

Do Mechanical Vegetation Treatments of Pinyon-Juniper and Sagebrush Communities Work?

A Review of the Literature

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TABLE OF CONTENTS

Executive Summary	4
1. Introduction	9
1.1 Ecological Importance of Pinyon-Juniper Woodlands.....	10
1.2 Ecological Importance of Sagebrush Systems.....	10
1.3 Historical and Current Ecological Context of Pinyon-Juniper and Sagebrush Systems	11
1.3.1 Pinyon-Juniper Systems.....	11
1.3.2 Sagebrush Systems.....	13
1.3.3 Expansion of Pinyon-Juniper Woodlands into Sagebrush Systems.....	15
2. Methods	16
3. Vegetation Treatment Objectives	17
3.1 Vegetation Structure.....	17
3.1.1 Herbaceous Cover and Diversity in Pinyon-Juniper Woodlands.....	17
3.1.2 Herbaceous Cover and Diversity in Sagebrush Communities.....	20
3.2 Fuels Management.....	22
3.2.1 Pinyon-Juniper Systems	23
3.2.2 Sagebrush Systems	24
3.3 Wildlife Habitat.....	25
3.3.1 Pinyon-Juniper Treatments.....	25
3.3.2 Sagebrush Treatments.....	27
3.3.3 Greater Sage-Grouse	30
3.4 Ecosystem Function.....	33
3.4.1 Soil Stability.....	33
3.4.2 Watershed Productivity	36
3.4.3 Carbon Sequestration and Climate Change.....	38
3.5 Livestock Grazing.....	39
3.5.1 Livestock and Vegetation Functional Groups in Pinyon-Juniper Communities	39
3.5.2 Livestock and Vegetation Functional Groups in Sagebrush Communities	40
3.5.3 Exclosure studies.....	41
4. Data Gaps & Recommendations	43
5. Summary and Conclusions	46
Literature Cited	48
Appendix: Studies Summarized in Figures	70

EXECUTIVE SUMMARY

Vegetation manipulation treatments in pinyon (*Pinus* spp.) - juniper (*Juniperus* spp.) and sagebrush (*Artemisia* spp.) plant communities are increasing at a rapid rate on public lands. These vegetation types have changed significantly over the last few centuries, and in some cases so have their fire regimes, making management goals on public land more difficult to attain. Managers are turning to mechanical vegetation treatments in an effort to restore vegetation, manage fuels, improve wildlife habitat, increase water flow, and reduce soil erosion. This literature review summarizes research on the degree to which these objectives have been met based on our review of over 300 scientific studies, reports and articles. We also summarize available information on post-treatment land management and its effects on the long-term success or failure of vegetation treatment projects. Finally, we discuss data gaps and conclude with recommendations from the literature.

The term “mechanical treatment” used in this literature review refers to all activities that remove or reduce vegetation by mechanical means. This includes chaining, mastication, Dixie harrowing, drill seeding, and hand cutting. Miller et al. (2005) and Stevens et al. (1999) are sources on the different mechanical treatment methods covered in this review.

We systematically collected and reviewed sources beginning with a search of keywords in Google Scholar and Science Direct search engines. We attempted to find common conclusions in the literature for the various environmental responses to treatments; following the methods of Bombaci and Pejchar (2016), all sources that had comparisons between pre- and post-treatment effects or between treated and untreated control sites were used to create summary charts showing negative, positive, or no significant effects of treatments on several response categories: herbaceous ground cover in both sagebrush and pinyon-juniper treatments, wildlife response to sagebrush treatment (with sage-grouse treated separately), soil erosion and water runoff, and hydrological related variables. We also reprint Bombaci and Pejchar’s summary chart for the response of wildlife other than sage-grouse to pinyon-juniper treatments.

RESULTS

Herbaceous Functional Groups

The responses of grasses and forbs to mechanical treatments were highly variable (see figures on pages 17 and 20). Many factors influence herbaceous plants, including how long after treatment the data were collected (studies in this review ranged from 1 year to 25 to 30 years post-treatment). These studies need to be further analyzed with additional meta-analyses and statistical methodology. With that caution in mind, we found that:

- In pinyon-juniper communities, most data points (64%) showed no significant effect of treatments on perennial grasses and forbs. However, where there were significant results, treatments elicited more positive responses (increases in cover) in grasses and forbs than negative responses (29% and 7%, respectively). Non-native annuals showed increases in cover in half of the data points. The other half showed no significant

impact from mechanical treatments. Non-native annuals showed no negative effects from treatment.

- In sagebrush communities, most data points (56%) again showed no significant effects of treatments on grasses and forbs. Of the studies that did show a response, forbs had only slightly more positive responses (23%) than negative (19%). Grasses, however, showed far more positive responses (33%) than negative (8%). For non-native grasses and forbs, studies were almost evenly divided between no significant response and positive response (24% and 26%, respectively). This group had no negative effects from treatment.

One general response across studies and geographic locations is the poor performance of perennial forbs in many treatments. Some researchers speculate that overgrazing may be the cause, but others think climate change and changing precipitation levels may explain this pattern.

Fuels Management

Prior to European contact, fire frequency in pinyon-juniper woodlands varied with community and site characteristics but was thought to be rare in general. In the case of persistent pinyon-juniper woodlands, the fire cycle was on the order of hundreds of years. When fires did occur they were often severe. Factors such as fire suppression, grazing, the spread of flammable exotic species, and climate change impact the fire dynamics of these communities today. Wildfire control via fuels reduction is the goal of some vegetation treatments. We could not create a summary chart for this variable due to scarcity of studies of the same topic, but recent studies suggest that climate has a greater influence on fire activity than fine fuels and biomass. Other researchers found that the surface disturbances associated with mechanical treatments may facilitate cheatgrass (*Bromus tectorum*) expansion and lead to increased fires. At present, there is little research supporting the contention that removing pinyon and juniper reduces incidence of fire.

Although some studies suggest that fire return interval in sagebrush communities is 10 to 40 years, there are no data to support that. In fact, other researchers indicate it may be between 50 to 150 years or more. Since half of the studies in our review showed that treatments increase flammable non-natives, they may actually shorten the fire cycle rather than restore the natural fire regime.

Wildlife

Studies on the effects of treatments on wildlife are variable. Fifty percent of the data points on sagebrush treatments indicated positive effects on wildlife, 23% showed negative effects, and 27% had no significant effect (see figure on page 28). For pinyon-juniper treatments, we reprint the results of Bombaci and Pejchar (2016), which summarizes this literature (see figures on pages 25 and 26). While they broke down their results into responses of small mammals, ungulates, birds and invertebrates to mechanical removal of pinyon-juniper woodlands (and also reported results of thinning treatments and mechanical removal plus burning), they found that the general trend across studies was for non-significant results of mechanical removal.





The exception was for birds where, especially for pinyon-juniper obligates such as pinyon jays (*Gymnorhinus cyanocephalus*), there is a negative response to tree removal. Apart from Brewer's sparrow (*Spizella breweri*), which showed positive responses to pinyon-juniper removal treatments, most sagebrush-obligate birds showed no significant response. However, many studies are conducted fairly soon post-treatment. Longer-term studies found significant differences between treated and untreated sites, with species sorting out according to their habitat needs.

Managing habitat for wildlife is complex. Species often specialize for specific habitat conditions, and what benefits one species may be a detriment to another. The best strategy is to maintain heterogeneous, patchy mosaics across the landscape of vegetation types in all stages of succession. This argues against large expanses being treated with one method that creates a single homogenized vegetation community.

The effects of treatments on greater sage-grouse were treated in a separate summary chart. Of the five studies of pinyon-juniper treatments, three showed positive effects and two showed non-significant effects. Of the 11 studies of sagebrush treatment effects, four were positive, three were negative, and four showed no significant effects (see figure on page 31).

Soil Stability

Mechanical treatments disturb soils, which often leads to an increase in erosion. Whether this is a short-term effect that diminishes as herbaceous vegetation increases or a long-term effect exacerbated by increased exotics is dependent on multiple variables. Where biological soil crust is a component of soil stability, its removal can increase wind and water soil loss. The majority of studies we reviewed (74% of data points) showed no significant response of either run-off or erosion to mechanical treatment. Some studies (5% of data points) find treatments decrease runoff and erosion, but others studies (21% of data points) find treatments increase runoff and erosion. (see figure on page 34) Techniques that leave slash or wood chips in place result in significantly less erosion in some, but not all, studies. Seeding after treatment is recommended. Hand thinning is the least disruptive method of treatment to soils.

Watershed Productivity

Studies investigating whether vegetation treatments increase water yield, either at the surface or in ground water recharge, have varying results (see figure on page 36) depending on study site characteristics (e.g., elevation, vegetation type, timing, amount, and type of precipitation). Several literature reviews aggregating other results have concluded that treatments do not reliably increase water yield on a watershed scale, although water availability may increase in local areas. Other studies suggest that areas with higher precipitation levels have a greater possibility of increasing water availability than areas with less precipitation.

Carbon Sequestration

Research into the carbon sequestration potential of pinyon-juniper woodlands is limited, but recent syntheses suggest that carbon is more effectively sequestered in vegetation biomass. The contention that trees should be re-

moved to reduce the incidence of wildfire, which would release more carbon into the air, is unfounded.

Livestock Grazing

Since livestock grazing is a widespread land use inextricably woven into vegetation dynamics throughout the West, we reviewed literature that addresses its relationship to mechanical treatments. One major finding is that most treatment research does not control for this activity, either before or after treatment. Many projects assess treatments in the short post-treatment period when livestock are absent from the site and vegetation is recovering. Few studies return and assess treatments on a longer term basis when livestock have returned to the site. Where they do, results are variable. Without this information, post-treatment changes in a site's resource condition cannot be definitively attributed to treatment effects. Failing to account for the effects of livestock grazing makes it difficult to assess the causal factors of ecosystem condition and draw implications for management.

RECOMMENDATIONS AND CONCLUSIONS

It is important to remember that most of the studies we reviewed reflect one point along the trajectory of treatment progress. Studies conducted shortly after treatment may have different results than those returning to the treatment after longer periods. As researchers learn more about the effects of these treatments, areas of study that require further exploration are becoming apparent. The data gaps we have identified range from understanding why perennial forbs generally perform poorly in restoration projects to the need for well-designed, long-term, replicated studies of the interaction between vegetation treatments and post-treatment livestock grazing. Using passive restoration to restore ecosystem function has not received enough attention in the treatment literature. There is a clear need for future literature reviews to use meta-analytic statistics to be able to draw stronger conclusions on the effects of treatments across varied data sets and regions. The increase in exotic annuals that has been reported from many studies may be a primary threat to persistence of ecosystems. The alarming possibility that treatments may facilitate continued expansion of these populations and degrade native communities calls for further scrutiny.

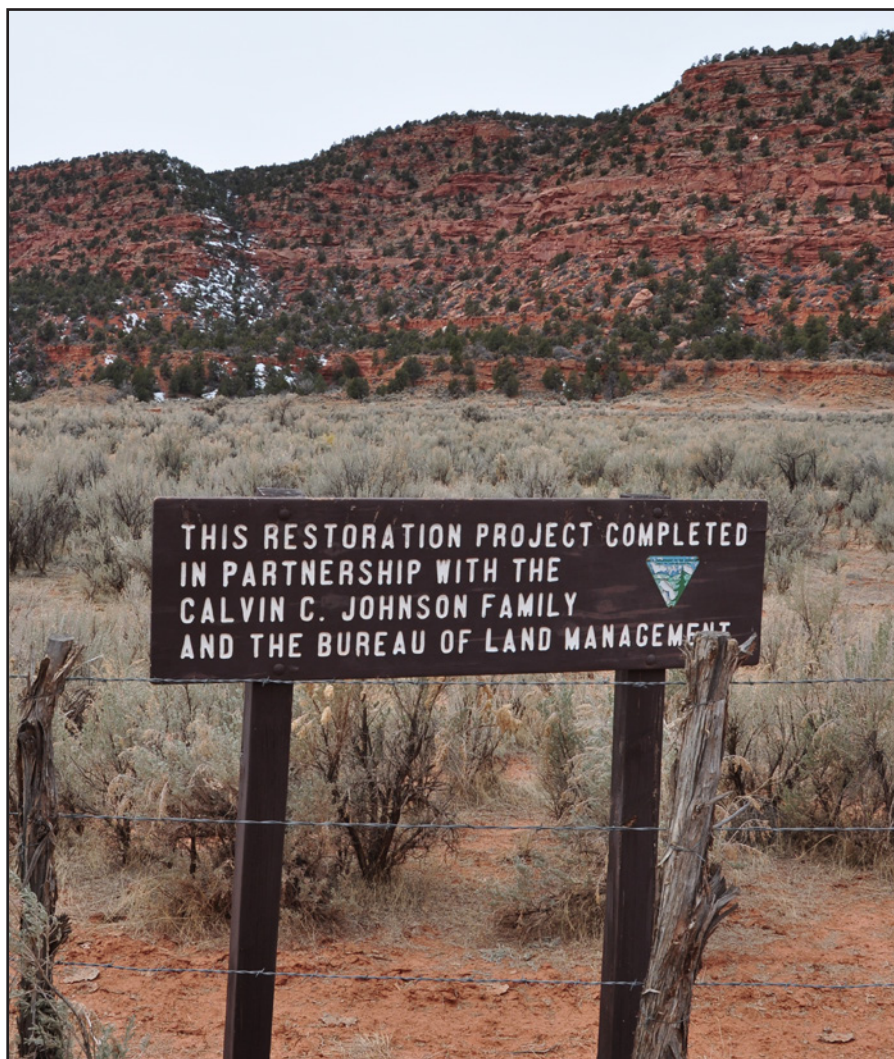
The disparity in responses to treatment is a clear indication that treatments are not "one size fits all." Planners must beware of applying the same mechanical treatments over vast areas of pinyon-juniper woodlands or sagebrush steppe vegetation communities with variable site characteristics. A careful treatment plan must be designed before implementation. Practitioners should conduct small-scale, pilot field tests with the proposed treatment method before applying it on a larger scale. This will prolong the time before treatments can be applied on a larger scale but this information is necessary to avoid resource degradation. Pilot studies should be followed by independent post treatment scientific validation, ideally with long-term monitoring of the site, to ensure that the proposed treatment method actually does lead to the intended ecological conditions. As changing climatic conditions make predicting the results and risks of mechanical treatments even more uncertain, public land managers should aim for more transparency in the decision process to explain the expectations for a project and the science guiding the planning effort.





Finally, it will be important to explore the reasons that most response variables in our summary charts show no significant difference between treatment and controls for more than half of the studies. This comports with Bombaci and Pejchar (2016), who also found a large amount of non-significant results in their own meta-analysis of the effect of treatments on wildlife. They say that these results may have several explanations: the metrics used were not appropriate to detect significant changes; the time frame of data collection was too short; responses lacked statistical power to detect differences; or, finally, treatments truly do not make much of a difference much of the time. They caution against drawing the latter conclusion until more meta-analyses can determine why so many studies obtain non-significant results.

However, if these non-significant responses truly indicate that mechanical treatments are not producing the desired results, then a re-evaluation of their efficacy or perhaps post-treatment management is necessary. As Archer and Predick (2014) have said, “Despite the considerable investments in personnel, equipment, fuel, chemicals, etc., associated with the application of various brush management practices, the recovery of key ecosystem services may not occur or may be short-lived and require subsequent interventions.”



(Photo: Kya Marienfeld)



Spruce Mountain, Nevada about four years after a juniper mastication project. (Photo: Laura Cunningham)

1. INTRODUCTION

Mechanical vegetation treatments in pinyon-juniper (*Pinus* spp.- *Juniperus* spp.) and sagebrush (*Artemisia* spp.) communities have substantially expanded in recent years to manage wildlife habitat, ecosystem health, wildfires, and forage for livestock. Hundreds of thousands of acres have been subject to some kind of management action and land managers have plans to continue practices into the foreseeable future. There is a growing body of research on the effectiveness of management actions in pinyon-juniper and sagebrush communities. This document provides a review of the existing literature on the effects of mechanical treatment in pinyon-juniper and sagebrush communities as a means to understand results and determine whether management actions are achieving goals and objectives.

Mechanical treatment methods in this review include chaining, mastication, Dixie harrowing, drill seeding, and hand cutting with chainsaws. In the chaining method, anchor chains from large destroyer or cruiser ships, 40- to 160-lb per link and 90 to 350 ft. long, are pulled between two crawler tractors traveling parallel to each other. Trees and shrubs in the path of the chain are uprooted, pruned, or topped. The Dixie harrow is a large, spike-toothed pipe implement that is pulled behind a single large tractor. The teeth of the harrow are at alternating angles, which causes it to grab and rip the sagebrush out of the ground leaving scarified bare soil.

A Bull Hog masticator is a large metal drum attached to a front end loader or excavator. It shreds trees and other vegetation into mulch which is typically left on site, and sometimes burned in place (Miller et al. 2005; Stevens et al. 1999).

Below, we begin with an overview of pinyon-juniper and sagebrush communities and their ecological importance. We then give the historical and current ecological context of pinyon-juniper and sagebrush ecosystems and their communities, to set the stage for today's most common objectives for mechanical treatment, and its effects on community characteristics. We then detail the methods used in this review, including how we distilled hundreds of studies into a handful of simple summary charts.

The bulk of this document seeks to distill major themes and trends from the abundant pool of literature on the outcomes of mechanical treatments in both pinyon-juniper and sagebrush systems. The different categories below reflect the most common objectives and justifications we see for mechanical treatments. This includes promoting herbaceous cover and managing fine fuels. We summarize studies on treatments that are designed to provide habitat for wildlife and determine the degree to which those goals are achieved. We address the effect of treatments on soil erosion, watershed produc-

tivity, and carbon sequestration. Livestock grazing, as a widespread land use that has effects on all of the previous elements and so inevitably interacts with vegetation treatments, is treated in its own section. We end with a summary and recommendations for future management.

1.1 ECOLOGICAL IMPORTANCE OF PINYON-JUNIPER WOODLANDS

Pinyon-juniper woodlands occur in ten states and cover large areas in many of them. These woodlands can be dominated by several species of pinyon pine and juniper (Lanner 1981; Mitchell and Roberts 1999; Tausch and Hood 2007), and are very biodiverse. One study found that at least 450 species of vascular plants and 150 species of vertebrates occur in pinyon-juniper woodlands (Buckman and Wolters 1987). Important game species such as elk (*Cervus canadensis*), mule deer (*Odocoileus hemionus*), and wild turkey (*Meleagris gallopavo*) are year-round residents in pinyon-juniper woodlands and depend on this habitat for food and cover (Martin et al. 1961; Nesom 2002). Maser and Gashwiler (1978) 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 bird species have mutualistic relationships with pinyon pine and pinyon-juniper woodlands: Clark's nutcracker (*Nucifraga columbiana*), Steller's jay (*Cyanocitta stelleri*), Woodhouse's Scrub-jay (*Aphelocoma woodhouseii*), and pinyon jay (*Gymnorhinus cyanocephalus*) (Balda and Masters 1980). These birds depend on pinyon-juniper woodlands for food and are the primary agents of dispersal and regeneration of pinyon pines. Older trees are more valuable for these birds. Pinyon pines may bear cones at 25 years of age, but they only produce significant quantities of seeds each season after reaching 75 to 100 years old (Balda and Masters 1980).

The pinyon jay is currently a species of conservation concern, given it is one of the landbirds declining the fastest and most persistently in the intermountain West, at an average rate of -3.6% from 1968 to 2015, according to the Breeding Bird Survey (Boone et al. 2018). Despite the population's falling by >50% over this period, the pinyon jay has not been widely studied, and little is known about the factors responsible for its diminishing numbers. The relationship with the population decline of pinyon jays and current management in western pinyon-juniper woodlands, including removal of trees for fuel reduction or to create or protect shrublands for the benefit of sagebrush-associated wildlife, has received little study. Thus, Boone et al. (2018) call for further research to clarify the causes of the pinyon jay's decline and devise approaches for management of pinyon-juniper woodlands that balance the interests of the pinyon jay and other species of concern tied to pinyon-juniper woodlands.

1.2 ECOLOGICAL IMPORTANCE OF SAGEBRUSH SYSTEMS

Perhaps no plant evokes a common vision of the semi-arid landscapes of western North America as does sagebrush (Kitchen and McArthur 2007). Historically covering 250 million acres of the western United States, sagebrush is considered a keystone species because it is ecologically influential and provides habitat for many plants and animals (Beck et al. 2012; Braun et al. 1977; Connelly et al. 2011; Khanina 1998; Knick et al. 2003). Sagebrush systems host scores of other species of native plants and at least 24 species of lichens (Rosentreter 1990). Many wildlife species are depen-



Pinyon jays are among the four bird species that have a mutualistic relationship with pinyon pine and pinyon-juniper woodlands. (Photo: Alan Schmierer)



Sagebrush obligates like the pygmy rabbit rely entirely on sagebrush communities (Photo: Bureau of Land Management)

dent on sagebrush communities for all or a portion of the year including deer, elk, over 100 species of birds, numerous invertebrates including 72 species of spiders, 18 species of beetles, 13 species of grasshoppers or katydids, 54 aphid species, and 32 species of midges (Beck et al. 2012; Braun et al. 1977 and references therein; Connelly et al. 2000 and references therein; McArthur et al. 1978; Peterson 1995; Rosentreter 1990; Welch 2005). Over two dozen wildlife species, such as pygmy rabbit (*Brachylagus idahoensis*) are sagebrush obligates, and rely entirely on sagebrush communities (e.g. Burak 2006; Crawford 2008; Green and Flinders 1980).

As forage, sagebrush species contain high levels of protein and other nutrients (Kelsey et al. 1982; Wambolt 2004; Welch and McArthur 1979) and are highly digestible (Striby et al. 1987; Welch and Pederson 1981). Seventeen mammals consume sagebrush (Beck et al. 2012; Welch 2005 and references therein; Welch and Criddle 2003), especially during the winter months (Peterson 1995).

Sagebrush has important qualities that contribute to soil and hydrological function. For example, big sagebrush (*Artemisia tridentata*) can create “islands of fertility” in the landscape (Welch 2005). Big sagebrush is characterized as a “soil builder” because the deep root system can extract minerals and water deep in the soil profile and bring nutrients and moisture to the soil surface for use by other plants (Chambers 2000; Doescher et al. 1984; Richards and Caldwell 1987; Welch 2005). Big

sagebrush communities also promote deep soil water storage because the plants allow a uniform accumulation of snow, delay snow melt, and can retard the development of ice sheets (Hutchison 1965). Sagebrush can “extend” water near the soil surface by shading soil beneath its canopy (Wight et al. 1992). The shading can prolong the period favorable for seedling establishment (Chambers 2000; Pierson and Wight 1991; Wight et al. 1992).

1.3 HISTORICAL AND CURRENT ECOLOGICAL CONTEXT OF PINYON-JUNIPER AND SAGEBRUSH SYSTEMS

1.3.1 Pinyon-Juniper Systems

In the western United States, there were historically an estimated 50 million acres of pinyon-juniper woodland (Gottfried and Severson 1994; Mitchell and Roberts 1999). These communities have large ecological amplitudes; their range can extend from the upper edge of salt desert shrub communities at the lowest elevations to the lower fringes of subalpine communities at the higher elevations (Tausch and Hood 2007; West et al. 1998). Pinyon and juniper trees are often associated with a range of sagebrush species and subspecies. Where they co-occur, sagebrush and woodland communities can have different states of co-dominance within the overall successional dynamics of the sagebrush/ woodland ecosystem complex of a particular landscape (Tausch and Hood 2007). How these codominant patterns influence both historical and current fire regimes and expansion of pinyon-juniper woodlands into sagebrush systems are covered in more detail below.

The pre-Euro-American historical fire regimes for pinyon-juniper woodlands in the Great Basin and Colorado Plateau have been a matter of some debate. They most likely varied greatly. Moreover, when discussing pre-settlement fire regimes, it is important to also consider the influence that aboriginal fire-setting, presumably in order to influence both wildlife habitat and resource foraging, was having on pinyon-juniper woodlands on the eve of Euro-American contact (Raisha et al. 2005). However, most researchers agree that the patterns of historical 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. The pattern and behavior of fire was closely related to the unique interactions of topography, soils, environmental conditions, and vegeta-



Most researchers have concluded that infrequent high-severity “crown” fire has likely been the dominant fire regime in most Intermountain West pinyon-juniper woodlands both historically and presently. (Photo: National Park Service)

tion composition present at that time on each landscape area of interest. Then, as now, larger fires tended to occur during periods of drought (Betancourt et al. 1993; Swetnam and Betancourt 1998). Insects, diseases, and native ungulates appear to have played a widespread but relatively minor role (Tausch and Hood 2007).

Literature reviews on the topic of historical fire regimes in pinyon-juniper woodlands have pointed out common areas of agreement among many ecologists. Most authors find that pinyon-juniper woodlands are susceptible to high-severity fires both now and in the past. Fire intervals vary depending on type of pinyon-juniper woodland and presence of non-native plants.

Romme et al. (2009) suggested that there are three types of pinyon-juniper vegetation, all of which have differences in understory composition and length of fire rotations: Persistent Pinyon-Juniper Woodlands, Pinyon-Juniper Savannas, and Wooded Shrublands. Persistent Pinyon-Juniper Woodlands, which can be found throughout much of the Colorado Plateau and Great Basin, range from sparse stands of small trees growing on poor substrates to dense stands of large trees growing on more productive substrates. These communities exhibit variable cover and the under-

story is often sparse with significant areas of bare ground. Fire is inherently rare. In fact, Romme et al. (2009) describe how many Persistent Pinyon-Juniper Woodlands exhibit little to no evidence that they ever sustained widespread surface fires; rather, high-severity “crown” fire was likely the dominant fire regime. Over time, these woodlands accumulate fuel and conditions become highly flammable, and fires are typically stand-replacing. Estimates on historical fire intervals in Persistent Pinyon-Juniper Woodlands vary from 400 to 600 years, based on best available fire scar data from across the West (Romme et al. 2009). Historically, Pinyon-Juniper Savannas, which are found further south and east in places such as New Mexico and Arizona, receive monsoon rains that likely shortened historic fire return intervals. These savannas have low to moderate density and cover of pinyon or juniper or both, with a well-developed understory of nearly continuous grass or forb cover. Shrubs may be present but are usually only a minor component. Wooded Shrublands tend to have the soil, climate, and natural disturbance patterns that favor shrubs as a major part of pinyon-juniper forests (Romme et al. 2009).

Romme et al. (2009) stressed that spreading, low-intensity, surface fires had a limited role in molding stand structure and dynamics of most pinyon and juniper woodlands. Historical fires in all pinyon-juniper-woodland types generally did not “thin from below” or 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 was true historically and for most ecologically significant fires today. The authors concluded that in many pinyon-juniper woodlands, stand dynamics are driven more by climatic fluctuation, insects, and disease than by fire.

In a synthesis of fire ecology and management of pinyon-juniper systems in southern Utah, Tausch and Hood (2007) explain the history of fire in the region before Euro-American settlement. Deeper soils in the canyon bottoms and swales in pinyon-juniper woodlands were generally more productive for herbaceous species, and thus had higher fire frequencies. As soils become shallower, such as on steeper topography, the abundance of perennial herbaceous species becomes more limited. Shrubs and low trees are more competitive on these substrates because their deeper roots can exploit water trapped in cracks in the rocks—water that is not available to herbaceous species with shallow roots. Fires appear to have been less frequent, increas-

ing the probability of dominance by trees, which can often be several centuries old.

Baker and Shinneman (2004) also reported that low-severity surface fires were not common in pinyon-juniper woodlands, and they found no evidence that low-severity surface fires would have consistently reduced tree density in moderate-density woodlands, even with sagebrush or grassy understories. Although the authors found some evidence that surface fires may occur in higher elevation pinyon-juniper ecotones with ponderosa pine (*Pinus ponderosa*), they found little data to support the idea that fires spread widely in pinyon-juniper savannas at lower-elevation ecotones. Baker and Shinneman (2004) documented 126 wildfires in pinyon-juniper woodlands since Euro-American settlement that were described in the literature, and of these, two were low severity, three were possibly mixed severity, and 121 were high severity. The authors concluded that there are no data to demonstrate that the frequency of high-severity fires has increased or decreased in pinyon-juniper woodlands since Euro-American settlement and that frequent fire interval estimates (i.e., 13 to 35 years) from other researchers (Brown 2000; Frost 1998; Hardy et al. 2000) were not supported. However, other studies have suggested recent regional increases in severe crown fires in pinyon-juni-

per woodlands relative to historical periods (e.g., Floyd et al. 2004), and some of these areas may continue to have more frequent fires where nonnative annual grasses (e.g., cheatgrass [*Bromus tectorum*]) have invaded (Floyd et al. 2006 studying pinyon-juniper systems specifically, and DellaSala 2018 and Finney et al. 2011 in context of fires in forested systems generally).

1.3.2. Sagebrush Systems

Today, sagebrush, and in particular big sagebrush, is found throughout western North America from southern Canada to Baja California (Kitchen and McArthur 2007 and references therein). The relationship between modern and pre-settlement distribution and condition of big sagebrush communities has been a matter of some debate (Peterson 1995; Young et al. 1979). One view holds that in response to livestock grazing practices and altered fire regimes, big sagebrush invaded large landscapes that were predominantly grasslands (Christensen and Johnson 1964; Cottam and Stewart 1940; Hull and Hull 1974). Other researchers posit that, with the exception of lands converted to other uses, the distribution of big sagebrush landscapes is essentially unchanged from historic times (Hironaka 1979; Johnson 1986; Welch 2005). This view is supported by arguments that expansion rates for sagebrush are too slow to account for significant range advances



The relationship between modern and pre-settlement distribution and condition of big sagebrush communities has been a matter of some debate. (Photo: Ray Bloxham)

in the suggested time frame of approximately 100 years (Welch 2005). Early written accounts produced by trappers, explorers, immigrants, and settlers have been interpreted to support both positions (Dorn 1986; Kitchen and McArthur 2007; Knight 2014).

Historical fire regimes in sagebrush communities vary greatly depending on the environmental setting and sagebrush community types (Douglas Shinneman, personal communication, November 2018; Kitchen and McArthur 2007). Moreover, when discussing “pre-settlement” fire regimes it is important to also consider the influence that aboriginal fire-setting, presumably in order to influence both wildlife habitat and resource foraging, was having on sagebrush systems on the eve of Euro-American contact (Raisha et al. 2005). A review by Welch and Criddle (2003) indicated that the fire return interval in sagebrush-grass communities and big sagebrush communities is likely between 50 and 125 years (Welch 2005; Whisennant 1989; citing Wright and Bailey 1982). In Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis*) fire cycles historically were of longer duration and average fire rotation likely ranged from 100 to over 300 years,



Lacking fire-adaptive traits, sagebrush is susceptible to highly damaging crown fires from which it does not recover quickly, sometimes requiring 25 to 100 years to regenerate. (Photo: Scott Schaff/USGS)

depending on climate, topography, plant composition, and ecological site characteristics (Baker 2011; Bukowski and Baker 2013; Wright and Bailey 1982). Big sagebrush communities can maintain themselves without the occurrence of fire (Lommasson 1948). The historic fire interval in mountain big sagebrush (*Artemisia tridentata* subsp. *vaseyana*) was more frequent than that of Wyoming big sagebrush (Miller and Heyerdahl 2008; Welch 2005 and references therein) because there tends to be more biomass in these understories due to higher rates of annual precipitation in mountain big sagebrush zones. Estimates of historic fire return intervals in mountain big sagebrush zones have been calculated to vary from 35 to 80 years (Arno and Gruell 1983; Heyerdahl et al. 2006; Kitchen and McArthur 2007; Miller and Rose 1999; Wright and Bailey 1982). After reviewing the literature Kitchen and McArthur (2007) suggested that historic fire return intervals averaged from 40 to 80 years for mountain big sagebrush and some productive basin and Wyoming big sagebrush communities and were as long as 100 to 200 years or longer for big and black sagebrush (*Artemisia nova*) sites with low productivity. Xeric and dwarf (e.g., *A. arbuscula*) sagebrush communities are generally more fuel limited and may have had historical fire rotations of several hundred years or more in some regions (Baker 2013; Bukowski and Baker 2013).

Biological and ecological characteristics of sagebrush suggest the species did not evolve with frequent fires. The genus lacks many of the features that other fire-adapted taxa have. For example, sagebrush is highly flammable and susceptible to damaging crown fires (Welch 2005; Welch and Criddle 2003). It is non-sprouting and must germinate from seeds (Pechanec et al. 1965; Tisdale and Hironaka 1981), which are not adapted to fire and are not present in the seedbank in high amounts (Welch 2005; Welch and Criddle 2003). After a fire, sagebrush does not recover quickly. Big sagebrush, for example, requires 25 to over 100 years (Baker 2011; Connelly et al. 2000; Kitchen and McArthur 2007; Shinneman and McIlroy 2016; Wambolt et al. 2001; Watts and Wambolt 1996; Welch 2005; Welch 2006). Sagebrush, depending on the species, has average life expectancies of 60 to 70 years and, in rare cases, may survive over 200 years (Ferguson 1964; Ferguson and Humphrey 1959; Welch 2005).

Kitchen and McArthur (2007) summarize the adaptation of big sagebrush to fire thusly: “[historically] big sagebrush solved the fire problem by producing highly competitive, yet disposable plants. It does not invest

resources in morphological or physiological adaptations to fire, as it never had to in its short evolutionary past. This was particularly true for the 2+ million years of the Pleistocene, during which time cooler climatic conditions would have rarely favored fire to the extent they do today. Sagebrush thrives on suitable landscapes as long as the fire-free intervals are sufficiently long to permit re-establishment of mature stands, and short enough to prevent displacement by forest or woodland” (citing Miller and Tausch 2001).

1.3.3 Expansion of Pinyon-Juniper Woodlands into Sagebrush Systems

The expansion of pinyon and juniper into sagebrush communities is well documented (Blackburn and Tueller 1970; Cottam and Stewart 1940; Miller and Rose 1999; Miller and Wigand 1994). Many studies have reported on the causes of pinyon-juniper expansion, including decreased fire frequency (Archer et al. 2011; Archer and Predick 2014; Bauer and Weisberg 2009; Burkhardt and Tisdale 1976; Miller and Rose 1999; Romme et al. 2009), overgrazing by livestock that reduces fuels and reduces competitive interactions between trees and herbaceous species (Archer et al. 2011; Archer and Predick 2014; Blackburn and Tueller 1970; Miller and Tausch 2001; Shinneman and Baker 2009), recovery of pinyon-juniper woodlands at a local scale in response to past clearing or logging (Ko et al. 2011; Lanner 1981; Romme et al. 2009; Sallach 1986), and some combination of these factors (Archer et al. 2011 and references therein). Some researchers report that drought or climate change can also trigger pinyon-juniper expansion (Archer et al. 2011; Barger et al. 2009; Fritts 1974; Miller and Wigand 1994), for example through enhanced atmospheric CO₂ concentrations (Soule et al. 2003).

Pinyon-juniper-woodland expansion into sagebrush communities has been correlated with the loss of wildlife habitat, increased erosion, loss of herbaceous species, non-native species invasion, and decreases in water quantity and quality (Baker and Shinneman 2004; Blackburn and Tueller 1970; Burkhardt and Tisdale, 1969, 1976; Soule and Knapp, 1999). Thus, current management of pinyon-juniper woodlands is often based on the assumption that removal of pinyon-juniper trees will reverse these conditions.

However, the evidence for expansion of pinyon-juniper into other community types (usually sagebrush) needs to be weighed within the context of the different types of pinyon-juniper woodlands (Persistent Pinyon-Juni-

per Woodlands, Pinyon-Juniper Savannas, and Wooded Shrublands) (Romme et al. 2009), their distribution and juxtaposition at a landscape scale, and the value of old pinyon-juniper woodlands. It is important to consider the ecological distinction between recently invaded sagebrush landscapes versus old pinyon-juniper woodlands. At the same time, it can be quite difficult to ascertain when an area is indeed a wooded shrubland and has been for hundreds of years or whether it was once sagebrush into which pinyon and juniper has expanded. In some cases the soil type and associated Ecological Site Description can help shed light on the true nature of the woodland and sagebrush association. In other cases the management goals might be the same (i.e., tree removal) regardless of whether the site is indeed a wooded shrubland on a soil type favoring pinyon and junipers trees, or an invaded sagebrush Ecological Site Type. Yet another uncertain area regarding pinyon-juniper expansion is environmental conditions that favor infilling of wooded shrublands over time, to the degree that they eventually resemble persistent woodlands. 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 (Tausch and Hood 2007). Because these systems are dynamic and highly variable across the landscape, successional status and associated ecosystems of pinyon-juniper woodlands are the result of complex interactions of topography, soils, environmental conditions, past patterns of disturbance, and successional processes through time (Tausch and Hood 2007).

Regardless of the reasons for pinyon-juniper expansion, practitioners who focus management attention on areas where woodlands have expanded into shrublands classify the stages of pinyon-juniper expansion into “Phase I, Phase II and Phase III” stages. Phase I woodlands are defined by the dominance of shrubs and herbaceous vegetation layers associated with early phases of woodland encroachment, and typically, tree cover is less than 10 %. Phase II woodlands are those in which trees, shrubs, and herbaceous vegetation share relative dominance, and pinyon and juniper cover is typically somewhere between 10% and 30%. In Phase III woodlands, trees dominate with cover typically exceeding 30% (Miller et al. 2000, 2005, 2008).

2. METHODS

We conducted a systematic review of the literature to evaluate and synthesize the effects of mechanical treatments of pinyon-juniper and sagebrush communities on several environmental response categories. These included vegetation cover and diversity, fuel loads, wildlife and its habitat (e.g., abundance and productivity), and ecosystem function (i.e., soil stability, watershed productivity, and carbon sequestration).

We investigated the effects of a number of different mechanical treatment methods on our response categories. We conducted a systematic review of the published literature to evaluate and synthesize the effects of mechanical treatments of pinyon-juniper and sagebrush communities on the following environmental response categories: herbaceous cover, wildlife, soil stability, carbon sequestration and watershed productivity. We used the ScienceDirect and Google Scholar online search engines, using keywords “pinyon”, “pinon”, “juniper”, “sagebrush”, “vegetation treatment”, “effects”, “wildlife habitat”, “vegetation”, “fuels management”, “erosion”, “soil”, “hydrology”, and “carbon sequestration”. This search, along with other serendipitous encounters, identified over 300 sources. Sources were distributed across a geographically extensive region and included a variety of plant communities.



Greater sage-grouse. (Photo: Tatiana Gettelman/USGS)

We created summary charts that describe the number of treatments with positive, negative, or non-significant effects for the following categories:

- Response of vegetation cover (i.e., perennial grasses, perennial forbs, and non-native annual grasses) to mechanical sagebrush treatment and to mechanical pinyon-juniper treatment,
- Response of wildlife (other than sage-grouse) to mechanical sagebrush treatment,
- Response of sage-grouse to mechanical sagebrush and pinyon-juniper treatment,
- Response of soil erosion related variables to mechanical treatment, and
- Response of hydrological related variables to mechanical treatments.

We include results of Bombaci and Pejchar (2016) for the response of wildlife to mechanical pinyon-juniper treatments, as it provided a thorough summary of this literature. To create the summary charts, we prioritized peer-reviewed research and only included studies that tested for significance between treatments and controls, where the control was either pre-treatment data or an adjacent non-treated area compared to the treated area. We considered one data point in the summary chart as the difference between the treatment and the control for one treatment method and/or in one sampling period and/or in one sampling site. Therefore, research that tested multiple treatments, sites, or years resulted in more than one data point in the summary charts. For vegetation response, if results (e.g., vegetative cover) could not be delineated into our vegetation cover categories (e.g., perennial grasses, perennial forbs, and non-native annual grasses), we did not include those data points. Perennial grasses and forbs included non-native species that are commonly found in seed mixes such as crested wheatgrass (*Agropyron cristatum*), intermediate wheatgrass (*Thinopyrum intermedium*), and small burnet (*Sanguisorba minor*). Where there were significant interactions with non-mechanical treatments (e.g., burns), we did not include those data points. We considered a p-value < 0.05 as the significance criteria.

3. VEGETATION TREATMENT OBJECTIVES

3.1 VEGETATION STRUCTURE

3.1.1 Herbaceous Cover and Diversity in Treated Pinyon-Juniper Woodlands

The goal of many pinyon-juniper woodland reduction treatments is to increase perennial grasses and forbs (Bureau of Land Management 1991, 2017; Healthy Forests Restoration Act of 2003). However, the majority (63%) of the total data points in the summary chart report no significant effect of treatments on these functional groups (Figure 1). Perennial grasses had more positive responses (32%) than negative (4%), but most (64%) were not significant. Similarly, perennial forbs showed positive responses (23%) more than negative (10%), but most were not significant (66%). An unintended consequence of pinyon-juniper treatments is the increase of non-native annual plants, particularly cheatgrass. Treatments either had positive (50%) or not significant (50%) effects on these plants.

Evaluating the response of perennial grasses, forbs, and invasive plants to pinyon-juniper treatments appears to be highly complex and dependent on many factors. These variables include the type of pinyon-juniper woodland community (Persistent Pinyon-Juniper

Woodlands, Pinyon-Juniper Savannas, or Wooded Shrublands) (Baker and Shinneman 2004; Romme et al. 2009); the degree of pinyon-juniper expansion into sagebrush communities or expansion within woodlands (e.g., Roundy et al. 2014a); whether the treatment type is chaining, mastication, or lop and scatter (e.g., Murphy and Romanuk 2012); the cover of shrubs and herbaceous species (including non-native vs. native) that exist before treatment (Bates et al. 2005; Rossman et al. 2018; Roundy et al. 2014a, 2014b; Stephens et al. 2016; Williams et al. 2017; Young et al. 2013a); which seeds reside in the seed-bank; whether the site is seeded afterwards and what the seed mix is comprised of (e.g., Roundy et al. 2014a); scale of the analysis (e.g., Rossman et al. 2018); length of post-treatment rest period from grazing (e.g., Bristow 2010); and the grazing regime once it commences. Yet another factor is the length of time after treatment the study was conducted. For example, Bates et al. (2005, 2007) found that long-term (i.e., 13 years) monitoring was required to document the dominance of perennial grasses after pinyon-juniper treatments. On the other hand, a study of 30-year-old pinyon-juniper chaining treatments in Nevada found that as cover of woody species increased, the cover of herbaceous species decreased, and treat-

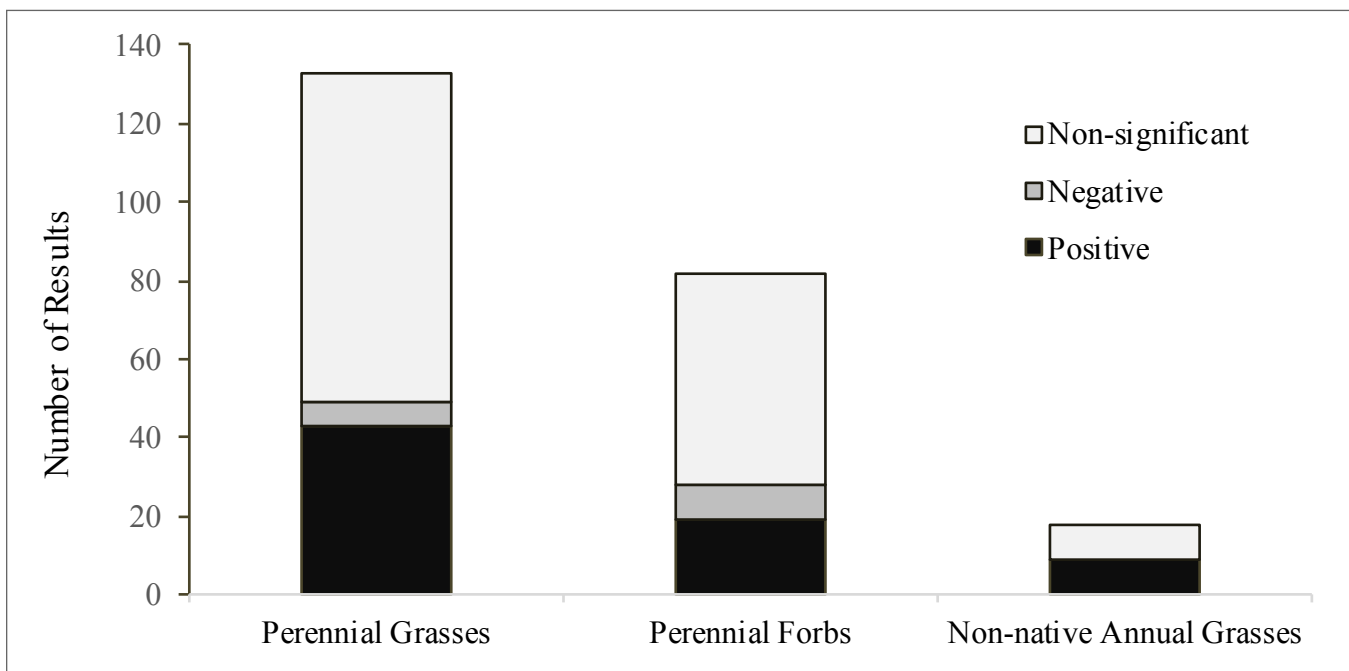


Figure 1. Number of pinyon-juniper treatment study results within herbaceous vegetation categories that found positive, negative, or non-significant results. The appendix lists all of the studies that contributed data points to this summary chart, and the response variables measured.

ment efficacy declined over time (Bristow 2010). Bates et al. (2017a) saw a similar response to a 25-year-old hand thinning treatment.

When significant results were reported, they were more often positive than negative. For example, Vaitkus and Eddleman (1987) found herbaceous production doubled after the removal of western juniper (*Juniperus occidentalis*) in Oregon. Provencher and Thompson (2014) compared pinyon-juniper treatments (i.e., chaining, bulldozing imitating chaining, lop-pile-burn, lop-and-scatter, feller-buncher and chipper, and mastication) in encroached black sagebrush communities in Nevada and Utah. After 4 years, forb cover increased in all treatments except mastication. Other studies find the mastication treatment method, in which trees are shredded in place, is effective in increasing herbaceous vegetation. Ross et al. (2012) reported that lop-and-scatter and mastication treatments increased herbaceous vegetation cover. Similarly, Young et al. (2013a) found mastication treatments increase perennial and annual grasses. Fornwalt et al. (2017) found that total species richness and cover of graminoids and native forbs in mulched sites were greater than untreated sites over 6 to 9 years. In Colorado, Stephens et al. (2016) found that three mechanical methods (i.e., chaining, roller-chopping, and mastication) each increased understory vegetation. Havrilla et al. (2017) also compared herbaceous cover in several treatment types over 6 years; mastication, lop-and-slash piled then burned, and lop-and-scatter followed by broadcast burn. They found that perennial herbaceous plants increased in all treatment plots. Havrilla et al. (2017) concluded that mastication with seeding is an effective method to remove fuels and recover the herbaceous layer with native plants.

To evaluate the general response of understory vegetation to tree canopy removal in conifer-encroached shrublands, Miller et al. (2014c) set up a region-wide study that measured treatment induced changes in understory cover and density. Eleven study sites located across four states in the Great Basin were established as statistical replicate blocks, each containing fire, mechanical, and control treatments. Different cover groups were measured prior to treatment and for three years thereafter. Tall perennial grass cover increased in the mechanical treatment in the second and third year. Non-native grass and forb cover did not increase in the mechanical treatments in the first year but increased in the second and third years. Perennial forb cover increased in the mechanical treatments. The recovery

of herbaceous cover groups was determined to be from increased growth of residual vegetation.

However, other studies detected decreases or no significant change in herbaceous understory following treatment. In New Mexico, Rippel et al. (1983) found that in response to pinyon-juniper chaining treatments, some shrub species increased but grass and forb biomass and cover declined until they were lower than the control plots. In Utah, Frey (2010) found that the herbaceous cover in pinyon-juniper chaining treatment sites was less than half of that in reference sites. Rubin and Raybal (2018) reported that two to three years after a mastication treatment untreated sites had four times the herbaceous cover of native and non-native plants than treated sites, although the treated sites had a much higher diversity of grasses and non-native forbs.

Pinyon-juniper treatments can lead to an increase in invasive and/or annual plants, particularly cheatgrass (Evans and Young 1985, 1987; Havrilla et al. 2017; Monaco et al. 2017; Provencher and Thompson 2014; Stephens et al. 2016). Cheatgrass can outcompete the forbs and grasses the treatment was intended to increase (Bates et al. 2007). Many studies found that mechanical treatments in pinyon-juniper woodlands may increase herbaceous production, but the increase in invasive, annual plants may not necessarily improve overall ecosystem conditions. For example, Vaitkus and Eddleman (1987) concluded that after juniper removal in Oregon, herbaceous production doubled but much of the increase came from annual plants. Davis and Harper (1989) reported significant increases in weedy annuals on chained treatments in Utah. Owens et al. (2009) observed increases in cheatgrass following lop and scatter/pile burn and mastication treatments in Colorado. Ross et al. (2012) found that in Utah cheatgrass was not present on control sites but it comprised more than 18% cover on lop and scatter/pile burn treatments and between 11% and 18% cover on mastication treatments. Bybee et al. (2016) found that the fine woody debris produced by mastication increased cover of both native and non-native herbaceous plants. Studies in Utah showed that the fine woody debris produced by shredding pinyon and juniper also has an effect on soil microbial activity and nutrient availability deep into the soil profile, even far away from the treatment site (Aanderud et al. 2017). This in turn influenced both native and non-native plants on a species-specific basis. For example, cheatgrass and some native grasses increased, while bluebunch wheatgrass (*Pseudoroegneria spicata*) decreased. The positive influence of fine



Left: Greenville Bench, Utah pinyon-juniper woodland in 2006, prior to a bullhog mastication treatment. Right: A photo of the same area six years after treatment, showing woody debris and substantial grass growth. While most of the total grass cover was native perennial, cheatgrass cover represented 40% of the total by 2013. (Photos: Sagebrush Steppe Treatment Evaluation Project [SageSTEP])

woody debris diminished over time for cheatgrass but increased for native grasses.

Though not focused on pinyon-juniper forest types, a meta-analysis that pooled 32 studies of mechanical thinning treatments combined with fire in coniferous forests conducted by Willms et al. (2017) found that the most consistent mechanical treatment effect was an increase in non-native species, which they ascribed to the ground disturbance associated with treatments. Seeding perennial native species after treatment is often recommended by many researchers and practitioners and can, in the short-term and sometimes long-term, help decrease the cover of invasive annual plants and grasses (Bates et al. 2011; Bates et al. 2014a, 2014b; Havrilla et al. 2017; Roundy et al. 2014a). On the other hand, post-treatment seeding is not a guarantee of reducing invasive herbaceous cover. Many researchers report low seeding success rates, particularly in arid and semi-arid sites (e.g. Beyers 2004; Wilder et al. 2018).

Bates et al. (2005, 2007) proposed that post-treatment annual grass dominance reported in many other studies was due to inadequate perennial grass density on the site before woody plants were removed and speculated that a pre-treatment density of one to two native

bunchgrasses per square meter was adequate to prevent cheatgrass dominance in low elevation Wyoming big sagebrush sites in Nevada. In the Great Basin, two to three perennial bunchgrasses per square meter is indicated (Michael Pellent, personal communication, January 2019). Roundy et al. (2014a) found that mechanical treatments had higher cheatgrass cover in pinyon-juniper woodlands with greater tree cover (i.e., more advanced forest succession), and sites with high cheatgrass cover before treatment was related to high cheatgrass cover after treatment. They also found that maintaining perennial cover could resist cheatgrass dominance, especially on warmer sites which are more susceptible to being dominated by invasive or annual plants.

The substantial variability in outcomes as illustrated by our summary chart might explain the preponderance of non-significant results in the studies we reviewed. With these confounding factors affecting the response of herbaceous vegetation, most authors in our review caution against applying their results to other systems, especially in other geographic regions. On the other hand, it is possible that the high number of non-significant results might not just be an artifact of the data; it could simply be that mechanical pinyon-juniper removal in order to

elicit understory response does not often produce the desired results.

3.1.2 Herbaceous Cover and Diversity in Treated Sagebrush Communities

Today sagebrush is not being cleared on our western rangelands at the rate it was in the 1940s to 1970s. However, many mechanical sagebrush treatments still occur for wildlife habitat improvement, fuels reduction, and other justifications (Pilliod et al. 2017). As with pinyon-juniper treatments, often the primary goal is to increase perennial grasses and forbs. However, the result of our summary chart shows that over half (56%) of the data points show no significant effect of sagebrush treatments on these functional groups (Figure 2). Perennial grasses had more positive responses (33%) than negative (8%), but most (58%) were not significant. The positive response of perennial forbs (23%) was slightly larger than the negative response (19%), but most responses were not significant (58%). An unintended consequence of sagebrush treatments is the increase of non-native annual plants, particularly cheatgrass. Treatments either had positive (48%) or not significant (52%) effects on these plants.

The response of perennial grasses, perennial forbs, and non-native annual plants to mechanical sagebrush treatments appears to be highly complex and dependent on many factors. Treatment results can vary depending on site conditions and factors such as sagebrush taxon present, elevation (e.g., Wilder et al. 2018), treatment

methods (e.g., Dahlgren et al. 2006), the cover of herbaceous species that exists before treatment (e.g., Chambers et al. 2017), which seeds reside in the seed-bank (e.g., Monaco et al. 2018), whether the site is seeded afterwards and what the seed mix is comprised of (e.g., Davies et al. 2014a; Monsen 2004), length of post-treatment rest from livestock, and the grazing regime once grazing commences (e.g., Wilder et al. 2018).

Some studies have shown short-term positive herbaceous responses to mechanical removal of sagebrush, especially if the removal is accompanied by seeding of herbaceous species (Monaco and Gunnell unpublished data, MS in prep; Wilder et al. 2018 and others). However, those successes may be limited (Svejcar et al. 2017; Wilder et al. 2018) or short-lived (Knutson et al. 2014; Peterson 1995; Pyke et al. 2013; Pyke et al. 2014; Svejcar et al. 2017). Other studies showed no effect of treatment on herbaceous response (Blaisdell 1953; Clary et al. 1985; Davies et al. 2011; Stringham 2010; Summers and Roundy 2018; Wamboldt et al. 2001). Some sites experienced reduced productivity or diversity of native grasses and forbs after treatment (Pechanec and Stewart 1944; Wamboldt et al. 2001; Watts and Wamboldt 1996). This was particularly evident in degraded sites with low resilience to disturbance (Chambers et al. 2014; Chambers et al. 2017; Davies et al. 2012) and where current land use is perpetuating degraded understory conditions (Bestelmeyer et al. 2015; Morris and Rowe 2014).

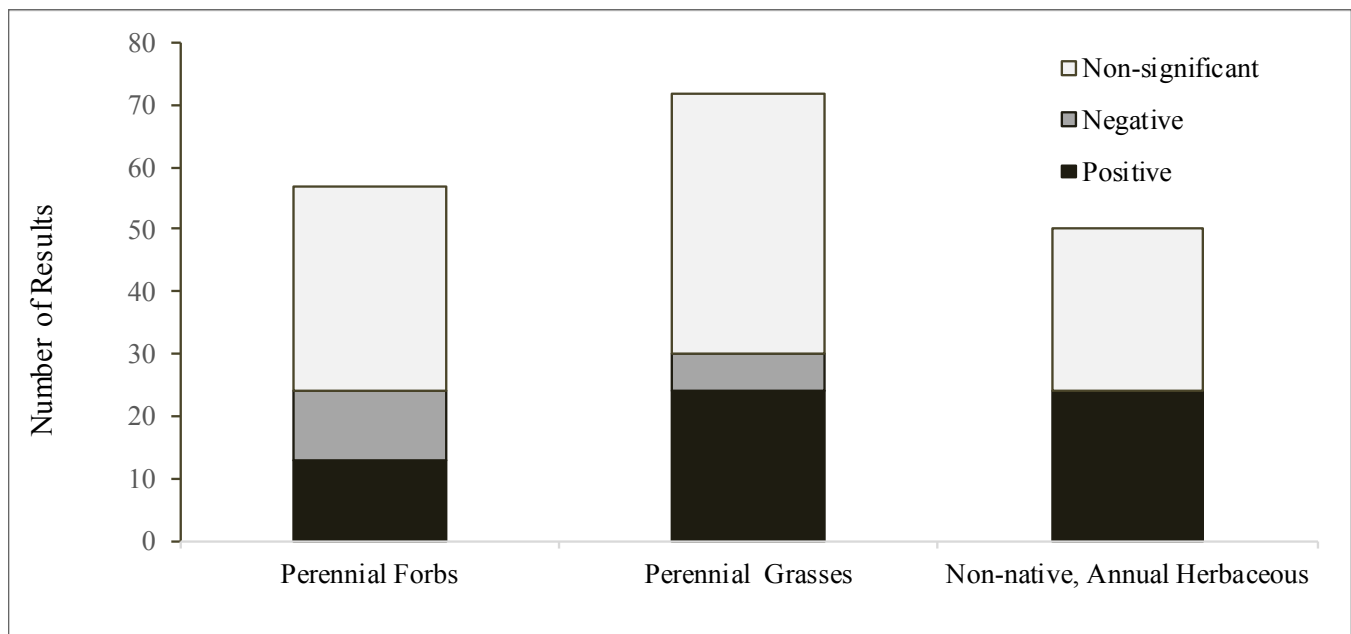


Figure 2. Number of sagebrush treatment study results within vegetation categories that found positive, negative, or non-significant results. The appendix lists all of the studies that contributed data points to this summary chart, and the response variables measured.



Left: Untreated sagebrush community in Grand Staircase-Escalante National Monument. Right: An adjacent parcel two years after treatment. (Photos: Laura Welp)

Wilder et al. (2018) conducted a meta-analysis of data from unpublished post-treatment monitoring reports from sagebrush treatments implemented by the Utah Watershed Restoration Initiative. This group is a coalition of private, state, and federal entities conducting tens of thousands of acres of vegetation treatments throughout Utah. By computing effect sizes, Wilder et al. tested for effects of sagebrush reduction on seeding success over short (1 to 4 years) and long (5 to 10 years) post-treatment periods. The study found that across sites, seeded perennial grasses increased over time but forbs declined. Some non-native species increased significantly more than natives. Results were likely influenced by seedbed conditions and site characteristics, particularly elevation, and the authors stressed that this should be taken into account during restoration planning. The authors found that poor performance of forbs after treatment is a common occurrence, especially in lower elevation sites that tend to be warmer and drier. This was attributed to possible overutilization by native and non-native grazers during critical periods of plant growth. However, while forbs showed a more favorable response over time at the higher elevation sites, both native and non-native grasses exhibited greater increases in cover and frequency at lower elevation sites, where resiliency is typically thought to be lower (Archer and Predick 2014; Chamber et al.

2017). We anticipate valuable information from another meta-analysis on a similar dataset (Riginos et al. in review). That study will analyze data from 94 sagebrush treatments in Utah to assess short- (1 to 5 years post-treatment) and long-term (6 to 12 years post-treatment) overall responses of sagebrush, perennial and annual grass and forb, and ground cover to sagebrush reduction treatments, whereas Wilder et al. (2018) was focused on seeding success.

Two papers report on herbaceous response to mechanical treatment methods in other types of southwest shrublands (Archer et al. 2011; Archer and Predick 2014). While these reviews focused on snakeweed (*Gutierrezia* spp.), mesquite (*Prosopis* spp.), and creosote bush (*Larrea tridentata*), the results may be relevant to sagebrush communities. Archer et al. (2011) found more than 80% of studies had positive herbaceous responses following brush management, especially in the range of 30 to 70 cm mean annual precipitation. The positive effects lessened as study sites decreased in elevation, however. The time since treatment was an important factor in assessing degree of success. The median first-year response of herbaceous vegetation was highly variable, with half of treatment sites responding positively and half negatively. After year two, a positive response became more consistent



A post-treatment seeded area near Moab colonized by invasive cheatgrass (Photo: Laura Welp)

and peaked in year five. The positive response decreased after 7 and 8 years but remained positive. The authors found that shrub removal treatments typically have neutral (30% of data points exhibiting <10% change) to positive (60% of data points exhibiting >10% increase) effects on grass and forb diversity. However, the few long-term data sets available suggest this response is relatively short lived (<15 year). The length of time that management treatments continue to have a positive effect on the herbaceous layer varied widely by treatment type, shrub species, effectiveness of the initial treatment, composition of the herbaceous vegetation, and soil properties.

As with pinyon-juniper treatments, many studies show that if invasive, annual plants are present on the site prior to sagebrush treatments, the cover of these species often increase after treatment. For example, Prevey et al. (2010) found three to four times more non-native herbaceous species in sites where sagebrush was removed than in undisturbed sites.

Researchers are focusing on ways to increase resistance of treatment sites to expansion of invasive, annual plants. Many authors underscore the need to seed after sagebrush treatments to prevent invasive, annual plants taking hold, particularly if the site is highly degraded

with a large amount of bare ground (Davies et al. 2018 and others). For example, Davies et al. (2012) found that mowing alone on degraded sagebrush sites does not increase herbaceous cover but facilitates the establishment of non-native grasses and forbs. They concluded that successful mowing treatments in degraded sites must be accompanied by seeding, weed reduction, or other management activities. Similar conclusions were drawn from long-term studies by Roundy et al. (2018) working with the Sagebrush Steppe Treatment Evaluation Project (SageSTEP). (SageSTEP is funded by the U.S. Joint Fire Science Program, Bureau of Land Management, and National Interagency Fire Center www.sagestep.org.) There is a high risk of cheatgrass invasion after treatment, especially in warmer and drier sites. To prevent increase of cheatgrass after treatment, restoration and maintenance of perennial herbaceous species should be facilitated with revegetation and appropriate post-treatment livestock grazing (Roundy et al. 2018).

The treatment success is also affected by the seed mix. Diverse seed mixes are better able to stabilize soils and prevent spread of invasive plants, but seed mixes that include non-natives may outcompete and impede native species (Davies et al. 2014a; Knutson et al. 2014; Wilder et al. 2018). Wilder et al. (2018) found that success of both native and non-native seeded species varied depending on seeding method (e.g., depth), timing, and post treatment grazing and browsing pressure.

Reisner et al. (2013) found that limiting the size and connectivity of gaps between vegetation in sagebrush communities is important for resistance to non-native species invasion. Biological soil crust limits non-native species cover in this way. They also suggest that cattle grazing, by reducing bunchgrasses and trampling biological soil crust, reduces resistance to non-native species invasion. Similar findings were found by Condon and Pyke (2018). They recommend that managers consider how treatments will impact these two functional groups when conducting restoration activities in sagebrush communities.

3.2 FUELS MANAGEMENT

While prescribed fire is often used as a treatment method in pinyon-juniper woodlands and sagebrush, it is not the focus of this literature review. However, land managers often use mechanical treatments to attempt to lighten fuel loads and for post-wildfire recovery efforts. This section describes wildfire in terms of how

it relates to the application and rationale for mechanical treatments in pinyon-juniper woodlands and sagebrush communities.

3.2.1 Pinyon-Juniper Systems

Land managers often prescribe pinyon-juniper treatments to lessen the likelihood of large, devastating crown fires, especially around populated areas or structures (e.g., Healthy Forests Restoration Act). Many researchers advocate treating pinyon-juniper woodlands at low- to mid-tree dominance index (i.e., Phases I and II). This will retain the shrubs on a site and increase ecosystem resilience and resistance by promoting herbaceous cover (Roundy et al. 2014a; Williams et al. 2017; Young et al. 2013a). They also recommend treating with methods such as cutting or mastication rather than chaining because there is less soil erosion and better seedling establishment.

However, climate data are now being incorporated into many pinyon-juniper treatment projects, and this is helping practitioners better understand an important component of fire in these systems today. Keyser and Westerling (2017) used 5-year climate variables to predict where high severity fires occur so that managers

can conduct more targeted fuels reductions. Several studies have found that climatological factors are more correlated with ignition of wildfires than amount of biomass in trees. Dennison et al. (2014), Holden et al. (2007), Westerling (2016), and Westerling et al. (2006) found that drying trends over the last 20 years had a greater influence on fire activity in dry pine forests, including pinyon-juniper woodlands, than fine fuels and biomass production. They concluded that fire risks are more strongly associated with increased spring and summer temperatures and an earlier spring snow-melt. However, while this is true of the amount of area burned or number of large fires, this may not be the case in terms of fire severity, in which fuel accumulation and continuity may be very important (Douglas Shinneman, personal communication, November 2018).

Surface disturbance associated with mechanical treatments facilitates cheatgrass expansion and may actually serve to increase incidence of fire (Roundy et al. 2014a). Young et al. (2015) found that removing trees reduced canopy fuel loads but surface fuel loads increased. The fine woody debris produced by mastication has been shown to increase the herbaceous layer,



Spruce Mountain, Nevada about four years after a juniper mastication project. Woodchip piles appear to have suppressed some forbs and native bunchgrass growth while non-native cheatgrass and pennyroyal were not suppressed. (Photo: Laura Cunningham)

including flammable cheatgrass (Aanderud et al. 2017). Redmond et al. (2013) also found an increase of fuels in chained treatments after 20 to 40 years. The previous section of our review details examples of mechanical pinyon-juniper treatments that increase both native and non-native herbaceous understories, all of which have the potential to increase not only post-treatment fuel loads but to potentially create conditions with fuel loads higher than they were historically, depending on whether the type of woodland is Persistent Pinyon-Juniper Woodland, Pinyon-Juniper Savanna, or Wooded Shrubland. Bates and Davies (2017) have speculated that burning slash in late fall to early spring and including a revegetation component on warmer sites with depleted understories may help. Young et al. (2015) also recommended conducting cool-season prescribed fires after treatments to reduce surface fuels. However, they note that the presence of cheatgrass at the site may impact success of this method. It should be noted that none of these mitigation practices appear to have been tested.

3.2.2 Sagebrush Systems

Sagebrush treatments are sometimes applied to re-establish what are thought to be natural fire intervals (staff of Grand Staircase-Escalante, personal communication, October 2018). Land managers sometimes estimate that sagebrush must be thinned by fire every 10 to 40 years to remain healthy and to reduce fire risk (e.g., Bunting et al. 1987; Davies et al. 2009b). However, depending on whether the sagebrush type is Wyoming big sagebrush or mountain big sagebrush, 10 to 40 year fire cycles may not be supported by the published science (see section 1.3.2).

Sagebrush treatments are effective at reducing the height, mass, and continuity of canopy fuels, but they may alter fire behavior in other ways (Archer and Predick 2014). Surface disturbance caused by treatments may promote invasive and annual plants, leading to increased fine fuels and fire (Bates and Davies 2017; Fornwalt et al. 2017; Roundy et al. 2014a; Williams et al. 2017; Young et al. 2013a). Rau et al. (2014) found that after treatments, dry sites with sandy soils or those with low soil-water holding capacity were most vulnerable to increases in invasive and annual plants. Chambers et al. (2014) found similar results. Their study indicated that resistance to invasive and annual plants is influenced by soil temperature and moisture regimes. Cool mountain big sagebrush sites were more resistant to invasive and annual grasses than warm, dry Wyoming big sagebrush sites.



A sagebrush treatment by Dixie harrow in Grand Staircase-Escalante National Monument. (Photo: Ray Bloxham)

Mean annual precipitation and temperature, soil texture, cover of perennial native herbaceous species, gaps between perennial plants, and other fuel sources should be considered by managers trying to prioritize sagebrush sites for treatment and select appropriate treatments to minimize invasive and annual plants (Chambers et al. 2014; Rau et al. 2014). Some researchers speculate on ways to reduce the risk of non-native invasion following sagebrush treatments. For example, Chambers et al. 2014 found that having at least 20 percent perennial native herbaceous cover on the site before treatment may minimize invasives in cooler, moister areas.

Land managers have recently started to construct a vast network of fuel breaks to reduce wildfire in sagebrush communities, especially in the Great Basin where cheatgrass-fueled fires have destroyed a significant amount of greater sage-grouse habitat. Shinneman et al. (2018) reviewed these projects and noted that, although there is anecdotal evidence supporting the practice, not enough data have been collected to verify the contention that fuel breaks reliably reduce wildfire in sagebrush communities. Furthermore, these projects may cause resource impacts in themselves. For example, construction may introduce more cheatgrass, impact plant communities by planting non-native wheatgrass, construct more roads, and create edge habitat that

may disadvantage some rare wildlife species such as pygmy rabbit. The authors discuss the need to balance habitat loss due to wildfire with habitat loss due to the creation of hundreds of kilometers of fuel breaks.

3.3 WILDLIFE HABITAT

Vegetation treatments are also conducted to improve wildlife habitat, particularly with the emergence of greater sage-grouse (*Centrocercus urophasianus*) as a conservation concern. Greater sage-grouse are considered an umbrella species; their conservation protects many other species that co-occur in sagebrush communities or are considered sagebrush obligates themselves (Carlisle et al. 2018a; Donnelly et al. 2017; Rowland et al. 2006). However, less is known about the effects of pinyon-juniper and sagebrush treatments on other species (Rich et al. 2005). Managing these communi-

ties to benefit wildlife is complex. Wildlife responses vary widely based on species interactions, temporal and spatial scales of treatments, and other variables (e.g., Fulbright et al. 2018). Wildlife species often specialize for specific habitat conditions. Changing habitat conditions often creates winners and losers. This aspect is an additional challenge when summarizing the mechanical treatment literature. As a general rule of thumb, maintaining landscape mosaics (heterogeneity) at the proper spatial and temporal scale or scales provides for maximum diversity and reduces disturbance patch size.

3.3.1 Pinyon-Juniper Treatments

Overall, studies of pinyon-juniper treatments to manage wildlife habitat have varied results. Bombaci and Pejchar (2016) provided a thorough review of studies that evaluated responses of wildlife to pinyon-juniper treatments (Figures 3 and 4). They found there was not

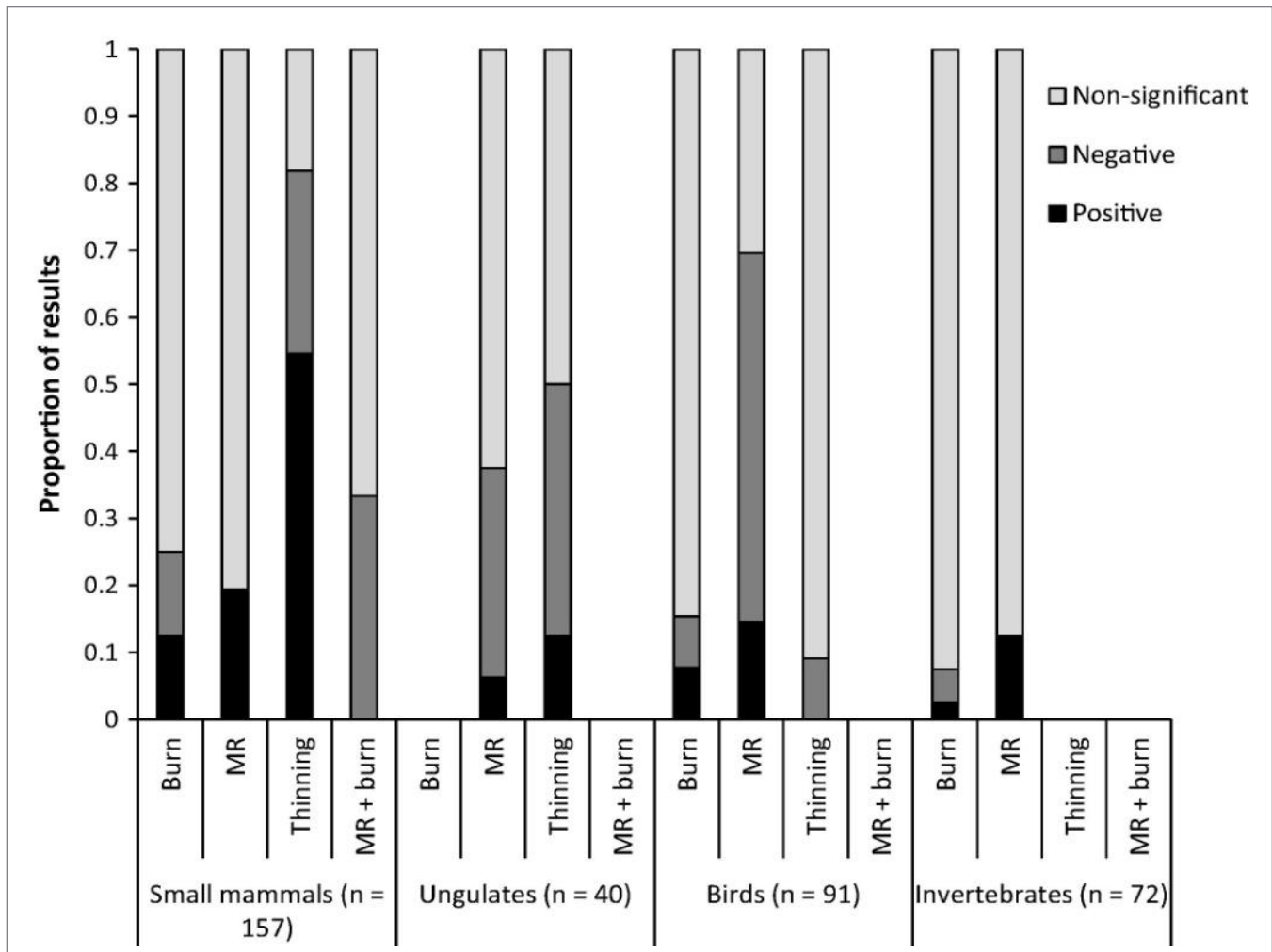


Figure 3. Proportion of study results within taxonomic groups and treatment methods documenting positive, negative, or non-significant responses to woodland reduction. Burn = prescribed fire; MR = mechanical removal (i.e., bulldozing, chaining, cutting, mowing, hydro-axing, roller-chopping, and uprooting); Thinning = any treatment method that retains some standing trees; MR + burn = mechanical removal and prescribed fire (reprinted from Bombaci and Pejchar 2016, with permission).

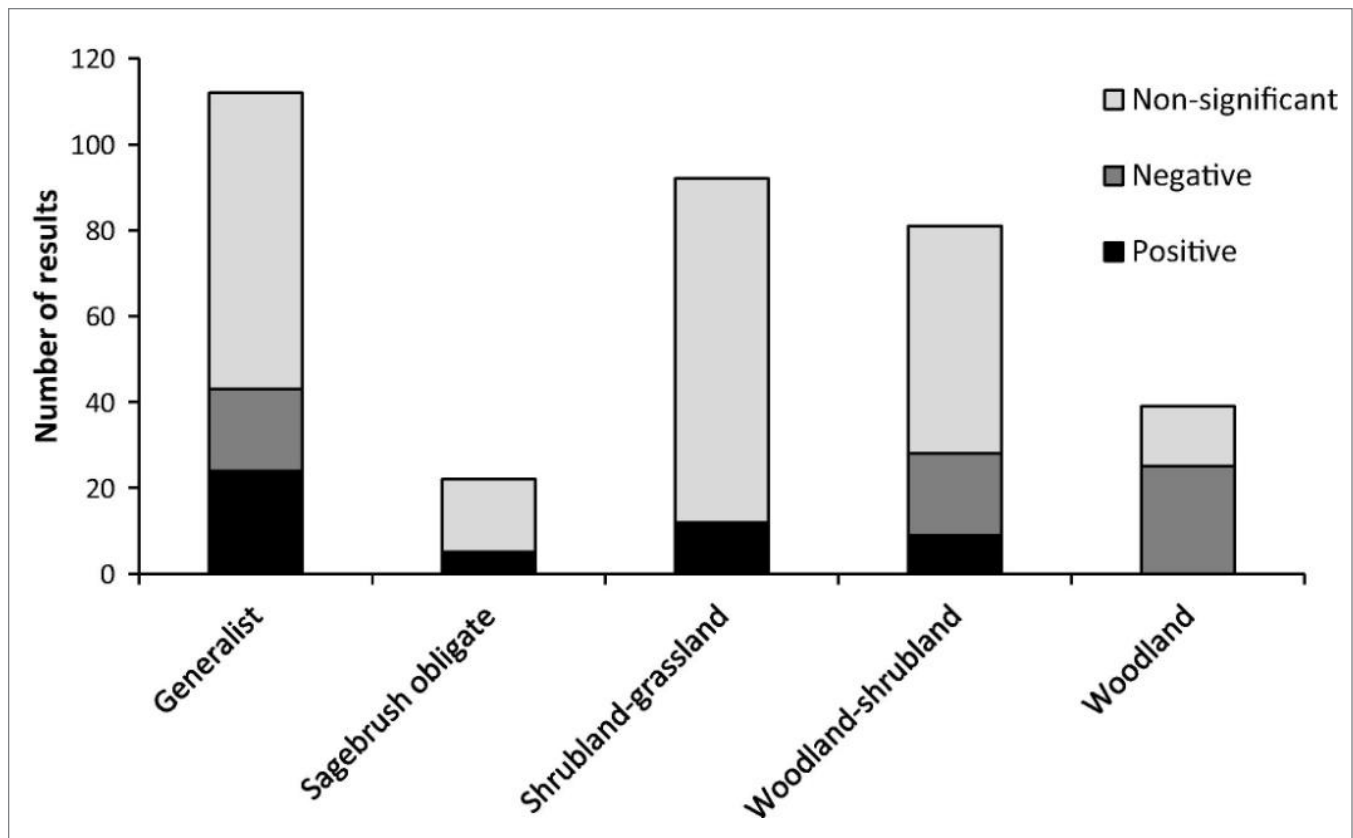


Fig. 4. Number of study results within habitat functional groups documenting positive, negative, or non-significant responses to woodland reduction treatments (reprinted from Bombaci and Pejchar 2016, with permission). The appendix lists all of the studies that contributed data points in Figures 3 and 4, and the response variables measured.

a consistent positive or negative response from wildlife overall and most study results were non-significant. Pinyon-juniper treatments are assumed to benefit sagebrush obligates, but evidence is lacking. The authors called for additional long-term research on larger study sites in order to better understand wildlife responses, especially for sagebrush obligates.

In addition to the generalized results for wildlife, Bombaci and Pejchar (2016) also reported results by taxon. Most of the few studies available on invertebrate responses to treatments were non-significant or inconsistent. For example, Kleintjes et al. (2004) found that invertebrate species richness and abundance increased significantly in cut and slash treatments, but McIver and Macke (2014) found that species responses to pinyon-juniper removal varied significantly by species.

For bird species that are pinyon-juniper obligates, there is a positive relationship with live trees and cover (Balda and Masters 1980; Francis et al. 2011). When pinyon-juniper woodlands were removed mechanically, most bird species, whether pinyon-juniper obligate or not, generally responded negatively and abundance was reduced (Bombaci and Pejchar 2016). However, in

most studies of thinning treatments there was an overall non-significant response and the retention of some tree cover may sustain birds. An exception was Crow and Van Riper (2010), who found that pinyon-juniper thinning treatments correlated to lower abundance for two woodland-associated birds (i.e., gray vireo [*Vireo vicinior*], chipping sparrow [*Spizella passerine*]). Bombaci et al.'s 2017 study supported the general findings of Bombaci and Pejchar (2016). They found overall bird use was higher within non-treated areas versus treatments, which was related to greater pinyon-juniper cover.

Sagebrush-obligate bird species would be expected to benefit when pinyon and juniper trees are removed from sagebrush stands. Bombaci and Pejchar (2016) found that, with the exception of greater sage-grouse (see section 3.3.3), most studies report that sagebrush-obligate bird responses were non-significant or negative to pinyon-juniper removal. Only Crow and Van Riper (2010) found thinning treatments benefitted a sagebrush obligate (Brewer's sparrow).

Studies of bird response to pinyon-juniper removal were generally conducted within a few years post-treat-

ment, and those results reflect short-term responses. But there are some long-term studies. For example, Gallo and Pejchar (2016) found significant differences in bird abundance between sites that were historically (40 and 60 years prior) chained versus sites that were undisturbed. While untreated pinyon-juniper woodlands had higher species richness, they were comprised of woodland-obligates. Treated sites contained greater abundance of shrubland-obligate species such as Brewer's sparrow. Holmes et al. (2017) found during their three-year study that sagebrush-associated species were positively correlated with pinyon-juniper treatments that were completed by mechanical hand removal. Brewer's sparrow, green-tailed towhee (*Pipilo chlorurus*), and vesper sparrow use increased when pinyon-juniper trees were removed, but gray flycatcher (*Empidonax wrightii*) use declined in treatment sites. This was attributed to their preference for habitats with juniper and taller sagebrush. Knick et al. (2017) studied the impact of pinyon-juniper reduction treatments on bird communities at almost 300 pinyon-juniper removal sites over seven years. They found that bird communities that had stable environments (<5% woodland reduction) experienced little change over the seven years post-treatment. In contrast, there were indications that bird communities at the 80 sites with >5% woodland reduction were shifting away from birds with woodland affinities towards more ecotone or grass- and shrub-associated species.

Small mammal responses to treatments that completely removed pinyon-juniper trees were generally non-significant (Figure 3). More studies found small mammals positively responded to thinning treatments; an important benefit was the increase in cover created by the slash piles left behind. For small mammals that prefer grassland habitats, they responded positively to removal of the pinyon-juniper canopy (i.e., bulldozed treatments). In general, many studies did not find significant responses to pinyon-juniper burning and mechanical removal treatments. In 2017, Bombaci et al. found no difference in small mammal (i.e., least chipmunk [*Tamias minimus*] and deer mouse [*Peromyscus maniculatus*]) abundance among pinyon-juniper treatments or control sites. They concluded the results were related in part to grass and herbaceous cover in sites.

Gallo et al. (2016) assessed mammal use (i.e., camera detections) between historically chained sites that retained their shrub layer afterwards and non-treated sites. Of the eight species analyzed, bobcat (*Lynx rufus*), mountain lion (*Puma concolor*), American black



Managing habitat for wildlife is complex. Species often specialize for specific habitat conditions, and what benefits one species may be a detriment to another. This argues against large expanses being treated with one method that creates a single homogenized vegetation community. Above: Gray vireo (Photo: Dominic Sherony)

bear (*Ursus americanus*), golden-mantled ground squirrel (*Callospermophilus lateralis*), and rock squirrel (*Otospermophilus variegatus*) responded negatively to chaining treatments. Chipmunk (*Tamias* spp.), mountain cottontail (*Sylvilagus nuttallii*), and coyote (*Canis latrans*) showed no significant response to the chaining treatments. Hamilton et al. (2018) found pinyon-juniper thinning treatments increased small mammal species richness, biomass, and density. Their treatments had a negative impact on one species: the pinyon mouse (*Peromyscus truei*), a pinyon woodland obligate.

Bombaci and Pejchar (2016) found that mechanical treatments have a mostly negative or non-significant effect on mule deer and elk. This was attributed to the important cover pinyon-juniper woodlands provide. Some studies found specific scenarios where deer and elk responded positively. The use of chained treatments was greater in the spring (Howard et al. 1987), smaller reduction treatments embedded within non-treatment areas were utilized more (Short et al. 1977), and mule deer fawns had greater survival in cleared (and managed) treatments (Bergman et al. 2015); all of which infer the importance of balance between cover, edge effects, and available forage on wild ungulates.

3.3.2 Sagebrush Treatments

A common goal of mechanical treatments in sagebrush communities is to increase vegetation diversity, particularly of grasses and forbs for forage and cover (Lutz et al. 2003; Winward 1991). Many researchers have assessed the impacts of mechanical sagebrush treatments

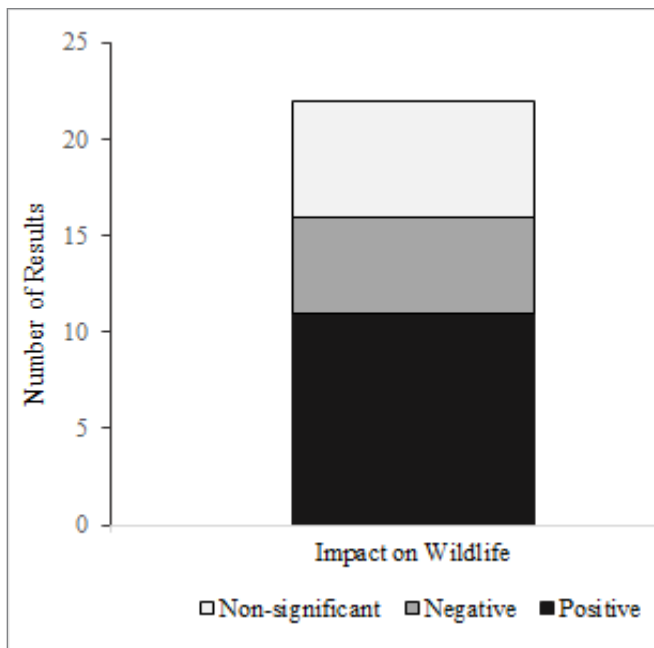


Figure 5. Number of sagebrush treatment study results that had positive, negative, or non-significant response on wildlife. The appendix lists all of the studies that contributed data points to this summary chart, and the response variables measured.

on wildlife, particularly for sagebrush obligates. We address mechanical treatment (of both sagebrush and pinyon-juniper woodlands) effects on sage-grouse in a separate section, below. The literature finds sagebrush treatments have varied impacts on wildlife, including big game species such as elk (Figure 5). Some of this variability may be ascribed to scale. Sagebrush treatments that focus on fuels removal may have different effects than larger treatments focusing on habitat restoration (Michael Pellant, personal communication, January 2019). Of a total of 22 data points, 11 (50%) showed a positive response to treatment by wildlife, five points (22%) showed negative responses and six (27%) were not significant.

Many studies focus on the effects of mechanical sagebrush treatment on bird species, in particular neotropical migrants. Some sources found that sagebrush treatments decrease the abundance of birds, including those categorized as sagebrush-obligates and sagebrush-associated. However, Norvell et al. (2014) found that results varied by bird species, and while none were likely to be extirpated as a result of the sagebrush reduction treatments, an important finding was the importance of lag time in the birds' response to habitat change. More recently Carlisle et al. (2018b) assessed how sagebrush mowing treatments (removal of sagebrush to a height of approximately 25 cm) impacted Brewer's sparrows,

sage thrashers (*Oreoscoptes montanus*), and vesper sparrows. Overall, mowing had negative or neutral impacts on the Brewer's sparrow and sage thrasher, while there were some beneficial impacts to vesper sparrow. After sagebrush was mowed, Brewer's sparrows and sage thrashers did not nest in the sites but approximately half of vesper sparrow nests were maintained. Where nests occurred, survival of nestlings did not differ in any species between treatments, but vesper sparrows nestlings in mowed areas had greater body condition than nestlings in non-mowed sites. Results may be influenced by scale. Sagebrush treatments that focus on fuels removal may have different effects than larger treatments focusing on habitat restoration.

Studies of invertebrate response to mechanical sagebrush treatments are limited but the ones in our review chiefly showed no response to mechanical treatment. McIver and Macke (2014) studied butterfly response to sagebrush treatments in the Great Basin. Most species showed little response to sagebrush treatments, perhaps due to high levels of spatial and temporal variability. Long-term monitoring would be necessary to draw conclusions about the impact of mechanical treatments on these species. Where there was a response, however, it was positive, likely because increased herbaceous plants provided food resources. The authors conclude that habitat changes such as those induced by treatments will favor some species over others, and it is necessary to provide a balance across the landscape in management activities and not treat too much at one time (McIver and Macke 2014). Studies by Hess (2011) in Wyoming indicate that mowing does not lead to an increase in weight or abundance of beetles and grasshoppers. Yeo (2009) studied harvester ant mounds before and after treatment in Idaho. Ant colonies initially showed an increase in the treated area relative to the control a few years after treatment. At the end of the study period (seven years after treatment), however, there were more colonies in the control than in the treatment. Environmental variables such as climate and precipitation may have heavily influenced the results.

Sagebrush is important forage for mule deer, elk, and pronghorn (Beck et al. 2012). Sagebrush treatments may be conducted based on the belief that as sagebrush plants age, their nutritional value for wildlife declines. However, several researchers have found sagebrush nutritional value is not correlated with age. Peterson (1995) and Wamboldt (2004) reported that there is no relationship between crude protein content and age of

big sagebrush. In addition, young big sagebrush have stronger chemical resistance to herbivores (Karban et al. 2006; Shiojiri and Karban 2006) and may be less palatable or provide less nutritious forage than older stands (Beck et al. 2012). Terpene levels (high quantities of terpene can degrade the forage value of sagebrush) in basin, mountain, and Wyoming big sagebrush are not affected by plant age (Kelsey et al. 1983). Davies et al. (2009a) found that experimental mowing of sagebrush (*A. tridentata* ssp. *wyomingensis*) increased its nutritional value, but this may not be biologically significant. The authors postulated that because sagebrush treatments reduce overall sagebrush density, cover, and volume, they may negatively impact ungulates, despite potential increases in nutritional value.

Other studies of mammal response to sagebrush treatment focus on pygmy rabbits, a sagebrush obligate (Green and Flinders 1980). In Utah, Flinders et al. (2005, 2006), Lee (2008), Pierce et al. (2011), and Wilson et al. (2011) studied the impact of sagebrush removal treatments that were included within mosaics of untreated areas. Pygmy rabbits utilized treatment areas but use was greater in non-treated areas (Lee 2008, Wilson et al. 2011). Wilson et al. (2011) and

Pierce et al. (2011) attributed the reduction in use within treatment areas to an avoidance of habitat edges, which are associated with an increase in predators and competitors. To support pygmy rabbits, several authors concluded that mechanical sagebrush treatments should include large areas of untreated areas or mosaics to provide pygmy rabbit habitat (Flinders et al. 2005, 2006; Lee 2008; Wilson et al. 2011).

While not specific to sagebrush communities, Fulbright et al. (2018) conducted a review on wildlife responses to brush (e.g., snakeweed, creosote bush and mesquite) treatments. Effects of shrub treatments on wildlife forage varied: 48% were positive, 31% were neutral (with no or short-term increases), and 20% were negative. In most cases, negative responses occurred where brush treatments reduced key shrub-associated foods, reduced browse plants, or increased thorns or secondary compounds. The authors report the potential benefits of brush management for wildlife are variable. Fulbright et al. (2018) recommend that managing brush for one species and benefitting all other species is not feasible. They recommend that shrub management considers the complexity of wildlife/biodiversity responses to brush management, including variation in species, functional



A literature review by Beck et al. (2012) concludes that the use of sagebrush treatments to benefit wildlife is not supported by the literature. Given the reliance of so many species on sagebrush, treating too many acres at once could lead to declines of some wildlife populations. (Photo: Tom Koerner/U.S. Fish & Wildlife Service)

group, seasonal use, potential changes in predator-prey relationships, invertebrate responses, and critical life-cycle phases of wildlife.

Beck et al.'s (2012) literature review on sagebrush treatment effects on wildlife concluded that the use of sagebrush treatments to benefit wildlife is not supported by the literature. They report that, given the reliance of so many species on sagebrush, treating too many acres at once could lead to declines of these species. They recommend land managers not implement sagebrush treatments until further study is available. Welch and Criddle (2003) concluded that as more acres of sagebrush communities are modified by development or converted into invasive, annual weeds, sagebrush reduction treatments are inadvisable because they may impact sagebrush obligate species' survival.

3.3.3 Greater Sage-Grouse

The current conservation status of greater sage-grouse (sage-grouse) has led many western states and habitat managers to call for increased conservation of the remaining sagebrush stands and rehabilitating or improving degraded sagebrush systems through various forms of treatment, which can include mechanical means. In 2015, the U.S. Fish and Wildlife Service determined that sage-grouse was not warranted for listing under the Endangered Species Act but identified habitat loss and fragmentation as key reasons for sage-grouse declines. The U.S. Fish and Wildlife Service (2010) also indicated that vegetation treatments may not be beneficial to sage-grouse and that the rationale for conducting them deserved further study. Habitat treatments for sage-grouse include treating sagebrush and removing pinyon-juniper woodlands.

Below we report on mechanical treatment effects on sage-grouse (Figure 6). This includes studies that investigated the effects of sagebrush treatments in occupied sage-grouse habitat and removal of pinyon-juniper trees in areas with sagebrush understories adjacent to sage-grouse habitat. The variables examined in these studies ranged from sage-grouse use/occupancy to lek attendance to nesting frequency to success of nesting or brood rearing.

Sagebrush Communities

In sagebrush communities, mechanical mowing or chaining treatments are sometimes used to alter sage-grouse habitat. Sagebrush treatments are designed to reduce cover of sagebrush, often with the goal of

allowing perennial grasses and forbs to increase and thus benefit sage-grouse. However, our summary chart shows a positive response by sage-grouse in only 36% of the data points. Negative (27%) and not significant (36%) responses were the majority. While our summary chart reports roughly equal numbers of data points that found positive versus negative versus non-significant effects on sage-grouse, many researchers have concluded that removal of sagebrush through a variety of means can have negative impacts on sage-grouse (Beck et al. 2012; Braun et al. 1977; Connelly et al. 2000; Fischer et al. 1996; Peterson 1995; Pyrah 1972; Swenson et al. 1987; Wallestad 1975).

Several studies found that mowing treatments do not lead to an increase in critical sage-grouse early brood-rearing requirements such as forb abundance, forb nutritional content, perennial grass cover or height (Hess, 2011; Hess and Beck, 2010, 2012, 2014), or weight or abundance of beetles and grasshoppers (Christiansen 1988; Scoggan and Brusven 1973).

The removal of big sagebrush by any means in sage-grouse winter or breeding habitats usually will have a negative or neutral effect on sage-grouse (Beck et al. 2012; Gates 1983; Martin 1990; Robertson 1991). Dahlgren et al. (2006) found no significant difference in sage-grouse use between mechanically treated plots and control plots, although sage-grouse brood use was higher in chemically treated than control plots, which the authors attributed to increased forb cover. However, in all treated plots, sage-grouse use was greatest within 10 m of the edge of the treatments where adjacent sagebrush cover was still available. In Colorado, Braun and Beck (1996) found that after 28% of the sagebrush in their study area was treated (from 1965 to 1970), the mean 5-year average of attending males on leks dropped 25%. And in a study of both mechanical and chemical treatments within a 0.5 km radius around four leks in 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 (Wallestad 1975). A study by Holloran and Belinda (2009) found that sage-grouse populations in Wyoming began declining with as little as 3.4% of the sagebrush cover removed. Autenrieth (1969) summarized the impacts of big sagebrush control on sage-grouse and recommended that sagebrush reduction treatments should never be conducted within 2 miles of a lek, in known sage-grouse wintering concentration areas, nor along streams, meadows, or secondary drainages, both dry and intermittent.

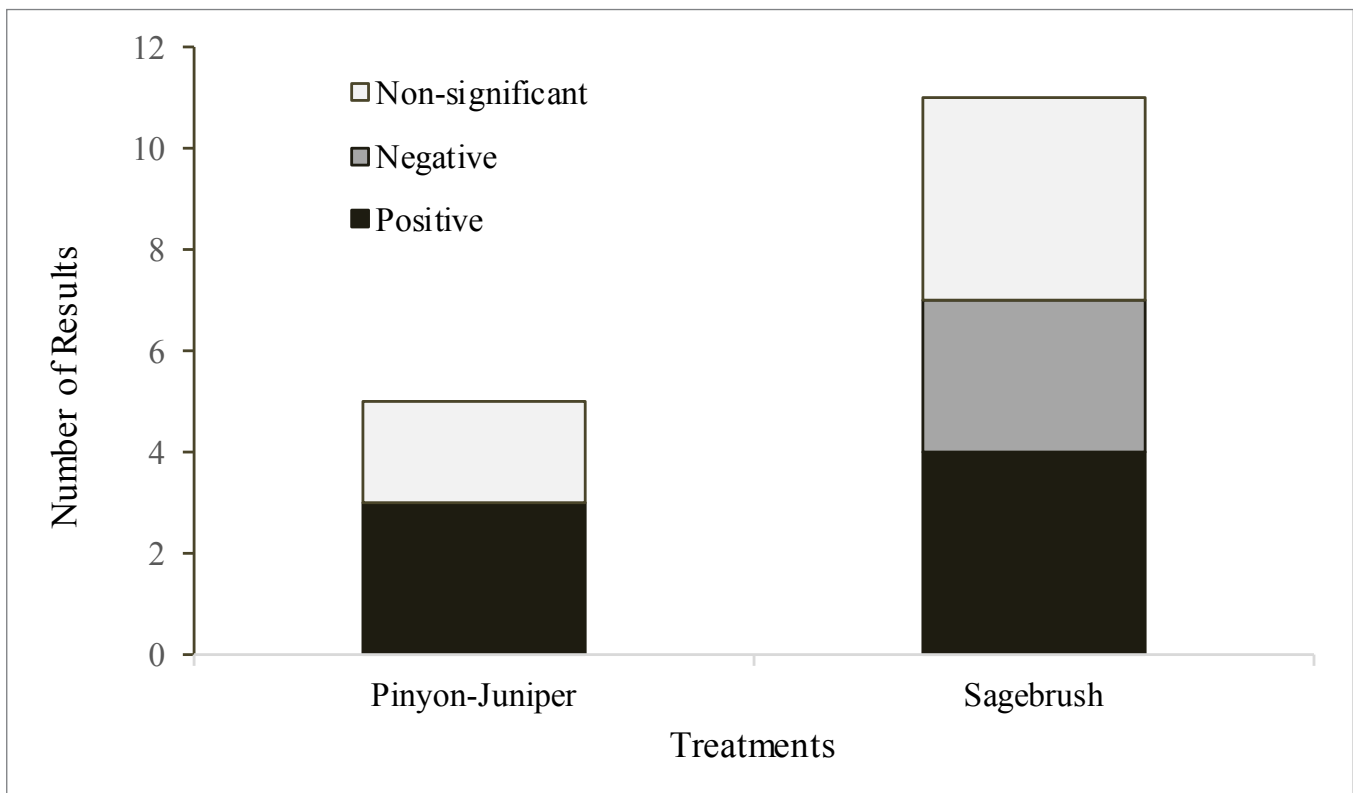


Figure 6. Number of pinyon-juniper and sagebrush treatment study results that elicited positive, negative, or non-significant response from sage-grouse. The appendix lists all of the studies that contributed data points to this summary chart, and the response variables measured.

In Utah, Graham (2013) examined the removal of sagebrush in occupied sage-grouse habitat for the purpose of “green-stripping” and establishing fire breaks by seeding forage kochia (*Bassia prostrata*). She found that, similar to other studies, sage-grouse selected for untreated areas. While treated areas were used to expand the size of the active lek to a larger area than was previously used, it was not reported whether this equated to an increase in males using the lek.

Several studies report that sagebrush treatments have positive effects on sage-grouse. Dahlgren et al. (2015) found that, at first, sage-grouse lek counts increased in sagebrush treatment sites relative to surrounding populations in untreated sites. At the time they were studying them, sage-grouse broods used plots of <200 ha treated sagebrush mosaics more than untreated sites. The higher lek counts in the treatment sites were sustained for nearly 15 years. However, with continued sagebrush treatments and adverse winter conditions, lek counts then declined to levels similar to surrounding areas. Dahlgren et al. (2015) hypothesized that sagebrush treatments increased availability of grasses and forbs to sage-grouse, but that cumulative annual reductions in sagebrush may have reduced availability of sagebrush cover for sage-grouse seasonal needs.

Stringham (2010) similarly reported sage-grouse use of sagebrush treatment sites during the breeding and early brood-rearing periods, but not winter. Baxter et al. (2017) developed resource selection function models using a 19-year telemetry data set (1998–2016) from northeastern Utah to evaluate response of sage-grouse to treatments. Statistical models were built using 418 locations to assess the influence of mountain big sagebrush treatments on sage-grouse habitat selection during the brooding period. They found that post-treatment sage-grouse selected areas that were inside treated areas or near treatment edges. In Utah, Ritchie et al. (1994) examined sage-grouse nests in areas that had been chained and seeded 25 years previously and compared those to areas that were untreated. They found that predation rates of artificial nests were higher in areas of untreated sagebrush, even though these areas had greater sagebrush cover, taller shrubs, and greater horizontal plant cover. They hypothesized that untreated sites may contain greater abundance of potential prey species, such as lagomorphs, and thus, may attract greater densities of sage-grouse predators.

Some studies have specifically investigated the effects of sagebrush mowing treatments on sage-grouse habitat. Hess and Beck (2010) evaluated treatments in



While studies show that pinyon-juniper removal adjacent to occupied sage-grouse habitat can benefit sage-grouse, several researchers have shown that removal of big sagebrush by any means in sage-grouse winter or breeding habitats will tend to have a negative or neutral effect on sage-grouse. (Photo: Tom Becker/Utah Division of Wildlife Resources)

the Bighorn Basin in Wyoming for sage-grouse habitat 10 to 18 years after treatment. They found no positive structural changes in vegetation in the treatment areas and concluded that treatments do not result in improved, long-term habitat conditions for sage-grouse nesting and early brood-rearing habitat.

Many researchers have recommended against sagebrush treatment in important sage-grouse habitats (Beck et al. 2012; Connelly et al. 2000; Fischer et al. 1996; Hess and Beck 2014; Woodward 2006). For example, Connelly et al. (2000) recommended treating no more than 20% of breeding habitat in Wyoming big sagebrush every 30 years. Beck et al. (2012) stated that “sagebrush is essential to maintaining native plants and limiting the invasion of non-native plants in sagebrush communities [and] future treatments should be limited to those that do not eliminate or greatly reduce sagebrush.” Fischer et al. (1996) noted that their findings did not support the idea that killing big sagebrush enhanced sage-grouse brood-rearing habitat. Woodward (2006) cautioned against removing sagebrush stands even if the herbaceous community is depleted and not ideal for sage-grouse. Hess and Beck (2010) stated, “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”

Pinyon-Juniper Woodlands

Sage-grouse avoid habitat containing pinyon-juniper trees, primarily because it offers perching habitat for avian predators (Connelly et al. 2000; Knick and Connelly 2011; Manzer and Hannon 2005; Severson et al. 2017). Habitat improvements for sage-grouse can be accomplished by removing trees of Phase I and perhaps also Phase II pinyon-juniper woodland expansion into sagebrush communities. We found no examples of studies that removing pinyon-juniper had negative effects on sage-grouse; effects were either positive (60%) or non-significant (40%). For example, Severson et al. (2017) found that sage-grouse use increased in sagebrush communities where pinyon-juniper trees were removed: the probability of sage-grouse nesting increased by 22% annually, female sage-grouse were 43% more likely to nest near treatments (within 1000 m), and 29% of the study birds shifted nesting activities into treatments. Sandford et al. (2017) demonstrated that where pinyon-juniper trees were removed, sage-grouse se-

lected for nest and brooding sites and the probability of nest and brood success declined as sage-grouse females selected sites farther from conifer removal areas where tree density was greater. Frey et al. (2013) found that in southern Utah, sage-grouse used pinyon-juniper treatments more than expected based on availability in the first two years of the study, and for the following two years, use evened out to what would be expected based on availability. However, the pinyon-juniper treatments had lower grass and forb composition and height when compared to sage-grouse habitat suitability standards. Frey et al. (2013) posited that the positive response in sage-grouse use in the pinyon-juniper treatments immediately after treatment, despite decreased herbaceous composition and height, suggest that suitable habitat is limited in the region. Frey (2018) continued to track 10 males and 3 hens in the southern Utah study from 2013 to 2016 and reported that, while the females appear to continue to use treatments more than would be expected, males strongly prefer to use untreated sagebrush in the study area. However, the sample size was too small to derive any conclusions from the data, and habitat use relative to habitat availability was not analyzed (Joshua O'Brien, personal communication, December 2018).

Recent studies agree with the very few studies on conifer removal effects on sage-grouse conducted before about 2010. For example Commons et al. (1999) found that pinyon-juniper removal in Gunnison sage-grouse (*Centrocercus minimus*) habitat in Colorado, in association with brush-beating to reduce height of mountain big sagebrush and deciduous brush, resulted in doubling numbers of male sage-grouse counted on treatment leks in years two and three post-treatment.

Two factors contribute to the efficacy of improving sage-grouse habitat with pinyon-juniper removal treatments. Removal of pinyon-juniper trees should occur in areas with existing sagebrush understory and in areas adjacent to occupied sage-grouse habitat (Cook 2015, Cook et al. 2017). In Utah, Cook (2015) and Cook et al. (2017) found that sage-grouse use of pinyon-juniper removal treatments was positively associated with sage-grouse occupancy in adjacent habitats.

In terms of focus specifically on annual and perennial forbs preferred by sage-grouse, Bates et al. (2017b) analyzed data sets from 16 previous and ongoing studies across the Great Basin characterizing cover response of perennial and annual forbs that are consumed by sage-grouse to mechanical, prescribed fire, and low-disturbance fuel reduction treatments in sage-

brush sites experiencing pinyon-juniper expansion. The studies they reviewed reported a mix of no change or measured increased or decreased perennial forb cover following pinyon-juniper cutting and fuel reduction. They reported that site potential appears to be a major determinant for gains in forb cover following conifer control. Additionally, the response of perennial forbs was similar regardless of conifer treatment when comparing prescribed fire, clear-cutting, and fuel reduction. Annual forbs favored by sage-grouse benefited most from prescribed fire treatments with smaller increases following mechanical and fuel reduction treatments (Bates et al. 2017b).

3.4 ECOSYSTEM FUNCTION

3.4.1 Soil Stability

Maintaining soil stability and preventing excessive erosion are critical functions for ecosystem health and a common objective for conducting vegetation treatments. Mechanical treatments disturb soils and remove vegetation, at least in the short term. Techniques that uproot plants can lead to the greatest degree of soil disturbance, thus adding to the risk of erosion (Pyke 2011). However, if treatments ultimately increase vegetation, soil stability can improve. The summary chart for this variable combines data points from pinyon-juniper (including treatments in Phase I, II, and III expanding woodlands) and sagebrush treatments (Figure 7). Soil stability in treatment sites is influenced by variables such as climate, geomorphology of the site, type of soil, livestock grazing regime, and establishment of invasive plants (Davenport et al. 1998). These variables vary widely across the range of pinyon-juniper woodlands in the western United States.

Most studies in the summary chart report a non-significant effect of treatments on erosion (63%) and runoff (80%). In this summary chart, a positive response was defined as a decrease erosion/runoff, and a negative response was defined as an increase erosion/runoff, and this should be kept in mind when reviewing Figure 7. In our review of the literature, we found that when there was a significant response to mechanical treatment, the majority was negative (30% for erosion, 16% for runoff). A smaller number of data points showed positive responses (7% for erosion, 4% for runoff).

Some researchers conclude that pinyon-juniper treatments reduce erosion over time by increasing vegetative cover on the soil surface (Farmer et al. 1999; Jacobs 2015; Richardson et al. 1979; Roundy et al.

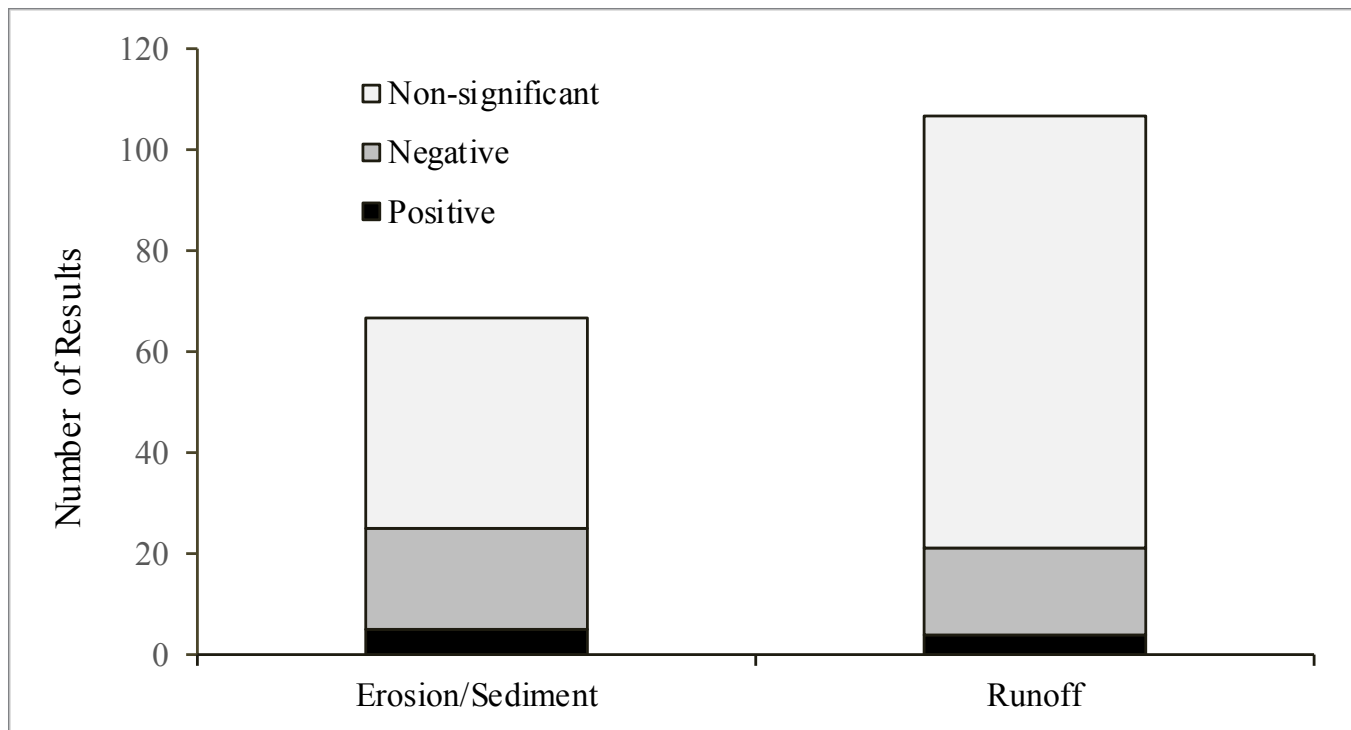


Figure 7. Number of pinyon-juniper treatment study results that had positive, negative, or non-significant response on erosion and runoff variables. The appendix lists all of the studies that contributed data points to this summary chart, and the response variables measured.

2016) and subsequent decreases in bare ground after treatments (Cline et al. 2010, Pierson et al. 2007a and 2007b, Pierson et al. 2013). Some posited that mastication of trees is effective at preventing soil loss on severely degraded areas with high rates of erosion (Pierson et al. 2014), ostensibly because of slash left behind from pinyon-juniper treatments, which can be particularly helpful on steep slopes in order to reduced post-treatment surface runoff and erosion (Noelle et al. 2017).

Other studies have found that pinyon-juniper treatments did not affect erosion or surface runoff. Ross et al. (2012) reported no significant difference in soil aggregate stability between pinyon-juniper lop and scatter treatments and control sites. In the study's mastication treatments, one site showed no significant difference between treatment and control and another site showed lower soil aggregate stability than the control. Brockway et al. (2002) found no significant difference in soil erosion rates between slash treatments and controls. In southwestern Colorado, mastication treatments in pinyon-juniper woodlands showed no difference in aggregate stability compared to untreated sites (Owens et al. 2009). Watershed-scale experiments in Arizona indicate no effect of mechanical pinyon-juniper removal on surface runoff (Clary et al. 1974). In Texas, Dugas et al. (1998) found that when Ashe juniper (*Juniperus*

ashei) cover was removed by hand cutting, surface runoff between treatments varied and the results were inconclusive. Gifford (1973) found that if debris is left in place, there was no significant difference in surface runoff between treated and untreated locations.

Surface runoff and erosion may increase as a result of pinyon-juniper treatments (Gifford 1973; Myrick 1971). Myrick (1971) found that chaining pinyon-juniper, burning the slash, and then seeding the site caused an increase in surface runoff in the 2 years following treatment. On chained with windrowed pinyon-juniper treatments, Gifford (1973) found surface runoff was greater compared to the untreated woodland sites, but about the same if debris was left in place.

The surface disturbance caused by mechanical treatment can impact biological soil crusts. These crusts prevent soil and wind erosion by protecting the soil surface and contributing to soil aggregate stability (Belnap and Eldridge 2001; Belnap and Gillette 1998; Gifford et al. 1970; Loope and Gifford 1972; Wilcox 1994). In semi-arid regions, it is the single most important stabilizer of the soil surface, and therefore, it primarily influences soil erosion (Bowker et al. 2008). Mechanical treatments remove crusts and the time for recovery varies: early succession components like cyanobacteria and chlorophyta return within one to two years



An erosion gully in a vegetation treatment and seeded area in Fisher Valley, Utah. (Photo: Laura Welp)



Patches of biological soil crusts remain after a pinyon-juniper mastication project in Kane County, Utah. (Photo: Neal Clark)

but mosses and lichens may take decades to develop (Belnap 1993; Belnap and Gillette 1997). Soils themselves in arid regions such as the Colorado Plateau can take 5,000 to 10,000 years to regenerate (Webb 1983), so post-treatment soil loss from erosion can be considered irreversible. A study of pinyon-juniper treatments in southern Utah found that bare soil was higher and biological soil crust cover lower than untreated comparisons even decades after disturbance (Redmond et al. 2013).

3.4.2 Watershed Productivity

Vegetation influences the ecohydrology of a site by affecting runoff and evapotranspiration. Vegetation treatments that remove woody plants are often assumed to increase water supply by reducing evapotranspiration and thus making more water available for streamflow and groundwater recharge. This has not been definitively demonstrated by most research, however (Seyfried and Wilcox 2006), as demonstrated by the mixed results summarized below (Figure 8).

Some research supports the contention that pinyon-juniper treatments increase soil moisture. In Utah, removing trees increased the time that shallow soil water was available to understory plants (Roundy et al. 2014b). Methods that leave debris in place, such as chaining and mastication, increased soil moisture and water in-

filtration compared to untreated woodlands (Bates et al. 2000; Gifford 1982; Gifford and Shaw 1973). Mollnau et al. (2014) found that transpiration and interception rates decreased when trees were removed, increasing soil water recharge. A study of karst systems in Texas (Huang et al. 20016) also concluded that where springs are present, removing woody plants had the potential to increase streamflow and groundwater recharge. Questions remain about the long-term persistence and relevant scale of this effect. In the Camp Creek watershed in Oregon, treatment sites produced significantly greater late season spring-flow rates and more days of groundwater availability than untreated sites (Deboodt et al. 2009; Ochoa et al. 2018). Pinyon-juniper treatments reduced the interception of snow, which allowed water to percolate through the soil and into the shallow aquifer. They mention that topography and precipitation level are important influences on the results and that more study is needed before expanding plot and watershed scale studies to regional landscapes (Ochoa et al. 2018). The strongest responses were in spring flow and soil moisture, whereas groundwater levels and intermittent streamflow declined to less than pre-treatment levels during late summer into fall. Also, the average annual precipitation in the study area was 13 inches/year, less than the generally assumed minimum for yielding measurable changes (Hugh Hurlow, personal communication, November 2018).

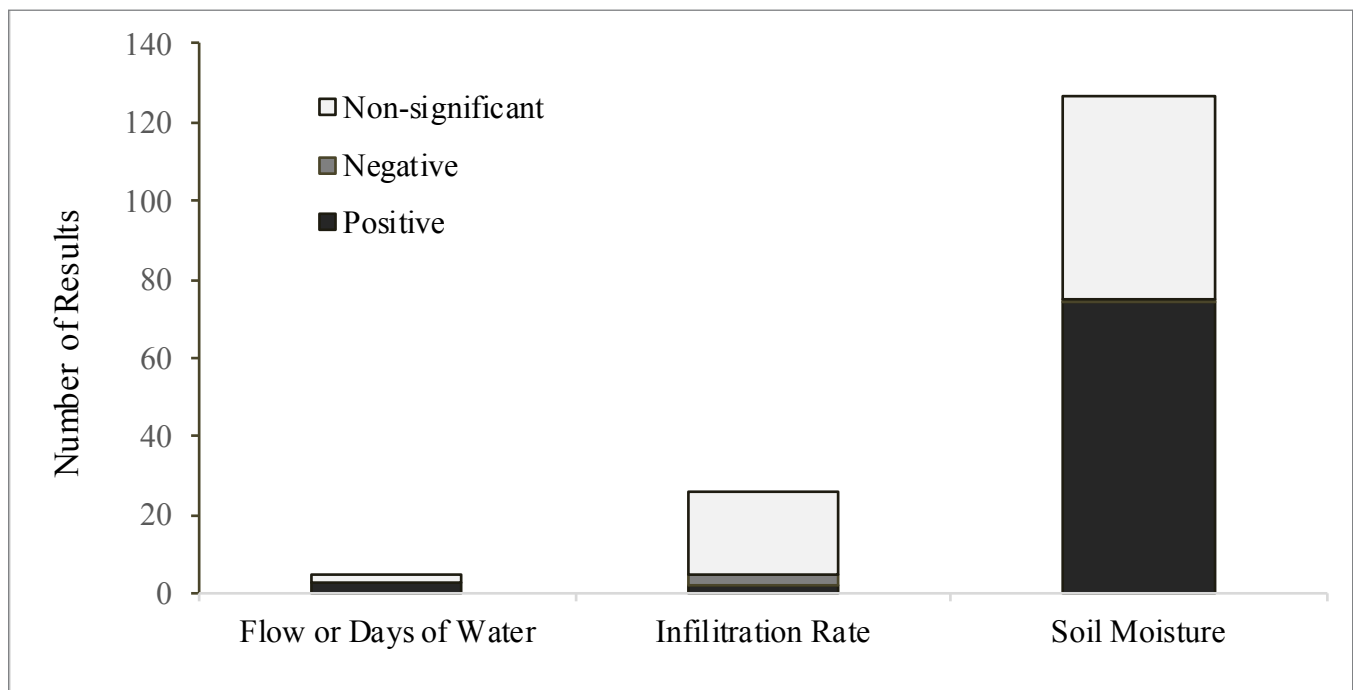


Figure 8. Number of pinyon-juniper treatment study results within flow/days of water, infiltration rate, and soil moisture categories that found positive, negative, or non-significant results. The appendix lists all of the studies that contributed data points to this summary chart, and the response variables measured.

Other studies did not find any significant changes in infiltration rates or water yield with treatments (Blackburn and Skau 1974; Brown, 1987; Cardella Dammeyer 2016; Clary et al. 1974; Collings and Myrick, 1966; Gifford et al. 1970; Ramirez et al. 2008; Renard 1987; Schmidt 1987; Wilcox and Huang, 2010). Research in Utah has shown that chaining decreased infiltration rates because it removed biological crusts, which absorb and retain a large amount of precipitation in desert watersheds (Loope and Gifford 1972). In addition, removal of junipers, when replaced with herbaceous vegetation and low shrubs, had little effect on deep recharge in this study. The increased transpiration from understory vegetation can compensate for decreased transpiration by trees, and therefore, tree removal may have little or no effect on runoff (Bazan et al. 2012; Cardella Dammeyer et al. 2016; Wilcox et al. 2003). This effect was also suggested by studies on beetle-killed pinyon and juniper stands. Guardiola-Claramonte et al. (2011) found a decrease in streamflow after a beetle infestation killed pinyon-juniper trees. They concluded that this was due to increased understory herbaceous cover and increased solar radiation reaching the ground, which together may have reduced overland flow by increasing understory transpiration and soil evaporation and thus compensated for tree removal. In the intermountain west, results are complicated by the fact that recharge is episodic. It may occur in rare years where enough precipitation is available and does not run off the landscape too quickly. Also, the time between recharge and observing measurable effects is variable, and there may not be results until several years after treatment (Hugh Hurlow, personal communication, November 2018).

Several literature reviews synthesizing this research concluded that tree removal treatments do not reliably result in increased water yield. Water availability may increase in localized areas but extending this effect to a larger scale is not warranted by the research (Seyfried and Wilcox 2006). Belsky (1996) wrote that studies showing that junipers intercept precipitation and transpire water cannot be used to conclude that this “lost water” would have increased flows in streams and springs. Carroll et al. (2016) agree, suggesting that accounts of greater streamflows in the early part of the century are a result of generally cooler and wetter climatic conditions rather than fewer pinyon-juniper trees. Archer et al. (2011) and Archer and Predick (2014) determined that “brush management does not necessarily produce the hydrological benefits that are commonly attributed to it. In most cases, these perceived benefits

are exaggerated and have not been documented, and there is little or no evidence that brush management is a viable strategy for increasing ground water recharge or stream flows at meaningful scales.” Zeimer (1987) agrees, asking, “Can water yields be increased through management of vegetation? Nearly all studies clearly show that the answer is yes. Will operational programs to increase water yields be successful? History has clearly shown that the answer is no, and there is little reason to believe that future attempts at an operational scale to increase water yields will be successful.”

Our review of the literature found that mechanical treatment effects on ecohydrology are highly site-dependent and unpredictable. For example, two sites with different vegetation types can receive the same amount of precipitation, but the rates of runoff and evapotranspiration can vary widely depending on the type of vegetation present. A study by Kormos et al. (2017) found that pinyon-juniper woodlands on a site in Idaho had higher snow density and depth, earlier snow melts, and greater evapotranspiration than adjacent sagebrush communities. Sagebrush collected more snow in drifts, which melted later in the season, delaying water delivery to the site. This difference in timing and amount of water availability affects vegetation dynamics and will similarly affect studies investigating ecohydrological responses of sites like this to mechanical treatment.

These mixed results can be explained by the fact that the potential for increasing water supply in semiarid regions is small per unit area (Seyfried and Wilcox 2006). Long-term watershed experiments have consistently recommended that forest cover must be reduced by at least 20% throughout the entire watershed to observe a measurable change in streamflow or water yield (Bosch and Hewlett 1982; Huff et al. 2000; Troendle et al. 2010). Changes in water yield are unlikely to be detected for reductions of less than 20% of wooded area or for relatively dry watersheds (Troendle et al. 2010). Niemeyer et al. (2017) developed a model to address streamflow associated with pinyon-juniper cover. They found that changes in streamflow are heavily dependent on the timing and amount of precipitation relative to evapotranspiration. In the southern range of pinyon-juniper woodlands, most annual precipitation falls in the summer when temperatures and evaporative demand are high. Therefore, reducing pinyon-juniper cover will have little effect on streamflow. Conversely, in the north and western portion of the pinyon-juniper woodland range, precipitation falls mainly in winter when evaporative demand is low and removing pinyon-juni-



Left: Onaqui, Utah pinyon-juniper woodland in 2006, prior to a bullhog mastication treatment. Right: A photo of the same area nine years after treatment, showing a positive herbaceous response. Studies examining hydrological characteristics such as soil moisture or infiltration rates after pinyon-juniper removal are mixed. Part of the reason for this could be increased transpiration from increased understory vegetation after treatment, which can compensate for decreased transpiration by trees. (Photos: SageSTEP)

per trees has a greater chance of altering streamflow. Under these conditions trees have substantially higher evapotranspiration rates relative to shrubs and herbaceous plants. The results of their model suggest that in cooler portions of the range-wide distribution of pinyon-juniper woodlands, there is potential for meaningful increases in streamflow with land cover change from trees to shrubs and grasses. However, as temperatures rise with climate change, this effect on streamflow may diminish even in these areas (Niemeyer et al. 2016, Niemeyer et al. 2017).

The results summarized here collectively suggest that woody removal (most of the studies involving pinyon-juniper removal) treatments are unlikely to result in increased streamflow in all circumstances. Such circumstances include replacement of trees with dense herbaceous cover, high solar radiation at the treatment sites (e.g., south-facing slopes), relatively small treatment areas (less than 20% of the watershed), and precipitation regimes characterized by precipitation during winter rather than growing-season months (Sara Goeking, personal communication, December 2018).

3.4.3 Carbon Sequestration and Climate Change

It has been suggested that removing trees to reduce woody fuels will help keep carbon sequestered in terrestrial pools rather than burned in wildfire and released into the air (Campbell et al. 2012a, 2012b; Rau and Bradley, in preparation [a]). Studies on this topic are often focused on forests other than pinyon-juniper woodlands (Campbell et al. 2012b; Hudiburg et al. 2009) and the research is still in the initial stages. However, all of the research in this review concluded that expansion of shrubs and trees sequesters carbon and treatments could result in a loss of carbon to a greater or lesser degree. Campbell et al (2012b) suggest that treatments removing trees do not mitigate carbon loss from forest fires and that expanding juniper woodlands are sequestering carbon. Hughes et al. (2006) also found that shrubs sequester significant amounts of ecosystem carbon, depending on age, soil type, and plant species. Throop and Lajtha (2018) studied the effect of juniper expansion and removal on carbon pools in a semi-arid sagebrush ecosystem in the Great Basin by coupling tree measurements to estimate landscape-level carbon pools. As juniper size increased so did carbon

storage (excluding deep soil carbon), suggesting that expansion of woody plants and subsequent brush management can have substantive impacts on ecosystem carbon pools. Rau and Bradley (in preparation [b]) also found that pinyon-juniper woodlands store a disproportionate amount of carbon relative to other Great Basin land cover types. In their study of shrublands in Australia, Daryanto et al. (2013) found that the treatment they examined with the most soil and vegetation disturbance (plowing followed by livestock grazing) resulted in the greatest loss of soil organic carbon. The treatment with the least amount of disturbance (protection from grazing and shrub removal) had the largest amount of soil organic carbon.

Archer and Predick (2014) report that the enhanced productivity accompanying woody plant encroachment in some bioclimatic zones can translate into increases in the above-ground carbon pool that can range from 300 to 44,000 kg C ha⁻¹ in < 60 years of woody encroachment. They stress, however, that these gains will be substantially and rapidly offset by reductions in aboveground standing woody biomass that follow brush management. Neff et al. (1990) found that pinyon-juniper expansion led to increased carbon sequestration into the upper layers of soil, but it was relatively short-term storage. Thinning and overstory removal will cause relatively rapid declines in surface soil carbon and nitrogen storage in some pinyon-juniper communities. In a study of carbon sequestration in eastern Oregon, Campbell et al. (2012b) found that juniper expansion did sequester carbon, although this was offset by juniper removal by fire or management prescriptions. However, Rau and Bradley (in preparation [b]) point out that the release of carbon through mechanical removal of woody biomass is likely to be less than that of prescribed burns and wildfire.

Archer et al. (2011) present a different perspective on expansion of woody species and carbon sequestration. They conclude, “The recognition that [woody plant] proliferation can substantially promote ecosystem primary production and carbon stocks may trigger new land use drivers as industries seek opportunities to acquire and accumulate carbon credits to offset CO₂ emissions. [Woody plant] proliferation in grasslands and savannas may therefore shift from being an economic liability in the context of livestock production to a source of income in a carbon sequestration context.” Daryanto et al. (2013) also suggested that “carbon farming” could provide an economically viable alternative to traditional land use practices in Australia.

3.5 LIVESTOCK GRAZING

3.5.1 Livestock and Vegetation Functional Groups in Pinyon-Juniper Communities

Inappropriate livestock grazing in many pinyon-juniper woodlands on the Colorado Plateau and intermountain West has diminished or altered herbaceous vegetation, leading to widespread degradation of understory conditions (Burkhardt 1996; Lanner 1981; Milchunas 2006; Nevada Division of Water Planning 2000). Some of the researchers who have studied pinyon-juniper woodland expansion into adjacent shrublands have concluded that the expansion of trees into formerly unoccupied sites is most likely due to livestock grazing which depletes native herbaceous vegetation and causes subsequent reductions in fire frequency (Burkhardt and Tisdale 1976; Eddleman 1987; Ellison 1960; Evans 1988; Miller and Wigand 1994; Neilson 1986; Young and Evans 1981). 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 . . . [T]he 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.” Other researchers, however, suggest that the distribution of pinyon and juniper trees was influenced less by grazing levels than by changing fire regimes, past climate, and the effect of precipitation on recruitment (Barger et al. 2009). The authors warn that predictions of warming climate and lower precipitation may indicate the potential for lower recruitment rates and pinyon regeneration. They recommend that managers take this into account in planning treatments.

In areas where herbaceous vegetation is in poor condition, particularly within sagebrush communities invaded or recolonized by pinyon-juniper trees, the signs of ecosystem degradation that are attributed to encroachment are often difficult to tease apart from the symptoms caused by livestock grazing. For example, sometimes decreased water infiltration and increased erosion is attributed to pinyon-juniper expansion but livestock can have even greater effects on water infiltration and erosion by reducing vegetative cover and disturbing and compacting soils by trampling (Fleischner 1994;



An old sagebrush treatment area used for cattle grazing, with untreated sagebrush community in the background. (Photo: Laura Welp)

Jones 2000 and references therein; McPherson and Wright 1990). In her review of the literature on the effects of pinyon-juniper treatments on western ecosystems, Belsky (1996) also reflected on the confounding effect livestock grazing may have on pinyon-juniper woodlands, noting that most of the earlier studies of juniper and pinyon-juniper removal were carried out on sites that were grazed by domestic livestock. Thus, the effects of livestock grazing and tree removal were 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. Furthermore, she noted that it was unknown whether herbaceous production would have differed if livestock grazing had been deferred, reduced, or eliminated after pinyon-juniper removal.

3.5.2 Livestock and Vegetation Functional Groups in Sagebrush Communities

Sagebrush removal is often proposed in sagebrush communities with reduced herbaceous functional groups under the assumption that shrubs are outcompeting desirable herbaceous species and removal of shrubs will restore them. However, in many cases utilization of herbaceous species by grazing is a complicating factor. 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 routinely accompany sagebrush treatments (Beck et al. 2012). Some authors posit that livestock grazing, rather than sagebrush cover, is the principal management practice and influencing factor that affects grass cover and height (Crawford et al. 1992; Rickard et al. 1975).

For example, studies by Davies et al. (2010, 2014b) compared fuel levels on moderately grazed plots with those on plots ungrazed by livestock for 70 years and found more litter and greater fuel continuity in ungrazed plots. Although sagebrush height and canopy diameter were higher in the ungrazed plots, total herbaceous cover was also one and a half times greater, and perennial bunchgrass cover was twice as great. The ungrazed plots also had more continuous perennial bunchgrass cover. Grazing, even at moderate levels, had a greater effect on reducing herbaceous cover (including native perennial bunchgrasses) than did the amount of sagebrush.

Other studies also show that grass canopy cover is higher in ungrazed areas, even in areas of high sagebrush canopy cover, and bare ground cover in these areas is low (e.g., Jones 2000; Mueggler and Stewart

1980; Peterson 1995; Welch 2005; Yeo 2005). 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 (Anderson and Holte 1981; Branson and Miller 1981; Pearson 1965). In the absence of grazing, sagebrush communities at their ecological potential have little bare ground and can be dominated by perennial grasses and biological soil crust (Peterson 1995; Welch and Criddle 2003). The seminal publication on sage-grouse in *Studies in Avian Biology* by Knick and Connelly (2011) states that “no evidence supports the belief that sagebrush dominance will continue at the expense of perennial grass cover or survival” (citing Pyke 2011).

Welch (2005) assembled the results of 29 separate studies that determined the amount of perennial grass production achieved by reducing big sagebrush by various means on different types of sites for varying periods of times after treatment. They 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. Canopy cover of big sagebrush was not significantly correlated with cover of graminoids, forbs, or bare soil. This suggests that the amount of perennial grass cover, or lack of it, in dense sagebrush stands is often not the result of competitive exclusion of sagebrush on grasses and forbs. Others have noted that differences in perennial grass production in big sagebrush stands have less to do with shrub cover than with soil type, annual precipitation, grass species, and especially grazing history (Pechanec and Stewart 1949; Peterson 1995; Welch 2005).

While not specific to sagebrush treatments, Reisner et al. (2013) found that limiting size and connectivity of gaps between vegetation is important to sagebrush resistance to invasion of non-native plants. Maintaining biological soil crust (i.e., limiting soil surface disturbance) also appears to reduce non-native plant cover. They suggest that cattle grazing, by reducing bunchgrasses and trampling biological soil crust, reduces resistance to non-native species invasion. Managers seeking to restore sagebrush systems should focus on restoring these two functional groups, which may require changes in grazing management to prioritize vegetative recovery. Chambers et al. (2017) stressed that one of the primary global change factors that threaten shrublands worldwide is loss of native perennial herbaceous species due to inappropriate livestock grazing.

3.5.3 Exclosure Studies

Since livestock grazing is a ubiquitous land use and mechanical treatments are often conducted to provide forage for cattle, post-treatment monitoring to evaluate the effects of grazing on treatments would seem important. Surprisingly though, in this review we found that only seven studies systematically addressed the effect of livestock grazing on mechanical treatments. Three of them were conducted by Gifford and others. In response to public health concerns over fecal contamination of water sources by cattle, Buckhouse and Gifford (1976) conducted a water quality survey one year after grazing resumed on a chained and seeded pinyon-juniper treatment that had been ungrazed for eight years. They found that fecal and total coliform production contamination from cattle showed no significant change. Busby and Gifford (1981) compared erosion and infiltration rates between three pinyon-juniper treatments: untreated control, chained with debris left in place, or chained with windrows. Grazing exclosures ranged from two to five years post-treatment. Infiltration increased on all sites as time since grazing increased. Treated plots protected from grazing



An old seeded area after treatment (and cattle grazing) in Grand Staircase-Escalante National Monument. (Photo: Laura Welp)



Sagebrush steppe ecosystem, Seedskaadee National Wildlife Refuge, Wyoming. (Photo: Tom Koerner/U.S. Fish & Wildlife Service)

the longest (five years) had higher infiltration rates than grazed plots on treated and untreated areas. Younger exclosures (two to four years) showed no significant difference in infiltration rates compared to grazed plots. The authors say that spring-fall grazing significantly reduced infiltration rates, as did grazing that removed 45 to 70% of the year's forage. Gifford (1982) also studied water storage in grazed and ungrazed chained pinyon-juniper treatments where slash was piled into windrows rather than allowed to remain in place. Grazing did not affect soil water storage even though the crested wheatgrass on the chaining was heavily utilized each spring (55 to 78%). Gifford attributes this lack of impact to the low cover of the crested wheatgrass (maximum 25% canopy coverage) on the treatment even without grazing. With such low vegetative cover to begin with, grazing did not make enough of a difference in evaporative conditions to modify soil water conditions.

Yeo (2005) compared long-term exclosures in sagebrush steppe and shadscale communities with adjacent grazed sites in Idaho. Meta-analysis of the data showed that grazing exclusion resulted in more cover of biological soil crust, bluebunch wheatgrass (a preferred forage species), and greater screening cover (a measure

of wildlife habitat). The cover of Sandberg bluegrass (*Poa secunda*) (an unpalatable species), bare ground, and other indicators of soil erosion was greater outside the exclosure than inside. Yeo concluded that livestock grazing can "limit the potential of native plant communities in sagebrush steppe ecosystems, and . . . the health of semiarid ecosystems can improve with livestock exclusion in the absence of other disturbances . . ." Yeo (2009a) then measured livestock effects on mechanical treatments applied to this area in 2003. He found that although treatments increased grasses in both grazed and ungrazed plots, exclosures had higher cover of preferred grasses. Bare ground was higher outside the exclosures. Forbs did not respond even in exclosures. A companion study (Yeo 2009b) showed that thatching ant colonies were unaffected by grazing levels in either treated or untreated sites.

Dittel et al. (2018) studied the effects of livestock grazing on a mechanical treatment that hand-thinned severely degraded Phase II juniper woodlands and left trees where they fell. They found that low intensity grazing with deferred rest-rotation did not appear to affect herbaceous species compared to the exclosure. The study period was only three years and the authors noted that longer post-monitoring would be beneficial.

4. DATA GAPS & RECOMMENDATIONS

Our review of the literature resulted in some general observations and recommendations. Treatment results are site specific, and broad conclusions about effects over wider landscapes are not yet substantiated by research. Aggregating and analyzing data from past studies in meta-analyses will provide stronger support for assumptions and point to areas where such support is lacking. There is also an urgent need for multi-year post-treatment monitoring. The few long-term post-treatment monitoring projects available show that initial results may change over time. The long-term influence of land uses such as livestock grazing (which is rarely controlled for in post-treatment monitoring) on treatments may account for some of this change. Climate change is another factor to account for in future research. For example, woody plants might decline across the West according to some climate models, perhaps obviating the impetus for removing them in the future.

a. Meta-analysis: Some researchers are turning to meta-analyses to understand the variability and complexity in the results of mechanical vegetation treatments. While we did conduct a type of meta-analysis through our summary charts for various response categories, true meta-analyses are needed that, through the use of Effect Size statistics, test for significant trends across large pools of data where the results of separate studies are data points in the analysis. This is the approach currently used by Wilder et al. (2018) and Riginos et al. (in review), who conducted a meta-analysis of data from unpublished post-treatment monitoring reports from scores of sagebrush treatments implemented by the Utah Watershed Restoration Initiative in order to test for overall effects of mechanical sagebrush reduction on sagebrush, perennial and annual grasses and forbs, and ground cover (Riginos et al. in review) and to test for overall trends in seeding success following mechanical sagebrush treatment (Wilder et al. 2018). And for the Utah Watershed Restoration Initiative pinyon-juniper treatments, Monaco and Gunnel (unpublished data, MS in prep) used Effect Size statistics to assess vegetation change at 165 pinyon-juniper treatment sites distributed across three ecoregions, three plant community types, two woodland, and two successional phases over a 15-year period. More meta-analytic approaches along these lines are sorely needed for other vegetation treatment response categories and variables.

b. Monitoring Needs: Many of the earlier studies on post-treatment outcomes have been short term studies, usually less than five years. As the body of literature grows and longer-term studies become available, new patterns of response may emerge (Bates et al. 2007; Beck et al. 2012). Beck et al. (2012) and Bombaci and Pejchar (2016) point out that most vegetation treatment studies have been on specific, fine-scale management actions that only address short-term effects immediately post-treatment. They recommend that experiments be conducted over longer-term temporal and spatial scales. We also are deficient in reference areas with which to compare treated areas, especially for sagebrush communities. Vegetation treatment projects should thus incorporate a system of large exclosures in the post-treatment study design. These will be invaluable in future attempts to understand effects of management.

c. Post-treatment land use: One of the biggest data gaps in the ecological restoration literature is well-designed, long-term replicated studies of the interaction between vegetation treatments and post-treatment livestock grazing. Few studies monitor success of mechanical treatments after livestock grazing is resumed. Many published studies of the effects of mechanical treatments do not mention post-treatment grazing management at all. Of the over 300 citations in this review, only seven reported on comparisons between grazed and ungrazed mechanical treatments, and of those, none monitored for longer than five years. These authors thought it possible that there would be additional changes in response variables that were not captured by the time period of the study. They call for longer-term monitoring. Grazing by big game and wild horses in recent treatments is yet another area that warrants further study.

The majority of studies that reported increased cover, frequency, productivity, or density of native perennial grasses or forbs following mechanical treatment were conducted in exclosures, or only sampled during the brief (often two years or two growing seasons) post-treatment livestock exclusion period. In studies where grazing did occur in the study area, it was usually characterized as light to moderate (e.g., Bates et al. 2009; Davies et al. 2018; Dittel et al. 2018) This level of use is not always explicitly described, but Davies et al. (2018) define it as between 35 to 45% utilization



A pinyon-juniper chaining project in Utah's West Desert. (Photo: Ray Bloxham)

with non-consecutive season grazing and periodic rest. Holechek et al. (2006) recommend no greater than 40% utilization and lower in drought conditions or on rangeland in poor condition. However, ungrazed or lightly grazed conditions are atypical on public lands, particularly in sagebrush communities, so these results may not represent the common management situation. Most sagebrush communities on public lands are grazed, many at more than moderate levels. In practice, many management units adhere to a “take half, leave half” strategy of 50% utilization (e.g., Ogle 2009; Oregon State University 1988; Pratt and Rasmussen 2001; Sprinkle, 2018) or even higher in seedings (Busby and Gifford 1981). Alternation of grazing season and periodic total rest of pastures is not a common management prescription. Moreover, the standard 2-year post-treatment deferral of grazing is not always adequate for recovery (Gottfried 2004), and it is not always complied with. 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 [T]ypically 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.

Sometimes sites rapidly regress into pretreatment conditions depending on post-treatment management (Archer et al. 2011) when they should be managed to support long-term, resilient ecosystem processes. We must address the underlying issues causing resource problems, not just respond to the symptoms. No treatment can be successful if post-treatment management, including livestock grazing levels, is not appropriate. Instead, the goal of treatments should be to maintain ecosystem function once processes are restored so as not to require treatment in the future.

d. Soil erosion: Mitigating soil erosion is a critical component of treatment planning. An emphasis should be placed on methods with less soil disturbance. This is most often hand thinning, which is resource-intensive and often discarded in favor of more efficient, but soil disrupting, methods. Soil stability is greatly enhanced by biological soil crust on some arid sites (e.g., Bowker et al. 2008), but mechanical treatments remove and destroy this beneficial functional group and potentially leave a treatment exposed to higher rates of erosion. While Bowker (2007) has shown that biological soil crust is readily propagated from inoculation, this field is in its infancy and is in need of more study and under variable environmental conditions. Facilitating biological soil crust re-establishment has the potential to more quickly return some sites to a higher state of ecological function, and this technique should be evaluated for incorporation into more restoration projects in arid and semi-arid areas. There has not been research on

this topic, but the suggestion that thick layers of mulch from mastication treatment are inimical to biological soil crust establishment seems unlikely to play out through research.

e. Pinyon-juniper woodland research on fire frequency and carbon sequestration: There is a great need for more information on the degree to which fuel reduction treatments result in fewer wildfires in pinyon-juniper communities. Some recent research cites climatic factors and human activity rather than pinyon and juniper fuel loads as the chief cause of increasing frequency and extent of wildfire. Other studies suggest that fire intensity might be influenced by the recent increase in trees. There is a consensus, however, that exotic annuals such as cheatgrass promote fire and efforts must be made to arrest their expansion to prevent catastrophic habitat degradation. Many studies note an increase in these species with treatment along with, or instead of, more desirable perennial grasses and forbs. Since this is such a big risk in many areas, applying uniform fire and structural treatments in pinyon-juniper woodlands for the purpose of reducing fire risk must only be undertaken with great caution. Areas that already have large populations of flammable exotics may be unsuitable for fuel reduction treatments, especially if future research indicates that treatments are not effective at reducing wildfire.

f. How best to predict treatment success? Land managers would benefit from additional training and access to various management tools in order to better evaluate sites and predict the likelihood of treatment success. Following the guidance in a technical reference such as Miller et al. (2014a), Miller et al. (2015), and Pyke (2015a and b) could improve effectiveness of treatments. In addition, since the ability to effectively predict outcomes of an individual mechanical vegetation treatment is limited, small-scale field tests and independent scientific validation are needed to ensure that the proposed treatment method actually does lead to the intended ecological conditions. Also, the possibility that recent pinyon-juniper expansion into a site might actually be recolonization from past human removal underscores the need for practitioners contemplating pinyon-juniper treatments to first determine the soil type and NRCS Ecological Site Type and associated Ecological Site Description for the proposed project area to determine whether pinyon and/or juniper are in fact the suitable and expected overstory species for that soil type and Ecological Site Type (Miller et al. 2014a; Miller et al. 2014b; Miller et al. 2015). These are help-

ful tools that should always be consulted when planning mechanical treatments or any other restoration efforts. Lastly, it should be kept in mind that changing habitat conditions, even if meant to benefit a myriad of species, will still almost always create winners and losers. When removing a habitat type from the landscape, whether it is sagebrush or pinyon-juniper woodland, maintaining heterogeneous landscape mosaics at the proper spatial and temporal scale provides for maximum diversity and reduces disturbance patch size for dependent wildlife.

Another goal of sagebrush treatments is to diversify the age classes of sagebrush. However, Beck et al. (2012) reported that large-scale treatments are more likely to result in even-aged sagebrush communities than plants in untreated sites. Other researchers have emphasized that gradual aging and death of individual sagebrush plants, rather than treatments that create even-aged stands, is a better process for achieving maximum diversity and an optimal vegetative pattern for wildlife habitat (Lommasson 1948; Passey and Hugie 1962).

Many studies pointed out the need to seed the site to encourage desirable vegetation, avoid increases of non-native plants, and reduce soil erosion. Wilder et al. (2018) recommend that seed mixes should be based on knowledge of species interactions to avoid allowing one seeded species to outcompete another. Many studies do not address the benefits of seeding with native versus non-native species. There may be important ecological impacts from seeding with non-natives if they outcompete native species, especially on a long-term basis or where a return to native species is desired. In practice, however, constituents of seed mixes are often based on what is available or least expensive. An effort should be made to cultivate locally adapted sources of seed by giving guarantees to businesses that their seeds will be purchased (McArthur and Young 1999).

Treatments are more successful when conducted before sites are highly degraded. Treatment dollars should be put into pinyon-juniper and sagebrush communities that are healthy enough, and before desirable perennial plant cover is lost, to resist non-native species invasion (Young et al. 2013b). Severely degraded sites may have passed a threshold that will require an inordinate effort to restore (Davies et al. 2012; Davies et al. 2016). This also speaks to proper land management that does not allow conditions to deteriorate in the first place. If funds are not available to address resource concerns as they arise, then at least efforts can be made to refrain from anthropogenic activities that make resource problems worse.

5. SUMMARY AND CONCLUSIONS

Do treatments accomplish the goals we intend for them? Do they prevent soil erosion, increase desired plant species, improve wildlife habitat, and restore ecological functioning? Treatment results are very specific to individual locations. Finding patterns in effects across a large geographic area and variety of site characteristics is difficult. As McIver et al. (2014) concluded, “[S]ubstantial among-site variation in key ecological attributes will likely always cloud our ability to predict specific outcomes for many sites. Interannual variation, especially in the availability of water in spring, blurs predictive ability further.” Archer and Predick (2014) agree, stating that “our ability to predict ecosystem responses to treatments is limited for many attributes, (e.g., primary production, land surface-atmosphere interactions, biodiversity conservation) and inconsistent for others (e.g., forage production, herbaceous diversity, water quality/quantity, soil erosion, and carbon sequestration).” The ecological legacies of past and current management make prediction of outcomes even more difficult (Monaco et al. 2018; Morris et al.

2011; Morris and Rowe 2014; Morris et al. 2014). The complexities involved in disentangling variables across such a wide variety of vegetation communities and ecological sites over the West may be best addressed with meta-analyses and the results used to inform future vegetation manipulations.

Where we could, we completed summary charts on the outcomes of hundreds of studies, grouped into six response categories (and reprinted two other summary charts by Bombaci and Pejchar 2016). Herbaceous understory responses to treatments were highly variable. In pinyon-juniper communities, most studies showed no significant effect of treatments on perennial grasses and forbs. However, where there were significant results, treatments elicited more positive responses (increases in cover) in grasses and forbs than negative responses. Non-native annuals responded positively in about half of the studies. The other half showed no significant response. In sagebrush communities, most studies showed no significant effects of treatments on



Land managers might take a step back and address the stressors under their control that may have contributed to the need for treatment in the first place before putting significant resources into very large treatments as a first course of action. (Photo: Ray Bloxham)

perennial grasses and forbs. Of the studies that did have significant responses, there were slightly more positive than negative responses for forbs. Perennial grasses, however, showed far more positive response than negative to treatment. For non-native exotic herbaceous species, studies were almost evenly divided between no significant response and positive response. Studies on the effects of treatments on wildlife are also variable. For some bird species, especially pinyon-juniper obligates, there is an overall negative response to treatments removing trees. Eleven of the 22 studies on sagebrush treatments did indicate positive effects on wildlife. Five studies showed negative effects, and six found no significant effects. Of the five studies of pinyon-juniper treatment effects on sage-grouse, three showed positive effects and two showed non-significant effects. Of the 11 studies of sagebrush treatment effects on sage-grouse, four were positive, three were negative, and four showed no significant effects. And in terms of soil-erosion related response variables, the majority of studies reviewed showed no significant response of either run-off or erosion to mechanical treatment. Some studies find treatments decrease runoff and erosion, but more studies find treatments increase runoff and erosion. Results for studies addressing hydrological effects of mechanical treatments similarly had mixed results, and other literature reviews we reviewed concluded that mixed results can reflect the very different precipitation regimes where studies are conducted.

The studies featured in this literature review indicate that treatments are not “one size fits all.” Ecosystems are comprised of complex biotic and abiotic factors, and vegetation treatments aiming to restore ecosystem function should take complexity into account to be successful. Managers need to consider multiple variables in planning treatments ranging from small-scale (e.g., soil texture, percent cover of herbaceous perennials) to large-scale (e.g., elevation, drought forecasts, dominant vegetation community). However, they are subject to the exigencies of time and funding, so often vast acres of vegetation communities with variable site characteristics are treated with the same method that had positive results somewhere else. In the long-term, it is possible to do more harm than good, especially if bare ground or non-native species increase.

Most of our summary charts showed that treatments had no significant results on the variables we chose to review. While there may be many reasons to explain this, the possibility that the results are an accurate assessment of treatment efficacy should also be consid-



The studies featured in this literature review indicate that treatments are not “one size fits all.” Ecosystems are comprised of complex biotic and abiotic factors, and vegetation treatments aiming to restore ecosystem function should take complexity into account to be successful. (Photo: Andrew Kuhn/National Park Service)

ered. Managers might take a step back and address the stressors under their control that may have contributed to the need for treatment in the first place before putting significant resources into very large treatments as a first course of action.

In the western United States, there were historically an estimated 50 million acres of pinyon-juniper woodland (Gottfried and Severson 1994, Mitchell and Roberts 1999) and almost 250 million acres of sagebrush steppe (McArthur and Plummer 1978, cited in Germino et al. 2018). The amount of remaining intact vegetation unaltered by vegetation treatments or other anthropogenic factors such as fire, grazing, climate change, water diversions, and similar change agents is shrinking. Nonetheless, millions of acres have been treated across the West and more treatments are proposed (USGS, Digital Land Treatment Library Home Page). The current pace of activity on the ground may be outstripping our understanding of the long-term effects of these treatments and our ability to plan better restoration projects.

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APPENDIX: STUDIES SUMMARIZED IN FIGURES

Figure	Citation	Habitat Type	Treatment Type	Effect measured	Unit Measured	Number of Data Points
1	Ansley et al. 2006	Pinyon-juniper Woodland	Chaining	% Cover	Vegetation	8
1	Bates 2005	Pinyon-juniper Woodland	Cut, debris in place	Density	Vegetation	30
1	Bates et al. 2000	Pinyon-juniper Woodland	Cut, debris in place	% Basal cover	Vegetation	18
1	Baughman et al. 2010	Pinyon-juniper Woodland	Thinning, buncher	% Cover	Vegetation	2
1	Everett et al. 1985	Pinyon-juniper Woodland	“Harvest”	% Cover	Vegetation	6
1	Havrilla et al. 2017	Pinyon-juniper Woodland	Mastication	% Cover	Vegetation	16
1	Huffman et al. 2013	Pinyon-juniper Woodland	Slash	% Cover	Vegetation	9
1	Juran et al. 2008	Pinyon-juniper Woodland	Chaining	% Cover	Vegetation	65
1	Owen et al. 2009	Pinyon-juniper Woodland	Mastication	% Cover	Vegetation	4
1	Provencher & Thompson 2014	Pinyon-juniper Woodland	Chaining, feller-buncher, lop & scatter, lop pile burn, mastication	% Cover	Vegetation	5
1	Redmond et al. 2013	Pinyon-juniper Woodland	Chaining	% Cover	Vegetation	3
1	Ross et al. 2012	Pinyon-juniper Woodland	Lop & scatter, mastication	% Cover	Vegetation	7
1	Schott et al. 1987	Pinyon-juniper Woodland	Cabling	% Basal cover	Vegetation	24
1	Skousen et al. 1986	Pinyon-juniper Woodland	Bulldozed, cabled, chained	% Cover	Vegetation	25
1	Skousen et al. 1989	Pinyon-juniper Woodland	Chaining	% Cover	Vegetation	1
1	Stephens et al. 2016	Pinyon-juniper Woodland	Chaining, mastication, rollerchop	% Cover	Vegetation	10
2	Chambers et al. 2014	Sagebrush	Mowing	% Cover	Vegetation	26
2	Dahlgren et al. 2006	Sagebrush	Aerator, Dixie harrow	% Cover	Vegetation	4
2	Davies et al. 2011	Sagebrush	Mowing	% Cover	Vegetation	9
2	Davies et al. 2012	Sagebrush	Mowing	% Cover	Vegetation	33
2	Monaco et al. 2018	Sagebrush	Chain harrow	% Cover	Vegetation	12
2	Omeara et al. 1981	Sagebrush	Chaining	% Cover	Vegetation	2
2	Prevey et al. 2009	Sagebrush	Hand thinning	Density	Vegetation	3
2	Prevey et al. 2010	Sagebrush	Hand thinning	% Cover	Vegetation	6
2	Pyke et al. 2014	Sagebrush	Mowing	% Cover	Vegetation	7
2	Skousen et al. 1986	Sagebrush	Bulldozing, chaining	% Cover	Vegetation	6
2	Skousen et al. 1989	Sagebrush	Cabling, chaining	% Cover	Vegetation	15

2	Stringham 2010	Sagebrush	Aerator	% Cover	Vegetation	16
2	Wambolt & Payne 1986	Sagebrush	Bush hog, plowing	% Basal Cover	Vegetation	32
2	Wilder et al. 2018	Sagebrush	Aerator, Dixie harrow	% Cover	Vegetation	8
3&4	Crow and van Riper 2010	Pinyon-juniper Woodland	Mechanical thinning	Mean relative abundance	Birds	1
3&4	Frey et al. 2013	Pinyon-juniper Woodland	Cutting & slash mulching	% use of total locations	Greater sage grouse	1
3&4	Howard et al. 1987	Pinyon-juniper Woodland	2-way cabling	Pellet deposition rates	Mule deer and lagomorphs	2
3&4	Jehle et al. 2006	Pinyon-juniper Woodland	Burning	No. birds/ha	Green-tailed towhee	1
3&4	Kleintjes et al. 2004	Pinyon-juniper Woodland	Cutting & slash mulching	Mean # butterflies/ transect; Mean no. species/transect	Butterflies	2
3&4	Knick et al. 2014	Pinyon-juniper Woodland	Burning	Mean no. of detections	Sagebrush obligate birds	1
3&4	Kruse 1994	Pinyon-juniper Woodland	Fuelwood harvesting	Counts (total no. captured/yr.)	Small mammals	1
3&4	Kundaali and Reynolds 1972	Pinyon-juniper Woodland	Uprooting all, thinning, uprooting & burning	Pellet counts	Desert cottontail	3
3&4	McIver and Macke 2014	Pinyon-juniper Woodland	Prescribed burning, cutting, or mowing	Mean total abundance	Butterflies	3
3&4	Montblanc et al. 2007	Pinyon-juniper Woodland	Burning	Mean abundance	Ants	1
3&4	Radke et al. 2008	Pinyon-juniper Woodland	Burning	Mean abundance	Lizards and Invertebrates	2
3&4	Reemts and Cimprich 2014	Pinyon-juniper Woodland	Hydro-axe and felling	No. of vireo territories	Black-capped vireos	1
3&4	Sedgwick and Ryder 1987	Pinyon-juniper Woodland	Chaining	Bird counts (no./100ha); small mammal counts (total # captured)	Small mammals and birds	2
3&4	Severson 1986	Pinyon-juniper Woodland	Bulldozing, burning, thinning	Counts (total # captured)	Small mammals	3
3&4	Short et al. 1977	Pinyon-juniper Woodland	Thinning, partial removal (bulldozing), complete removal, complete removal + burned slash	Pellet counts	Mule deer and Elk	8
3&4	Smith and Urness 1984	Pinyon-juniper Woodland	Burning	Counts (total # captured)	Small mammals	1
3&4	Willis and Miller 1999	Pinyon-juniper Woodland	Cutting (method not stated)	Counts (total no. captured/yr.)	Small mammals	1
5	Carlisle et al. 2018	Sagebrush	Mowing	Abundance	Birds	3
5	Davies et al. 2009	Sagebrush	Mowing	Percentage	Crude protein	4

Literature Review of Mechanical Vegetation Treatments (2019)

5	Lee 2008	Sagebrush	Mowing	Pellet abundance	Rabbit species	3
5	Pierce et al. 2011	Sagebrush	Dixie harrow	Remote cameras (edge distance model), pellet abundance	Rabbit species	12
6	Baxter et al. 2017	Sagebrush	Chain harrow and/or bushhog (mower)	Occupancy	Sage-grouse	1
6	Cook et al. 2017	Pinyon-juniper Woodland / Sagebrush	Mastication and chaining	Use	Sage-grouse	1
6	Dahlgren et al. 2006	Sagebrush	Dixie Harrow and Lawson Aerator	Pellet counts	Sage-grouse	3
6	Dahlgren et al. 2015	Sagebrush	Lawson aerator, disking, chain harrow	Number (flushing)	Sage-grouse	1
6	Frey et al. 2013	Pinyon-juniper Woodland / Sagebrush	Hand cutting	Use	Sage-grouse	2
6	Graham 2013	Sagebrush	Chain harrow	Pellet counts	Sage-grouse	1
6	Sandford 2017	Pinyon-juniper Woodland / Sagebrush	Chaining, lop & scatter, and mastication	Nest and brood success	Sage-grouse	1
6	Severson et al. 2017	Pinyon-juniper Woodland / Sagebrush	Chain saws and feller-busher	Frequency of nesting	Sage-grouse	1
6	Stringham 2010	Sagebrush	Lawson aerator	Pellet counts, occupancy	Sage-grouse	3
6	Swenson et al. 1987	Sagebrush	Ploughing	Lek attendance	Sage-grouse	1
6	Wallestad 1975	Sagebrush	Not specified	Lek attendance	Sage-grouse	1
7	Brockway et al. 2002	Pinyon-juniper Woodland	Cut	Erosion/sediment	Soil response (mm)	3
7	Gifford 1973	Pinyon-juniper Woodland	Chained	Erosion/sediment, runoff	Runoff yield (area cm), sediment yield(kg)	32
7	Gifford et al. 1970	Pinyon-juniper Woodland	Chained	Runoff	Runoff yield (Tons per acre)	20
7	Hastings et al. 2003	Pinyon-juniper Woodland	Cut (thinning)	Erosion/sediment	Sediment yield (Mass per unit area)	2
7	Jacobs 2015	Pinyon-juniper Woodland	Thinning	Erosion/sediment	Sediment yield (kg/ha)	1
7	Noelle et al. 2017	Pinyon-juniper Woodland	Slash	Erosion/sediment, runoff	Depth (mm), g/m ²	2
7	Owens et al. 2009	Pinyon-juniper Woodland	Mastication	Erosion/sediment, runoff	Median slake test score	4
7	Pierson et al. 2007	Pinyon-juniper Woodland	Hand cutting	Erosion/sediment, runoff	Rill runoff rate, rill sediment concentration, runoff yield, sediment yield	16

7	Pierson et al. 2015	Pinyon-juniper Woodland	Cut, mastication	Erosion/ sediment, runoff	Flow velocity, sediment yield, runoff yield	81
7	Ross et al. 2012	Pinyon-juniper Woodland	Lop & scatter, mastication	Erosion/ sediment	Median sediment yield	3
7	Roundy et al. 2016	Pinyon-juniper Woodland	Chaining	Erosion/ sediment, runoff	Runoff yield (L), sediment yield (g)	10
8	Bates et al. 2000	Pinyon-juniper Woodland	Cut	Soil moisture	Volumetric soil water	48
8	Cline et al. 2010	Pinyon-juniper Woodland	Shredding	Infiltration rate	mm/hour	12
8	Deboodt et al. 2009	Pinyon-juniper Woodland	Cutting	Flow/days of water, soil moisture	Number of days, cubic feet per second, gallons per minute, percent moisture	8
8	Gifford 1982	Pinyon-juniper Woodland	Chaining	Soil moisture	Water/152 cm soil profile	39
8	Mollnau et al. 2014	Pinyon-juniper Woodland	Cutting	Soil moisture	Gravimetric moisture	6
8	Roundy et al. 2014b	Pinyon-juniper Woodland	Cutting	Soil moisture	Number of days	4
8	Williams et al. 2018	Pinyon-juniper Woodland	Cutting, mastication	Infiltration rate, soil moisture	(mm * h ⁻¹) ¹ , Percent moisture	41