Wyoming big sagebrush (Wambolt 2004). This finding led Wambolt (2004) to conclude that crude protein levels will not increase by manipulating vegetative cover to favor early successional stages with many young plants. In addition, young big sagebrush plants, which are often all that can be found a few years after a sagebrush removal treatment, have stronger chemical resistances to herbivores (Shiojiri and Karban 2006, Karban et al. 2006). Thus, younger stands of sagebrush resulting from sagebrush treatment may be less palatable or provide less forage than older stands, further suggesting that both structural recovery and ontogeny are necessary for valuable wildlife habitat (Beck et al. 2012).

Using sagebrush and/or piñon-juniper mechanical treatment as a means for ecological restoration and watershed productivity

Miller et al (2005) reports that western juniper's "rapid expansion into neighboring communities the past 130 years has caused considerable concern because of increased soil erosion; potential reduced stream flows; reduced forage production; altered wildlife habitat; changes in plant community composition, structure, and biodiversity; and the replacement of mesic and semi-arid plant communities with woodlands."

When does sagebrush, piñon or juniper removal constitute ecological restoration? The Society for Ecological Restoration defines ecological restoration as "the process of assisting the recovery and management of ecological integrity. Ecological integrity includes a critical range of variability in biodiversity, ecological processes and structures, regional and historic context, and sustainable cultural practices" (SER 2004).

Piñon-juniper forests have been classified into three general types: persistent woodlands, piñon-juniper savannas, and wooded shrublands (Romme et al 2009). Persistent woodlands are characterized by soils, climate and natural disturbance regime that favor a mix of piñon juniper. Piñon juniper savannas include soil, climate and disturbance patterns that favor a balance of trees and native grasses. Wooded shrublands tend to have the soil, climate, and natural disturbance patterns that favor shrubs as a major part of piñon juniper forests. There is a great diversity of species within and between these forest types that describe the ecological characteristics of a site at its potential. Similarly, there is a diversity of sagebrush communities. Thus one standard template on ecological restoration is inappropriate.

Regardless of how exactly one defines ecological restoration, there are many indications from the literature that mechanical piñon-juniper and/or sagebrush treatment, especially if followed by mechanical drill seeding, can fail to meet goals of "ecological restoration and watershed health and productivity." For example the machinery involved with mechanical treatments can be extremely destructive to biological crusts (Gifford et al. 1970, Loope and Gifford 1972, Wilcox 1994, Belnap and Gillette 1998, Belnap and Eldridge 2001). The recovery time for the lichen component of crusts has been estimated at about 45 years (Belnap 1993). At this time the crusts may appear to have regenerated to the untrained eye. However, careful observation will reveal that the 45 year-old crusts will not have recovered their moss component, which will take an additional 200 years to fully mature (Belnap and Gillette 1997). Studies done outside of Blanding, Utah have shown that chaining of P-J led to decreased infiltration rates at the study site, and part of the reason given for this decrease was destruction of biological crusts, resulting from mechanical disturbance associated with chaining activities (Gifford et al. 1970, Loope and Gifford 1972; Gifford 1973). In addition to losses of biological crusts due to mechanical treatments, any sagebrush or P-J mechanical clearing technique that actually uproots the plants leads to the greatest degree of soil disturbance, thus significantly adding to the risk of post-treatment soil erosion (Pyke 2011). With soil losses due to erosion following destructive activities on the soil surface, the soils themselves take 5,000 to 10,000 years to naturally re-form in arid regions such the Colorado Plateau (Webb 1983), so this can be considered an irreversible loss. And, in semi-arid climates of the west, some soil properties take even longer to accumulate, on the order of tens of thousands of years (Gottfried et al. 1995).

One of the problems experienced with mechanical treatments in piñon-juniper and sagebrush habitat is that they can lead to degraded ecosystem conditions. One example is mechanical drill seeding (sometimes followed by chaining to "rough up" the soil) or mechanical clearing of dead P-J after a fire, both of which can lead to significantly increased wind erosion with post-fire treatments. An example of this has recently played out with the post-fire mechanical recovery treatments employed after the large 2007 Milford Flat fire in central Utah (Miller et al. in press). In the case of Millford flats, Miller et al. measured sediment fluxes (a measure of wind erosion) on unburned plots, burned-untreated plots, and burned plots treated by chaining and drill seeding. The results showed that sediment fluxes at the burned and treated sites in the first year of the study were 41 times greater than the unburned sites, and 78 times greater than the burned, untreated sites. The second year of the study reported that sediment fluxes at the burned and treated sites were 23 times greater than the unburned sites, and 15 times greater than the burned, untreated sites. These kinds of intense dust storms can be very detrimental to adjacent intact vegetative communities that were spared by the fire, with impacts ranging from decreases in photosynthesis, respiration, and transpiration; penetration of phytotoxic gaseous pollutants, decreased productivity, and even, eventually alterations in community structure (Farmer 1992). Additionally, Miller et al. found that soil aggregate stability was significantly lower in the burned, treated plots than both the unburned plots, and the burned, untreated plots (which incidentally had nearly identical levels of aggregate stability). One of the conclusions reached by Miller et al. (in press) was that aggressive mechanical treatments right after wildfire to removal dead trees and attempt to seed perennial grasses are not necessarily "one size fits all," and one of the most important things to consider at every site are the intrinsic erodibility factors in the soil. Moreover, the authors raise the concern that, "despite the common and costly practice of post-fire rehabilitation in western United States, few monitoring studies have been conducted to examine the overall effectiveness of such treatments (citing GAO, 2003[4]), and virtually no studies have evaluated treatment effects on wind erosion."

There are many examples in the literature of cases where mechanical clearing of piñon-juniper has led to increases in erosion by both air and water. This includes studies outlined by Myrick (1971) that indicate that chaining P-J and burning slash followed by seeding will cause an increase in precipitation runoff for the first couple years following treatment; a study by Baker et al. (1971) with Utah juniper that reported that cabled and chemically treated juniper sites had peak erosional discharges of 2.0 and 1.3 times greater than their woodland control sites; a study by Gifford (1973) at two sites in southern Utah which found that in all cases runoff was 1.6 to 5 times greater on chained and drill seeded plots compared to the untreated woodland plots; and a study on the Cibola National Forest in New Mexico that found that sites where P-J was removed had greater soil loss due to erosion than the control sites (Brockway et al. 2002). Another measure of erosion potential for a site is soil aggregate stability; multiple studies have reported that mechanical treatments in P-J habitats have lower aggregate stability than pre-treatment conditions and/or non-treated control plots (e.g. Ross et al. 2012).

Another issue that fits into the theory of greater watershed productivity post piñon-juniper treatment is hypothesized increased water yield and increased ground water recharge as a result

[4] The U.S. General Accounting Office (2003) conducted an intensive analysis of emergency stabilization and rehabilitation projects in the U.S. Departments of Agriculture and Interior and found that neither department could report on the effectiveness of their projects.

of P-J clearance from a site (because of the claim that juniper in particular consumes more water through its roots compared to other vegetation at the site). Studies and reports by Collings and Myrick (1966), Clary et al. (1974), Schmidt (1987), Brown (1987), Wilcox (1994), Nesom (2002), and Ramirez et al. (2008) have found that this is simply not the case (also see review by Belsky 1996).

In her review of the literature on juniper expansion, Belsky (1996) lists the litany of complaints lodged against piñon-juniper and its expansion, ranging from "causing springs and small streams to dry up"; "endangering fish and aquatic life"; "increasing overland water flow and soil erosion"; "reducing the diversity of plants and wildlife"; and "reducing forage production for livestock and wildlife." Belsky goes on to report that scientific evidence for most of these beliefs, however, is lacking. She also explains why studies that show that junipers intercept precipitation and transpire water cannot be used to conclude that this "lost water" would have increased flows in streams and springs. Many researchers working in the southwest and Great Basin have failed to find lower intrinsic water infiltration rates or more erosion in P-J communities than in other communities (Gifford et al. 1970, Gifford 1973, Clary et al. 1974, Schmidt 1986, Heede 1987, Renard 1987, Evans 1988).

As covered earlier in this literature review, any kind of land treatment that clears the existing vegetation and disturbs the soil (so all mechanical treatments but also fire and chemical treatments) can result in increases in exotic annuals, especially cheatgrass, when these species are present in the system before treatment (see references above).

In their closing discussion of their study on the impacts of mowing to sage-grouse habitat in Wyoming, Hess and Beck (2010) summarize mechanical sagebrush treatment impacts on sagebrush ecosystem health and productivity. They close with a note of caution when using mowing treatments in big sagebrush habitat because of the higher percentage of bare ground found on mowed sites compared to both reference areas and burned sites, which can potentially lead to erosion (Sherman and Buckhouse 1984, Hofmann 1991), resources for exotic plant invasion (Burke and Grime 1996, Bergquist et al. 2007) and thus increased fire frequency (West 2000, Baker 2011) and lower ecosystem productivity (Watts 1998).

Using mechanical treatment to respond to encroachment of woody plants

Historic extent of piñon-juniper forests in the intermountain West. Encroachment of piñon-juniper forests is claimed to lead to negative impacts including loss of wildlife habitat, increased erosion, loss of herbaceous species, increase in conditions conducive to weed invasion, and decreases in water quantity and quality (Blackburn and Tueller, 1970, Burkhardt and Tisdale 1976, Soule and Knapp 1999, Wilcox and Davenport, Wall et al. 2001, 1995 Baker and Shinneman 2004, Rowland et al. 2008).

Today's management of juniper and piñon pine forests is often based on the assumption that piñon-juniper encroachment has occurred nearly everywhere in the Great Basin and intermountain West in the last 130 years. The Sagebrush Steppe Treatment Evaluation Project (SageSTEP) has assembled a number of studies that argue that there has been significant increases in both density and distribution of piñon and juniper woodlands during the past 130 years (Burkhardt and Tisdale 1969; Tausch et al. 1981; Tausch and Tueller 1990; Tausch and West 1995; Bunting et al. 1999; Bates et al. 2000; Miller et al. 2000; Roberts and Jones 2000;

Miller and Tausch 2001; Miller et al. 2005; Tausch and Hood 2007; Miller et al. 2008; Romme et al. 2009).

To fully understand the claim that there has been substantial expansion, or "encroachment", of piñon-juniper forests in the intermountain West, one needs to look further back in time and understand the pre-settlement history of piñon juniper forests, as well as the large-scale juniper and piñon forest removal that occurred 130-160 years ago. We, and other researchers, posit that much of the perceived "encroachment" of piñon-juniper woodland in the last 130 years, especially in the southern Great Basin and parts of the southwest, is actually a sign of recovery of these woodlands to their historic range, in response to past and excessive human logging (also see Sallach 1986, and Romme et al. 2009).

Ranching and farming by settlers in the early to mid 19th century relied on fences and buildings built from local wood sources. Prior to the advent wire fences (which was well into the 20th century), large quantities of wood were required to construct solid wood fences. Juniper fence posts, the preferred choice, can last 30 or more years, longer than any other species for untreated wood (Morrel et al. 1999). Beginning in the mid 19th century, particularly in the southern and central Great Basin, piñon and juniper wood was the preferred source for mine supports, lumber for buildings, fuel for smelting minerals, processing lime for mortar and plaster, and manufacturing bricks (Lanner 1981). Charcoal, mostly from piñon pine, and created either in earthen-covered pits or kilns, was the favored fuel throughout the southern Great Basin in the late 19th century, especially for ore smelting. Huge amounts of piñon pine were used in the kilns, for example between15 to 45 cords of piñon pine were used at a time in a charcoal-making process that took one to two weeks to complete (Lanner 1981). Produced in many local towns, charcoal was a commodity sold regionally, mostly to mines, and sold in the late 19th century for the incredibly low price of twenty one cents per bushel (Bartholomew 1996), or $5 in today's dollars.

Throughout the late 1800's across the Great Basin, mining towns consumed enormous quantities of local juniper and piñon pine. The Comstock mines in western Nevada consumed 18 million board feet of timber annually, much of it for mine timbers (Lanner 1981). Mining operations in Eureka, Nevada burned 17,850 bushels of charcoal daily. At the time, charcoal was the greatest mining expense, even more than labor. The arrival of the railroad in the west amplified economic activity in the Great Basin and Intermountain West, leading to a further increase in piñon pine logging (Graham and Sisk 2002). By about 1870, the more than six hundred charcoal makers had depleted the forests for a distance of 50 miles around Eureka and many other communities (Lanner 1981). This band of removal of forests around towns and mining areas was widespread, and included mining towns throughout Utah. By the end of the 19th century the piñon pine forests in the Great Basin had been radically reduced to a few stands mostly in places too rugged to cut. The Paiutes complained about the miners cutting down what was referred to as the "Indian orchards," which provided traditional winter food (Lanner 1981).

With the replacement of wood with mined coal (readily brought in on the new railroads) at the turn of the 20th century, these forests began to come back. Looking at the recent past, the recovery of the piñon-juniper forests across the West is sometimes misinterpreted as encroachment. Today juniper woodlands currently occupy 8.6 million acres (3.5 million ha) in the northern Great Basin, of which 95% occur in areas likely logged as part of the mining and are today woodlands of 120 years or less in age (Miller and Rose 1995; Miller et al. 2000). Prior to settlement, old juniper stands were most likely to have occupied large areas similar to the lands that they today occupy. The oldest western juniper aged to date is 1600 years old and is located on Horse Ridge, Oregon (Miller et al 2005). Prior to the mining era, we would expect to see old juniper throughout its historic range. Yet today, the majority of piñon-juniper stands that are surveyed for age in the Great Basin forests are less than 150 years of age.

In the 20th century, more piñon and juniper forests across the intermountain West began to be chemically treated or mechanically removed for range management purposes (Nesom 2002). Figures are hard to find but Lanner (1981) estimates that three million acres of woodland were converted into livestock pastures between 1960 and 1972 alone in the Great Basin and Intermountain West. So, what we see today in many cases is piñon-juniper simply recolonizing places where they were dominant but then chained in the 1940's to 1970's. Because these species are slow to establish and grow, the process of recolonization can take decades (Baker and Shinneman 2004). It can take so long, in fact, that people often forget that the area had once been chained, or perhaps had a high severity fire that killed off all the trees many decades prior, and so what is actually natural recolonization is often mistaken for encroachment.

How to address the "piñon-juniper encroachment" theory. In his chapter in Studies in Avian Biology No. 38 on guidelines for conducting sagebrush restoration projects, Pyke (2011) reports that if a site where ecological restoration is desirable but old growth juniper is present, that this is a clear indicator that the site is intrinsically a juniper forest and may not be appropriate for tree removal for restoration purposes.

One of the best methods to use address possible cases of piñon-juniper encroachment into apparent non P-J habitat is to use the Potential Natural Community descriptions in the U.S. Natural Resource Conservation Service (NRCS) Ecological Site Descriptions (ESD) associated with specific soil map units mapped in the NRCS Service soil survey for the area of concern[5]. It is helpful to compare the NRCS potential natural community description (which indicates whether or not P-J communities should be on the site) to current site conditions. If P-J communities are where the NRCS Ecological Site Description of Potential Natural Community says they should be, then it is not encroachment. Rather, it is likely recolonization from a past disturbance event, whether it is a 30-year old chaining, logging from 150 years ago, or recovery from a fire that was 300+ years ago.

An example of this analysis was done for the BLM Cedar City Resource Area in southwestern Utah. Using recent aerial images, soil survey range site GIS data, the NRCS Ecological Site Descriptions, and GAP vegetation coverages, the current piñon-juniper forest coverage was compared to potential coverage in the Cedar City Field Office area. The potential coverage was derived from the Ecological Site Description correlations to soil survey map units. The current extent of piñon juniper forest in the Cedar City Field Office area closely matches the area where this vegetation type is expected to be. The Ecological Site Descriptions based on the soil map units indicate that fully 32% of the Resource Area is expected to be piñon-juniper forest. Today, the P-J is basically right where it should be – and covers 34% of the resource Area. Therefore one cannot say that these areas have been "encroached." In addition, the same sort of analysis

[5] Soil surveys are produced by the Natural Resources and Conservation Service and provide useful information on the nature of soils and their potential plant communities based on field survey data. Soil survey data, including GIS data, are available at http://www.ut.nrcs.usda.gov/technical/soils/index.html.

was preformed for sagebrush. The Ecological Site Descriptions based on the soil map units indicate that 29% of the Resource Area is expected to be sagebrush lands, and in actuality 30% of the Resource Area is dominated by sagebrush – right where it is predicted to be.

Another example of this analysis was completed for a recent Environmental Assessment (EA) for a proposed piñon-juniper control project in the Dark Canyon pasture of the Indian Creek BLM allotment in southeast Utah. The EA used the term "encroachment" to describe P-J occurrence in this pasture, yet according to what was know about pre-settlement conditions in this area, most of the Dark Canyon Plateau was historically a P-J system. The proposed treatment GIS layers (polygons) in the Indian Creek allotment were overlaid with the NRCS Ecological Site Description and soil data that describes percent P-J expected for these polygons. It was discovered that out of the 6,348 acres proposed for treatment in the Dark Canyon Plateau area, 3,839 acres (or about 60%) of the treatment areas are expected to be P-J, based on the ESD for that soil type (NRCS 2010). Therefore, the term "encroachment" was recommended to only be used to describe the increased density of trees in Beef Basin (the other part of the Indian Creek allotment targeted for treatments), and the term "recolonization" to be used to describe increased density of trees on the Dark Canyon Plateau. It is noteworthy that the BLM in this EA many times referred to the "**continued** maintenance [past chaining, etc.] to reduce pinyon-juniper encroachment," under-scoring the reality that there has been an ongoing attempt over time to repeatedly thwart the recolonization of P-J in this area.

Mechanical treatments in light of livestock grazing

It is important to investigate the evidence that as sagebrush canopy cover increases perennial grass cover decreases (and vice-versa), in the context of livestock grazing. Treatments are often proposed to reduce sagebrush canopy cover in order to encourage more production of herbaceous ground cover (see issue #1 above). But, in ungrazed areas, higher density sagebrush canopy cover can be found along with high canopy cover for grasses and low bare ground percentages (e.g. Mueggler and Steward 1980, Peterson 1995, Jones 2000, Welch 2005); and this includes multiple studies that have simultaneously tracked increases in sagebrush cover alongside significant increases in grass cover after areas have been protected from grazing (e.g. Pearson 1965, McLean and Tisdale 1972, Anderson and Holte 1981,Branson and Miller 1981). Sagebrush communities at their ecological potential have little bare ground and can be dominated by perennial grasses and cryptobiotic soils in the absence of grazing as is traditionally practiced (Peterson 1995, Carter 2000, unpublished data, Welch and Criddle 2003). The seminal publication on sage-grouse in *Studies in Avian Biology* by Knick and Connelly (2011) states, "no evidence supports the belief that sagebrush dominance will continue at the expense of perennial grass cover or survival" (citing Pyke 2011). In his review of the literature on this topic, Welch (2005) reported that canopy cover of big sagebrush is not significantly correlated with either graminoid cover, forb cover, or bare soil cover. Others have noted that differences in perennial grass production in big sagebrush stands have less to do with shrub cover than it has to do with soil type, annual precipitation, grass species, and especially grazing history differences (Pechanec and Stewart 1949, Peterson 1995, Welch 2005). Further, if high sagebrush cover really leads to decreased cover of perennial grasses and native forbs in shrub interspaces, how can we explain situations where, on grazed sagebrush stands, virtually the only perennial grasses and native forbs that can be found are crowded underneath shrubs? (Welch and Criddle 2003, Welch 2005, O'Brien 2007).

Welch (2005) assembled the results of 29 separate studies that were conducted to determine the degree of perennial grass production that was achieved by reducing or eliminating big sagebrush by various means, on different types of sites, and for varying periods of times after treatment. Some of the studies involved seeding perennial grasses and forbs after the treatments, and others did not. He then compared these results to measurements of perennial grasses in ungrazed big sagebrush sites, and found that ungrazed or undisturbed big sagebrush sites produce about the same amount of perennial grasses as treated sites where the big sagebrush has been removed (Welch 2005).This suggests that the amount of perennial grass cover, or lack of it, in dense sagebrush stands, is often the result of excessive livestock grazing and not competitive exclusion of sagebrush on grasses and forbs.

The majority of studies reviewed for this synthesis which reported increased cover, frequency, productivity and/or density of native perennial grasses and/or forbs following mechanical treatment in either sagebrush habitat or piñon-juniper sites were monitored during the post-treatment (often two year or two growing season[6]) livestock exclusion period on the treatment site. Studies that monitored success of treatment for periods longer than two years following treatment, and after livestock grazing has been brought back to the site, were in the minority. Many published studies of the effects of mechanical treatments do not mention post-treatment grazing management at all, leaving the reader to guess to what degree the post treatment conditions are affected by livestock grazing. There are also cases where post treatment conditions that are protected from grazing are compared to adjacent non-treated conditions that are still being grazed (examples given in Welch 2005). In the case of this literature review, of the roughly 65 individual studies on the effects of various treatments in sagebrush and piñon-juniper woodlands reviewed, we found that only about eight reported that the post-treatment effect of livestock grazing was controlled for in any meaningful, well-designed way. These included Gifford 1973, Buckhouse and Gifford 1976, Busby and Gifford 1981, Gifford 1982, Wilcox 1994, Yeo 2009a and 2009b, and Teague et al. 2010.

In terms of piñon-juniper systems, the long history of livestock grazing in many piñon-juniper woodlands on the Colorado Plateau and intermountain West has both diminished and altered herbaceous vegetation, leading to widespread desertification of understory conditions (PRIA 1978, Laner 1981, Burkhardt 1996, Nevada Division of Water Planning 2000, Graham and Sisk 2002, Milchunas 2006).

The literature also reflects that many of the researchers who have studied piñon-juniper expansion into adjacent shrublands have concluded that the (true) expansion (not recolonization) is most likely due to livestock grazing and subsequent reductions in fire frequency (e.g. Ellison 1960, Burkhardt and Tisdale 1978, Young and Evans 1981, Eddleman 1987, Neilson 1957,

[6] A note on the customary 2-year rest period following treatment. Although the following excerpt from Miller et al. (2005) refers to post-fire juniper management, it is relevant to this issue: "Introduction of livestock after burning in western juniper woodlands has not received adequate scrutiny…typically two years of grazing rest is prescribed following fire. This requirement has never been tested experimentally. Decisions regarding livestock reintroduction should be made based on the response of vegetation following treatment. With slow community recovery, rest may be required beyond the standard 2-year time frame." Other articles reviewed in this literature review agree with Miller et al. that the two-year rest period after treatment is often inadequate (e.g. Gottfried 2004).

Evans 1988, Miller and Wigand 1994). This is especially true of researchers who have studied the interaction of livestock grazing and western juniper expansion in the Great Basin. As Miller et al. (2005) summarize, "Introduction of livestock in the 1860's and the large increase of animals from the 1870's through the early 1900's coincide with the initial expansion of western juniper woodlands. Season-long grazing by the large numbers of domestic livestock during this period is believed to have reduced fine fuel loads…the lack of fire and decreased competition from herbaceous species probably contributed to an increase in shrub density and cover, thus providing a greater number of safe sites for western juniper establishment."

In many situations with both degraded piñon-juniper and sagebrush communities, but particularly with sagebrush communities invaded or recolonized by P-J, the signs of ecosystem degradation that are attributed to encroachment are often impossible to tease apart from the symptoms caused by livestock grazing (since most sagebrush lands in the southwest and intermountain West are grazed by livestock). For example, sometimes decreased water infiltration and increased erosion is blamed on juniper expansion; yet, juniper expansion has been shown to have fewer effects on water infiltration and erosion than livestock, which reduce vegetative cover and disturb and compact soils with their hooves (McPherson and Wright 1990, Fleischner 1994, Wilcox 1994, Jones 2000 and references therein).

In her review of the literature on the effects of piñon-juniper treatments on western ecosystems, Belsky (1996) reflects on the confounding effect livestock grazing has on piñon-juniper woodlands: "Most of the earlier studies of juniper and pinyon-juniper removal were carried out on sites that were grazed by domestic livestock. The effects of livestock grazing and tree removal were therefore confounded, making it difficult to determine whether the resulting changes in biotic communities and ecosystem function were due to reduced tree densities, changes in livestock abundance and utilization patterns, or their interactions. It is also unknown to what degree herbaceous production would have differed if livestock grazing had been deferred, reduced, or eliminated. Without studies in which these two variables are controlled and investigated individually, it is impossible to ascertain the true impacts of western juniper on northwestern range ecosystems."

Before any treatment method is attempted at a sagebrush or piñon-juniper site, small-scale field tests and independent scientific validation need to occur to ensure that the proposed treatment method actually does lead to the proposed ecological restoration. Such studies need to account for all significant factors influencing recovery, such as the impacts of livestock grazing. In fact by far the biggest hole that currently needs to be addressed in the ecological restoration literature is a well-designed, long-term replicated study to explore the long-term interaction between sagebrush/grassland or sagebrush/PJ vegetation treatments and post treatment livestock grazing. A good example of the type of research that needs to be done on this front is currently underway with a team from USDA-Agricultural Research Service in Logan, which is two years into a 20+ year experiment at Kennecott Utah Copper that is investigating the effects of two different livestock return times (2 years, and after the site resembles the potential natural community), in addition to non-grazed treatment, on replicated Dixie-Harrowed pastures which include non-harrowed controls. A rigorous examination of the interaction of livestock grazing and mechanical sagebrush treatment, as is underway at Kennecott Utah Copper, is sorely needed, especially in this time of climate change.

In their recent literature review on the impacts of big sagebrush treatment on wildlife habitat, Beck et al. (2012) similarly point out the problems with many of the past studies used to determine the effects of sagebrush treatments and manipulations. They report that assessments of the effects of these treatments on diversity, density, or productivity of wildlife such as shrubland birds most often has been derived from studies of specific, fine-scale management actions and that most studies address short-term effects immediately post-treatment. Beck et al. (2012) go on to recommend that, going forward, future planned experiments designed to rigorously address the impacts of these treatments on wildlife need to be conducted over different temporal and spatial scales than they have historically been conducted, in order to better understand the response of a variety of species that are dependent on sagebrush habitats.

These calls for improved research regarding the interaction of both sagebrush and piñon-juniper treatments and post-treatment livestock grazing underscore the need for land managers to take more care with post treatment land management, especially on public lands. In most cases, grazing as is commonly practiced returns to the site quickly and the site regresses into pretreatment conditions rapidly. A case in point are the many sites across Utah and other places in the West where land managers re-treat the same site every 15 to 30 years. One has to wonder whether these land managers are simply responding to the symptoms, and not addressing the underlying problems and original causes for degradation at the site. This literature review logically leads to the conclusion that post-treatment changes in grazing practices should accompany both sagebrush and piñon-juniper treatments in order to reach agency goals for those sites, for the long term.

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