January 12, 2021

By Email and Certified Mail

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Aurelia Skipwith, Director
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Re: Sixty-day Notice of Intent to Sue for Violations of Section 7 of the Endangered Species Act Relating to the Bureau of Land Management's Failure to Properly Consult with the U.S. Fish and Wildlife Service Regarding Impacts to Listed Species From Activities Conducted Under the Programmatic Environmental Impact Statement for Fuel Breaks in the Great Basin and the Programmatic Environmental Impact Statement for Fuels Reduction and Rangeland Restoration in the Great Basin.

Dear Secretary Bernhardt and Director Skipwith:

The Center for Biological Diversity, the Sierra Club, the Southern Utah Wilderness Alliance, and Western Watersheds Project hereby provide notice, pursuant to Section 11(g) of the Endangered Species Act ("ESA"), 16 U.S.C. §§ 1531–1544, that the Bureau of Land Management ("BLM") and the U.S. Fish and Wildlife Service ("FWS") are in violation of the ESA. BLM has failed to properly consult with FWS regarding the impacts to threatened and endangered aquatic species from two major land management initiatives in the Great Basin region—the February 2020 Programmatic Environmental Impact Statement for Fuel Breaks in the Great Basin ("Fuel Breaks PEIS") and the November 2020 Programmatic Environmental Impact Statement for Fuels Reduction and Rangeland Restoration in the Great Basin ("Fuels Reduction PEIS"). Further, FWS has arbitrarily and capriciously concurred with BLM's conclusion that activities conducted under the Fuel Breaks PEIS and the Fuels Reduction PEIS are not likely to adversely affect threatened and endangered terrestrial species.

I. Project Description

Both the Fuel Breaks PEIS and the Fuels Reduction PEIS authorize large-scale disturbance of vegetation, soils, and wildlife habitat across a six-state, 223-million-acre project area. The Fuel Breaks PEIS would allow for the construction of 11,000 miles of linear fuel breaks, each up to 500 feet wide, while the Fuels Reduction PEIS would permit high-impact vegetation removal methods including chaining, mastication (wood-chipping), prescribed fire, so-called "targeted" livestock grazing, and herbicide application across 38.5 million acres of public lands. Both the Fuel Breaks PEIS and Fuels Reduction PEIS rely on unproven methods and untested assumptions, meaning that both projects together comprise a grand experiment in intensive land management on a scale never before attempted. *See* D.J. Shinneman, et al., A Conservation Paradox in the Great Basin—Altering Sagebrush Landscapes with Fuel Breaks to Reduce Habitat Loss from Wildfire, USGS Open File Report 2018-1034 (2018); A. Jones, Do Mechanical Vegetation Treatments of Pinyon-Juniper and Sagebrush Communities Work? A Review of the Literature (Wild Utah Project 2018).

Notwithstanding the ambitious and unprecedented nature of these projects, BLM and FWS have ignored or minimized potential impacts to threatened and endangered species, in violation of the ESA. According to BLM's own analysis, the project area is home to over 130 listed species. Many of these threatened and endangered species are endemic to isolated and ecologically unique environments, meaning that even seemingly small impacts fuel break construction or fuels reduction could have significant implications for their long-term survival and recovery. BLM, however, failed to even consider impacts to listed aquatic species, assuming without analysis that "design features" such as riparian buffers will prevent all direct, indirect, and cumulative impacts across the entire project area for the foreseeable future. BLM and FWS also failed to adequately consider impacts to listed terrestrial species, focusing only on direct impacts from project activities and ignoring various foreseeable indirect and cumulative consequences of the large-scale ecological manipulation authorized under each PEIS. These failures violate Section 7 of the ESA and put several imperiled species at unacceptable risk of extinction.

II. Legal Background

The Endangered Species Act is "the most comprehensive legislation for the preservation of endangered species ever enacted by any nation." *Tenn. Valley Auth. v. Hill*, 437 U.S. 153, 180 (1978). In enacting the ESA, Congress indicated that its purpose is "to provide a means whereby the ecosystems upon which endangered species and threatened species depend may be conserved [and] to provide a program for the conservation of such . . . species" while also declaring its policy "that all Federal . . . agencies shall seek to conserve [such] species." 16 U.S.C. § 1531(b)-(c). The Supreme Court has described the ESA as "a conscious decision by Congress to give endangered species priority over the 'primary missions' of federal agencies." *Hill*, 437 U.S. at 185.

The “heart of the ESA” is the section 7 consultation requirement. *W. Watersheds Project v. Kraayenbrink*, 632 F.3d 472, 495 (9th Cir. 2011). Section 7 requires each federal agency to “insure that any action . . . is not likely to jeopardize the continued existence of any endangered species or threatened species or result in the destruction or adverse modification of habitat of such species.” 16 U.S.C. § 1536(a)(2). “Only after the [agency] complies with [Section 7] can any activity that may affect the protected [species] go forward.” *Pacific Rivers Council v. Thomas*, 30 F.3d 1050, 1055-57 (9th Cir. 1994). Importantly, the Ninth Circuit has recognized Congress’ intent that in making determinations under this section, agencies must “give the benefit of the doubt to the species.” *Connor v. Burford*, 848 F.2d 14441, 1454 (9th Cir. 1988) (citing H.R. Conf. Rep. No. 96-697, 96th Cong., 1st Sess. 12, reprinted in 1979 U.S.C.C.A.N. 2572, 2576 (Dec. 11, 1979).

The first step in complying with section 7 is to obtain “a list of any listed or proposed species or designated or proposed critical habitat that may be present in the action area.” 16 U.S.C. § 1536(c)(1); 50 C.F.R. § 402.12(c)(d). If a listed species “may be present” in the action area, the agency must complete a biological assessment to determine if the proposed action “may affect” the listed species. 16 U.S.C. § 1536(c)(1); 50 C.F.R §§ 402.12(f), 402.14(a), (b)(1). Any agency action that “may affect” a listed species or critical habitat gives rise to the formal consultation requirement under Section 7. *Karuk Tribe of Cal. v. United States Forest Serv.*, 681 F.3d 1006, 1027 (9th Cir. 2012).

In the Ninth Circuit, “the minimum threshold for an agency to trigger consultation with the Wildlife Service is low.” *Kraayenbrink*, 632 F.3d at 496. “[A]ny possible effect, whether beneficial, benign, adverse, or of an undetermined character, triggers the formal consultation requirement.” *Id.* (citing 51 Fed. Reg 19,926, 19,949 (June 3, 1986); *Cal ex rel. Lockyer v. U.S. Dept. of Agric.*, 575 F.3d 999, 1018-19 (9th Cir. 2009)); *see also Ctr. for Biological Diversity v. United States BLM*, 698 F.3d 1101, 1122 (9th Cir. 2012) (“Essentially, petitioners need to show only that an effect on listed species or critical habitat is plausible.”).

ESA Section 7(d), meanwhile, prohibits the action agency from making any “irreversible or irretrievable commitment of resources” prior to the conclusion of consultation which would have “the effect or foreclosing the formulation or implementation of any reasonable and prudent alternative measures.” 16 U.S.C. § 1536(d); *Conner*, 848 F.2d at 1453.

Although “tiering” is not described anywhere in the ESA or its implementing regulations, see *NRDC v. Rodgers*, 381 F. Supp 2d 1212, 1227 n.27 (E.D. Cal. 2005), courts in the Ninth Circuit have occasionally approved of “tiering,” or “programmatic environmental analysis supplemented by later project-specific environmental analysis in the ESA context.” *Id.* at 1227; *see also Gifford Pinchot Task Force v. United States Fish and Wildlife Serv.*, 378 F.3d 1059, 1067-68 (9th Cir. 2004). Even where tiering is appropriate, however, it is improper for an agency to completely defer analysis of particular types of impacts to future site-specific consultations. *See, e.g. Conner*, 848

F.2d at 1457-58 (holding that FWS violated the ESA because it did not prepare a "comprehensive" biological opinion prior to project implementation "assessing the potential impacts of all post-leasing activities").

III. Notice of Violation

BLM has violated ESA Section 7 with respect to aquatic species. Both the Fuel Breaks PEIS and the Fuels Reduction PEIS list waterways, ponds, springs, and riparian areas as "analysis exclusion areas." Fuel Breaks PEIS at 4; Fuels Reduction PEIS at 2-1. Neither document contains any analysis of the likely direct, indirect, or cumulative impacts to these areas from project implementation. Although BLM prepared a biological assessment ("BA") to accompany each PEIS, the BAs expressly exclude aquatic species from their analyses. *See* Revised Biological Assessment for the Programmatic Environmental Impact Statement for Fuel Breaks in the Great Basin ("Fuel Breaks BA"), Appendix B; Biological Assessment for the Programmatic Environmental Impact Statement for Fuels Reduction and Rangeland Restoration in the Great Basin ("Fuels Reduction BA"), Appendix B. Without such analysis BLM cannot fulfill its duty under the ESA to "insure" that PEIS-authorized activities do not "jeopardize the continued existence of any endangered species or threatened species or result in the destruction or adverse modification of habitat of such species." 16 U.S.C. § 1531.

BLM explains throughout each PEIS that the application of "design features" and "conservation measures" would result in "less than significant impacts" to aquatic resources. *See, e.g.*, Fuel Breaks PEIS Appendix G; Fuels Reduction PEIS Appendices G & O. But this is not the relevant standard under the ESA. As explained above, "[A]ny possible effect, whether beneficial, benign, adverse, or of an undetermined character, triggers the formal consultation requirement." *Kraayenbrink*, 632 F.3d at 496. Moreover, a close examination of BLM's analysis and the sources cited therein reveals that both adverse and beneficial impacts to aquatic species are at least likely. For example, the Fuels Reduction PEIS admits that vegetation removal treatments may cause "short-term impacts on water quality," while both PEISs anticipate long-term beneficial impacts to aquatic ecosystems from reduced wildfire activity.

BLM's "no effect" conclusion rests largely on the implementation of aquatic buffer zones adapted from the U.S. Forest Service's INFISH riparian protection framework. *See, e.g.*, Fuels Reduction PEIS at 2-1 (citing U.S. Forest Service, Inland Native Fish Strategy Environmental Assessment, Decision Notice, and Finding of No Significant Impact (1995) ("USFS 1995")). However, INFISH admits that its intent is to "reduce" impacts to riparian areas from activities such as timber harvest and grazing, not eliminate them entirely. The Forest Service anticipated that under INFISH "some adverse effects on riparian and aquatic habitat" and "risks to water resources" would be "reduced," but nowhere does it claim that adopting the INFISH framework would result in "no effect" to listed species. *See* USFS 1995 at III-10 to III-15. Further, INFISH assumes that site-specific projects affecting ESA-listed species would be subject to formal Section 7 consultation. *Id.* at F-3. That is not the case under either the Fuel Breaks PEIS or the Fuels Reduction PEIS—each respective BA states that it is "intended to satisfy ESA Section 7 consultation" obligations for "project-level actions," and the further consultation would not occur for "treatments that fall[] within the scope" of the PEIS. Fuel Breaks BA at 1-1; Fuels Reduction BA at 1-1.

While a "tiered" approach to formal consultation is permissible under some circumstances, *see Gifford Pinchot Task Force*, 378 F.3d at 1067-68, BLM is not permitted to entirely ignore potential impacts to listed species—at either the programmatic or project level—based on a general assumption that such impacts will be "less than significant." BLM's failure to consult with FWS regarding impacts to listed aquatic species therefore violates the ESA.

BLM and FWS have also violated the ESA with regard to terrestrial species. BLM concludes in each BA that fuel breaks and fuels reduction treatments are "not likely to adversely affect" listed terrestrial species due to the implementation of "design features" and "conservation measures." But while design features and conservation may help BLM avoid or mitigate some of the direct impacts of project implementation, BLM fails to consider indirect or cumulative impacts such as habitat fragmentation, or the long-term ecological changes that may result from the proposed actions. As noted, each PEIS contemplates intensive vegetation alteration and removal across a vast geographic area, from southeastern Oregon to southern Utah. BLM further states that its main purpose is to alter basic ecological processes within this vast area, including wildfire frequency, hydrologic cycling, and the natural process of ecological succession over time from one vegetative state to another. Changes to these fundamental processes could result in significant long-term impacts to listed species, especially species with restricted ranges like the Columbia Basin pygmy rabbit and the Utah prairie dog. Widespread ecological changes could also alter predator-prey dynamics, with impacts to large predators such as the grizzly bear. Finally, BLM and FWS entirely fail to consider the impact of global climate change on species distribution and habitat use over the anticipated lifetime of the projects.

BLM's conclusion that these widespread, intensive, and ultimately experimental land management techniques are "not likely to adversely affect" listed terrestrial species is therefore invalid because it fails to consider relevant factors, draw a reasonable conclusion, or "give the benefit of the doubt to the species." *Connor*, 848 F.2d at 1454 (citing H.R. Conf. Rep. No. 96-697); *see also Mont. Wilderness Ass'n v. Fry*, 310 F. Supp. 2d 1127, 1149 (D. Mont. 2004). For the same reasons, FWS's concurrence with BLM's "not likely to adversely affect" conclusion is unreasonable and contrary to the ESA.

Additional ESA violations and harm to listed species are imminent because BLM has already started authorizing projects based on its programmatic analysis without any further project- or species-specific analysis or consultation. *See, e.g.*, Decision Record for the Wendell Cattle

Targeted Grazing (DOI-BLM-ID-T030-2021-0006-DNA) (Dec. 15, 2020), available at: https://eplanning.blm.gov/eplanning-ui/project/2009723/570. Such authorizations represent "irreversible or irretrievable commitment of resources" that cannot lawfully proceed without valid Section 7 consultation. 16 U.S.C. § 1536(d); *Conner*, 848 F.2d at 1453.

IV. Conclusion

BLM has violated the Endangered Species Act by failing to initiate formal consultation with FWS regarding impacts to threatened and endangered aquatic species. Further, BLM and FWS have not adequately considered impacts to terrestrial listed species. If BLM and FWS does not act to remedy these violations in sixty days, we will file suit in federal court. Please contact us if you have any questions or if you would like to discuss this matter.

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References/Attachments:

A. Jones, Do Mechanical Vegetation Treatments of Pinyon-Juniper and Sagebrush Communities Work? A Review of the Literature (Wild Utah Project 2019).

D.J. Shinneman, et al., A Conservation Paradox in the Great Basin—Altering Sagebrush Landscapes with Fuel Breaks to Reduce Habitat Loss from Wildfire, USGS Open File Report 2018-1034 (2018).

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

A Review of the Literature

February 2019

Lead Editor: Allison Jones, Wild Utah Project

Peer Reviewers: Sara Bombaci (Colorado State University), Marc Coles-Ritchie (Grand Canyon Trust), John Connelly (retired, Idaho Department of Fish and Game), Walter Fertig (Washington State Natural Heritage Program), Sara Goeking (USFS - Rocky Mountain Research Station), Hugh Hurlow (Utah Geological Survey), Stan Kitchen (USFS - Provo Shrub Sciences Laboratory), Tom Monaco (USDA - Agricultural Research Service), Mike Pellant (Bureau of Land Management, Idaho), Doug Shinneman (USGS - Forest and Rangeland Ecosystem Science Center), Nick Van Lanen (Bird Conservancy of the Rockies)

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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 suggests 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 removed 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.

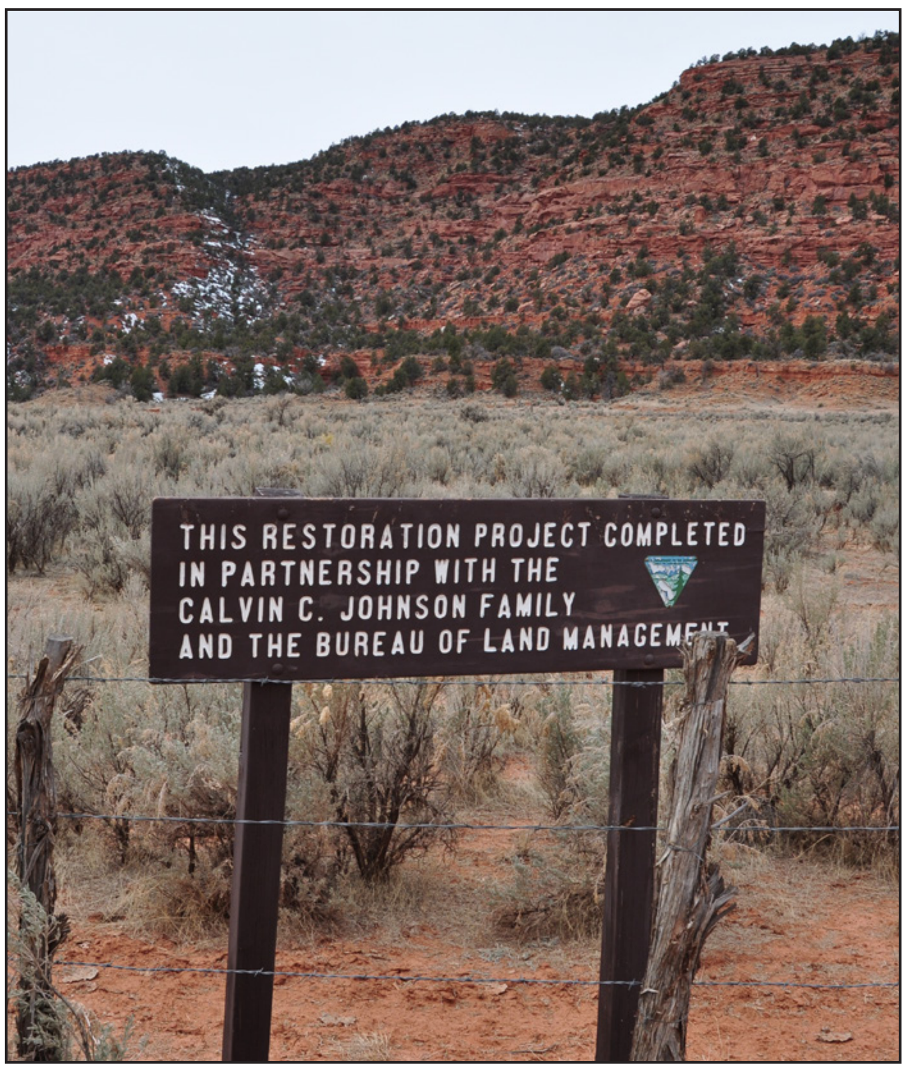
Recommendations and Conclusions

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 productivity, 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.

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

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-

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 understory 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, increasing 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-juniper 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 adaption 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 CO_2 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-Juniper 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-

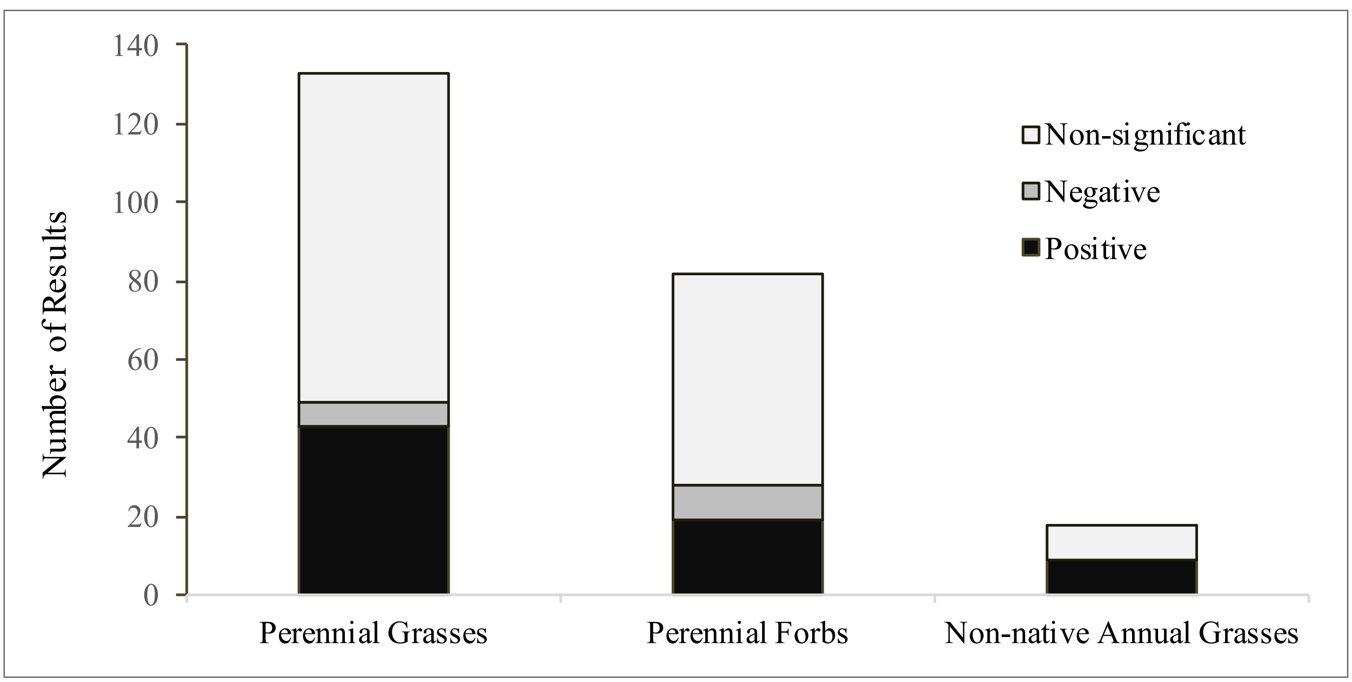


Figure 1. Number of pinyon-juniper treatment study results within herbaceous vegetation categories that found positive, negative, or non-significant results. The appendix lists all of the studies that contributed data points to this summary chart, and the response variables measured.

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 seedbank (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; Wambolt et al. 2001; Watts and Wambolt 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).

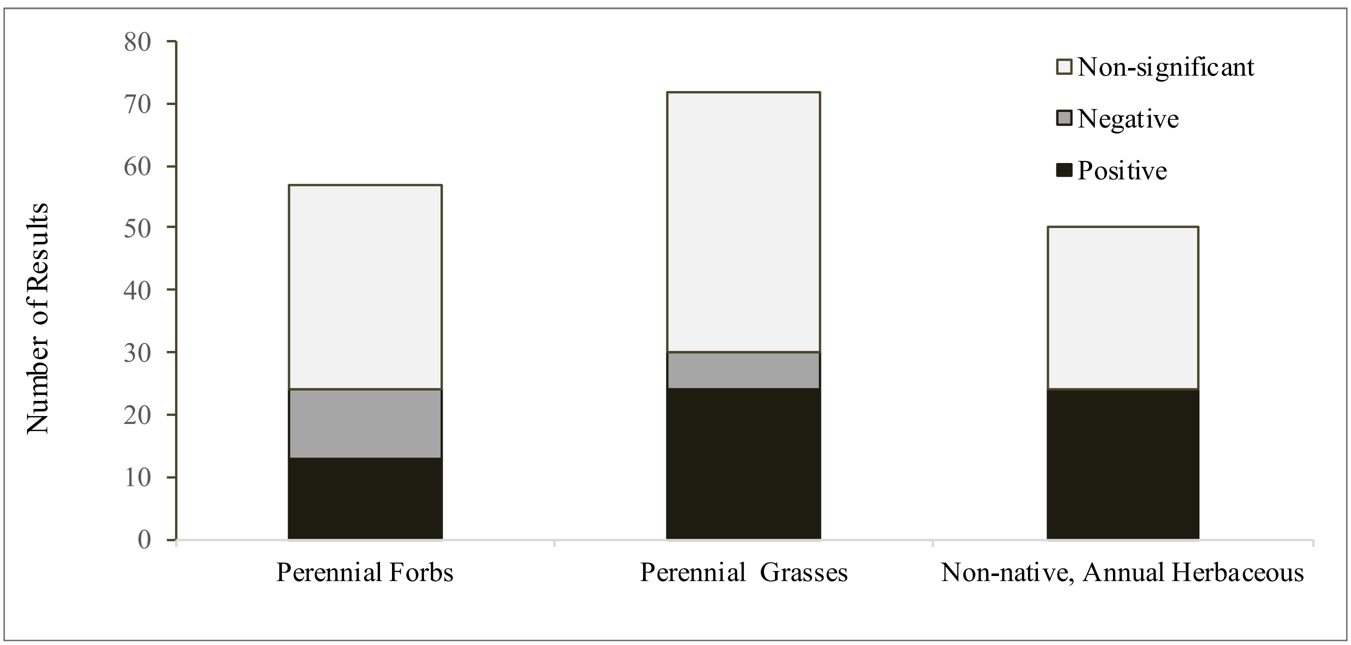


Figure 2. Number of sagebrush treatment study results within vegetation categories that found positive, negative, or non-significant results. The appendix lists all of the studies that contributed data points to this summary chart, and the response variables measured.

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 (*Prosopsis* 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 snowmelt. 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 wheatgrasses, 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 communities 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

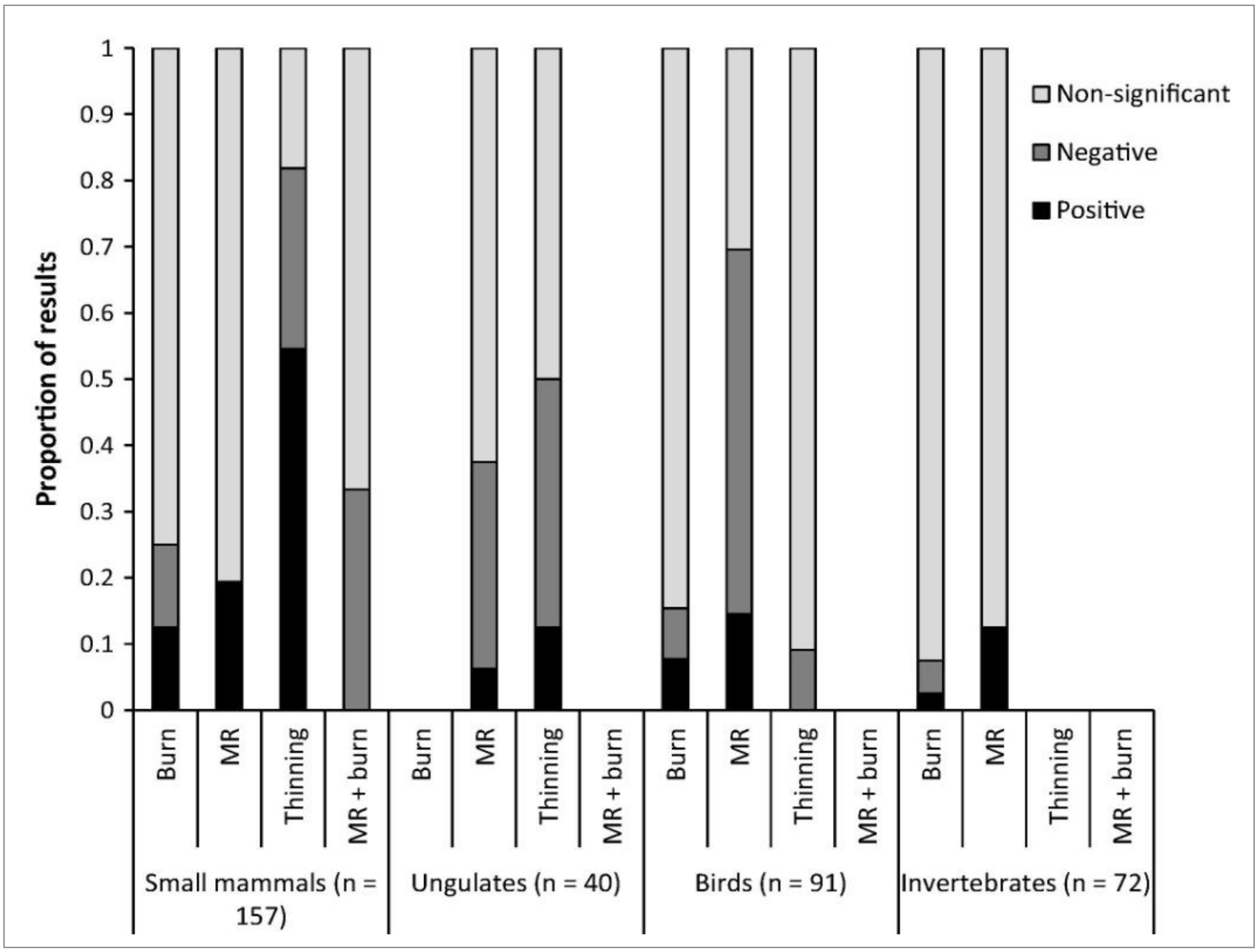


Figure 3. Proportion of study results within taxonomic groups and treatment methods documenting positive, negative, or non-significant responses to woodland reduction. Burn = prescribed fire; MR = mechanical removal (i.e., bulldozing, chaining, cutting, mowing, hydro-axing, roller-chopping, and uprooting);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

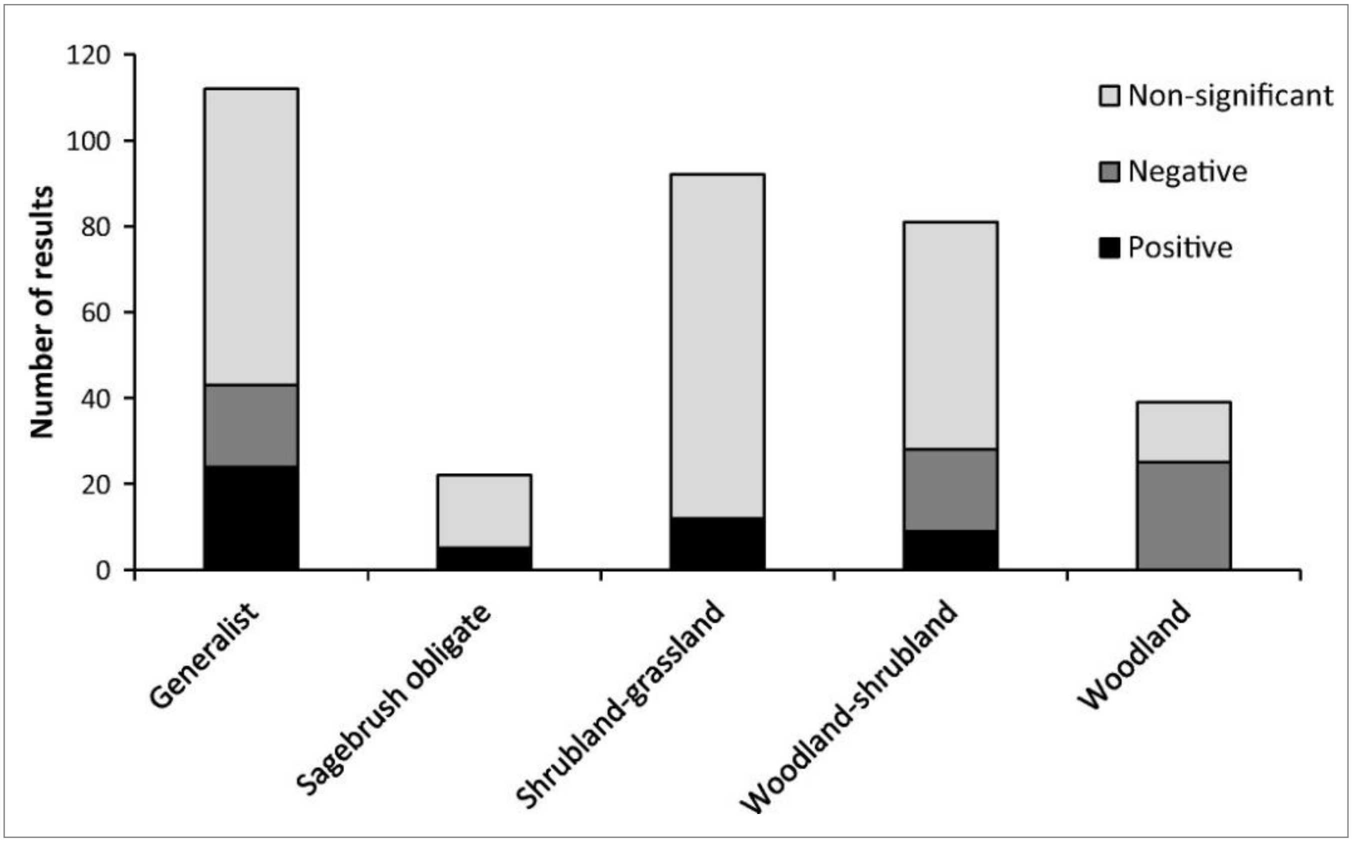


Fig. 4. Number of study results within habitat functional groups documenting positive, negative, or non-significant responses to woodland reduction treatments (reprinted from Bombaci and Pejchar 2016, with permission). The appendix lists all of the studies that contributed data points in Figures 3 and 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-treatment, 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 bear (*Ursus americanus*), golden-mantled ground squirrel (*Callospermophilus lateralis*), and rock squirrel (*Otospermophilus variegatus*) responded negatively to chaining treatments. Chipmunk (*Tamius* 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.

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)

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

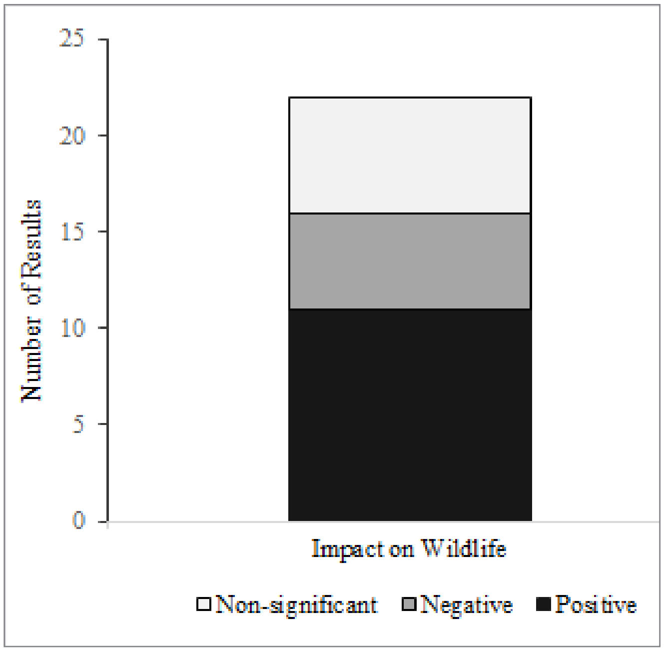


Figure 5. Number of sagebrush treatment study results that had positive, negative, or non-significant response on wildlife. The appendix lists all of the studies that contributed data points to this summary chart, and the response variables measured.

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.

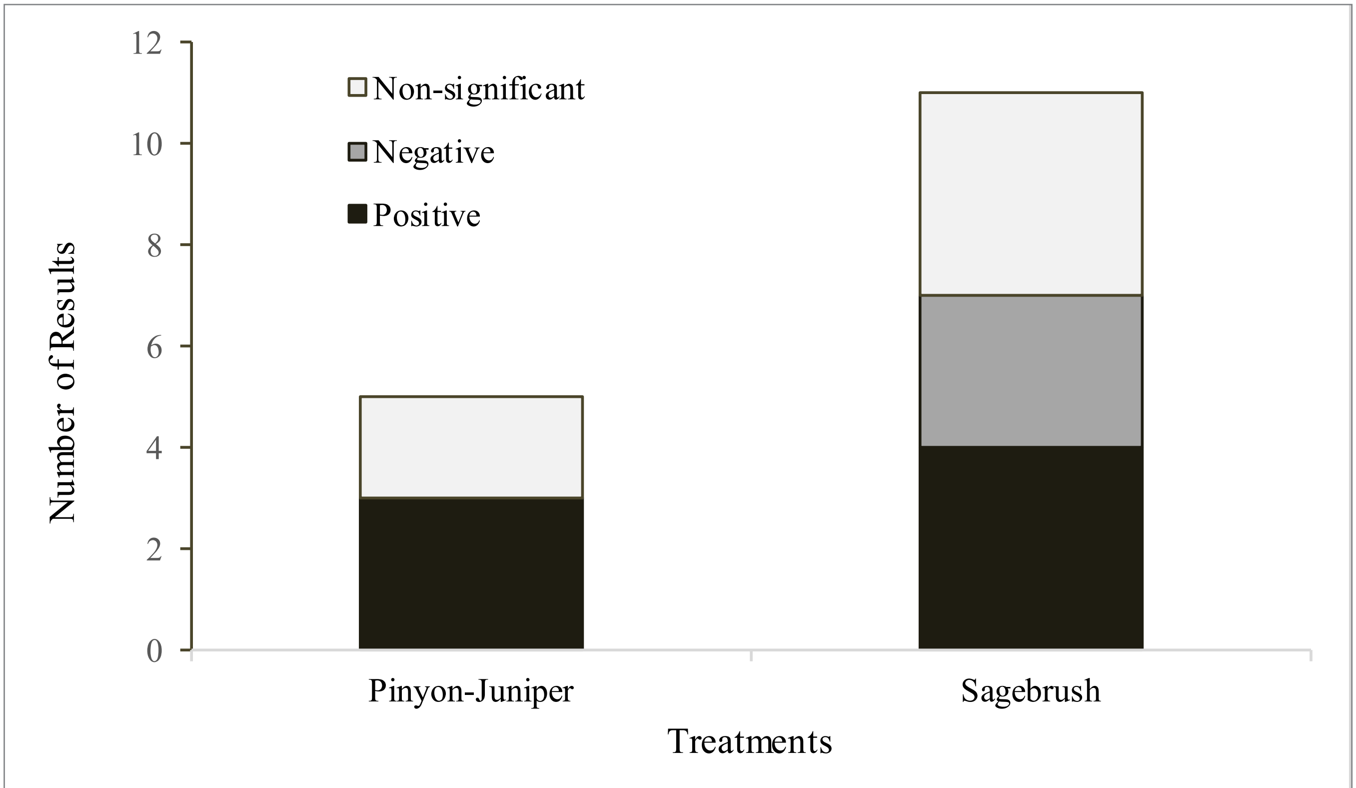


Figure 6. Number of pinyon-juniper and sagebrush treatment study results that elicited positive, negative, or non-significant response from sage-grouse. The appendix lists all of the studies that contributed data points to this summary chart, and the response variables measured.

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 selected 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 sagebrush 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 a. 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 a 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.

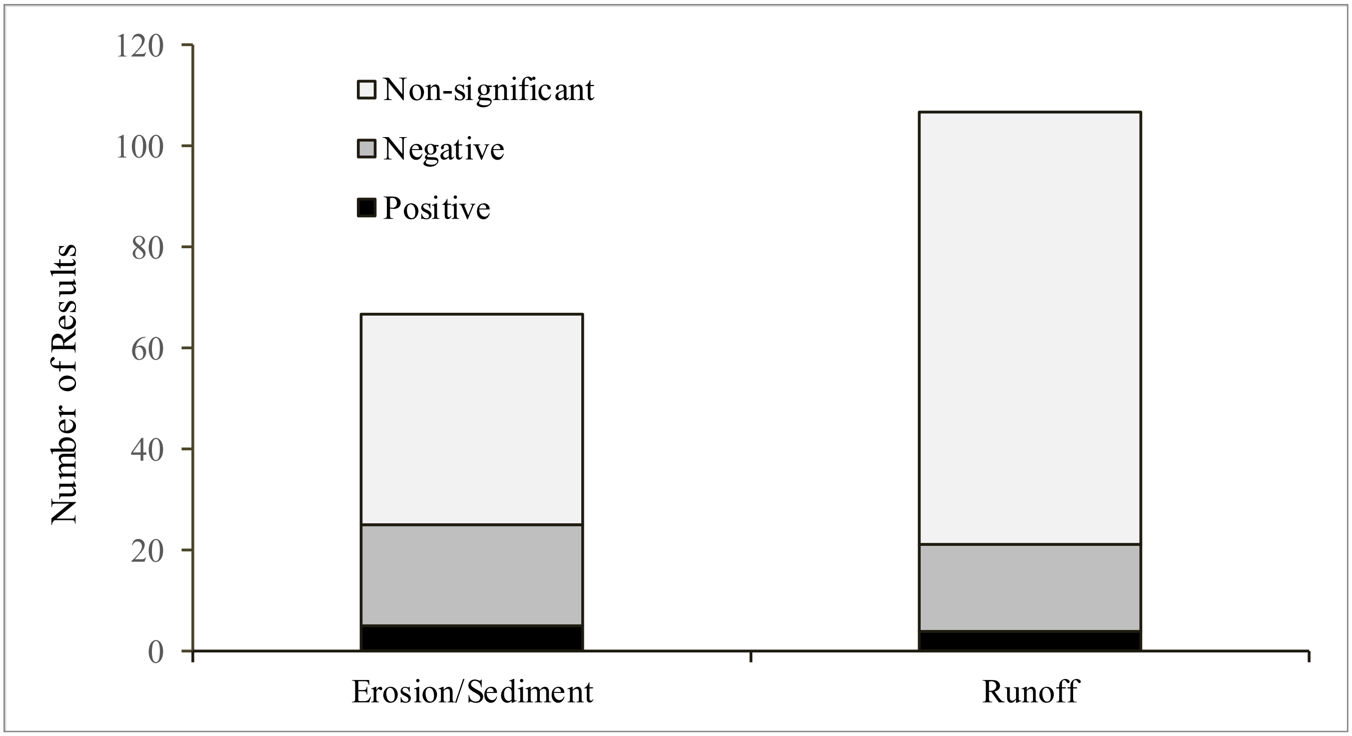


Figure 7. Number of pinyon-juniper treatment study results that had positive, negative, or non-significant response on erosion and runoff variables. The appendix lists all of the studies that contributed data points to this summary chart, and the response variables measured.

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 infiltration 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).

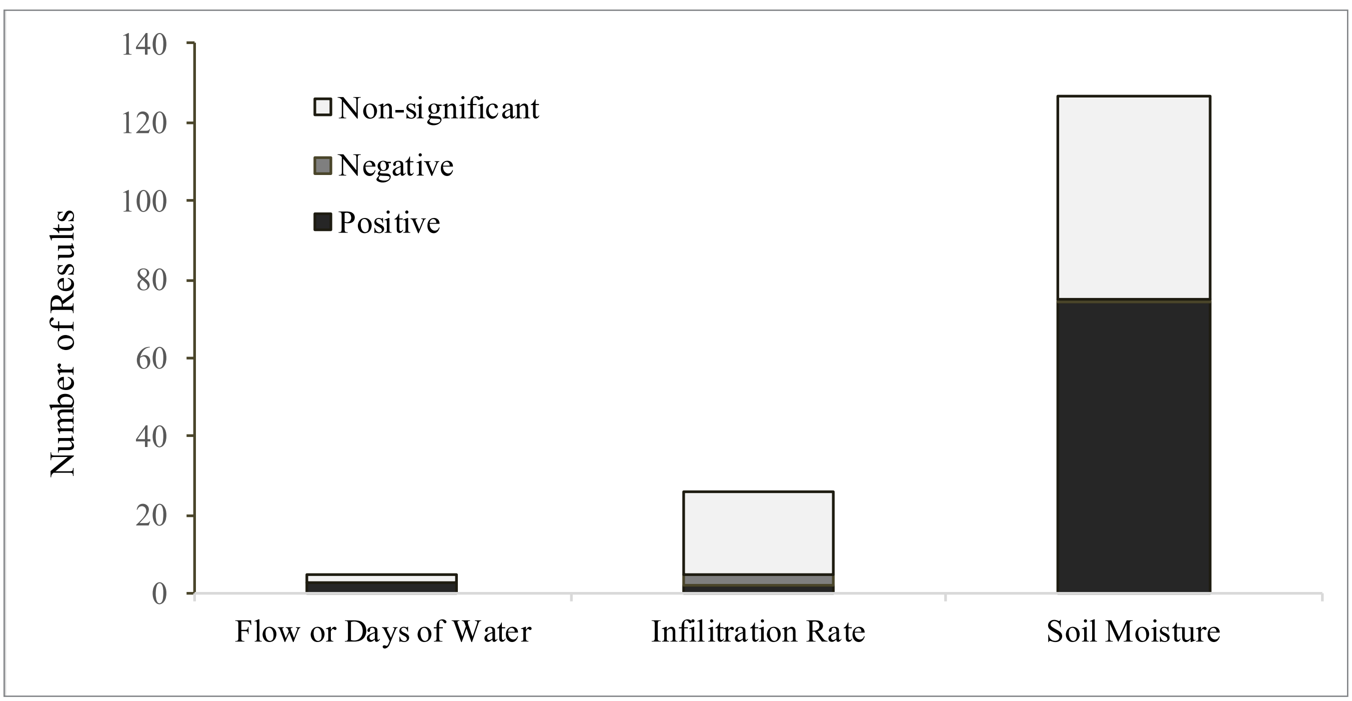


Figure 8. Number of pinyon-juniper treatment study results within flow/days of water, infiltration rate, and soil moisture categories that found positive, negative, or non-significant results. The appendix lists all of the studies that contributed data points to this summary chart, and the response variables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

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)

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 deposi-tion 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 # butter-flies/ transect; Mean no. spe-cies/transect	Butterflies	2
3&4	Knick et al. 2014	Pinyon-juniper Woodland	Burning	Mean no. of detections	Sagebrush obli-gate birds	1
3&4	Kruse 1994	Pinyon-juniper Woodland	Fuelwood harvesting	Counts (total no. captured/ yr.)	Small mammals	1
3&4	Kundaeli and Reyn-olds 1972	Pinyon-juniper Woodland	Uprooting all, thin-ning, uprooting & burning	Pellet counts	Desert cotton-tail	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 abun-dance	Ants	1
3&4	Radke et al. 2008	Pinyon-juniper Woodland	Burning	Mean abun-dance	Lizards and Inverte-brates	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

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/m2	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 sedi-ment 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	Infilitration 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	Infilitration rate, soil mois-ture	(mm * h -1)1, Percent mois-ture	41

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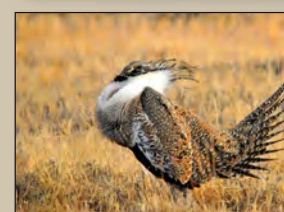
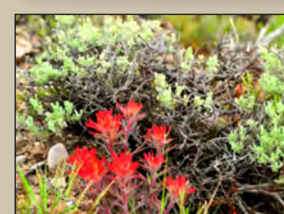
A Conservation Paradox in the Great Basin—Altering Sagebrush Landscapes with Fuel Breaks to Reduce Habitat Loss from Wildfire





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Cover:
Left: Photograph showing mowed fuel break in southwestern Idaho. Photograph by U.S. Geological Survey.
Right (top to bottom): Brewer's sparrow (*Spizella breweri*), Greater sage-grouse (*Centrocercus urophasianus*), Indian paintbrush (*Castilleja angustifolia*) and Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*. Photographs by Tom Koerner, U.S. Fish and Wildlife Service. Wildfire in southwestern Idaho. Photograph by Douglas Shinneman, U.S. Geological Survey, 2010.

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U.S. Department of the Interior
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Conversion Factors

U.S. customary units to International System of Units

Multiply	By	To obtain
	Length	
inch (in.)	2.54	centimeter (cm)
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
mile, nautical (nmi)	1.852	kilometer (km)
yard (yd)	0.9144	meter (m)

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

$$°F=(1.8×°C)+32$$

Datums

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Elevation, as used in this report, refers to distance above the vertical datum.

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A Conservation Paradox in the Great Basin—Altering Sagebrush Landscapes with Fuel Breaks to Reduce Habitat Loss from Wildfire

By Douglas J. Shinneman[1], Cameron L. Aldridge[1], Peter S. Coates[1], Matthew J. Germino[1], David S. Pilliod[1], and Nicole M. Vaillant[2]

Abstract

Interactions between fire and nonnative, annual plant species (that is, “the grass/fire cycle”) represent one of the greatest threats to sagebrush (*Artemisia* spp.) ecosystems and associated wildlife, including the greater sage-grouse (*Centrocercus urophasianus*). In 2015, U.S. Department of the Interior called for a “science-based strategy to reduce the threat of large-scale rangeland fire to habitat for the greater sage-grouse and the sagebrush-steppe ecosystem.” An associated guidance document, the “Integrated Rangeland Fire Management Strategy Actionable Science Plan,” identified fuel breaks as high priority areas for scientific research. Fuel breaks are intended to reduce fire size and frequency, and potentially they can compartmentalize wildfire spatial distribution in a landscape. Fuel breaks are designed to reduce flame length, fireline intensity, and rates of fire spread in order to enhance firefighter access, improve response times, and provide safe and strategic anchor points for wildland fire-fighting activities. To accomplish these objectives, fuel breaks disrupt fuel continuity, reduce fuel accumulation, and (or) increase plants with high moisture content through the removal or modification of vegetation in strategically placed strips or blocks of land.

Fuel breaks are being newly constructed, enhanced, or proposed across large areas of the Great Basin to reduce wildfire risk and to protect remaining sagebrush ecosystems (including greater sage-grouse habitat). These projects are likely to result in thousands of linear miles of fuel breaks that will have direct ecological effects across hundreds of thousands of acres through habitat loss and conversion. These projects may also affect millions of acres indirectly because of edge effects and habitat fragmentation created by networks of fuel breaks. Hence, land managers are often faced with a potentially paradoxical situation: the need to substantially alter sagebrush habitats with fuel breaks to ultimately reduce a greater threat of their destruction from wildfire. However, there is relatively little published science that directly addresses the ability of fuel breaks to influence fire behavior in dryland landscapes or that addresses the potential ecological effects of the construction and maintenance of fuel breaks on sagebrush ecosystems and associated wildlife species.

This report is intended to provide an initial assessment of both the potential effectiveness of fuel breaks and their ecological costs and benefits. To provide this assessment, we examined prior studies on fuel breaks and other scientific evidence to address three crucial questions: (1) How effective are fuel breaks in reducing or slowing the spread of wildfire in arid and semi-arid shrubland

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[1]U.S. Geological Survey.
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ecosystems? (2) How do fuel breaks affect sagebrush plant communities? (3) What are the effects of fuel breaks on the greater sage-grouse, other sagebrush obligates, and sagebrush-associated wildlife species? We also provide an overview of recent federal policies and management directives aimed at protecting remaining sagebrush and greater sage-grouse habitat; describe the fuel conditions, fire behavior, and fire trends in the Great Basin; and suggest how scientific inquiry and management actions can improve our understanding of fuel breaks and their effects in sagebrush landscapes.

Introduction

The Threat of Wildfire to Sagebrush Ecosystems and Wildlife in the Great Basin

Sagebrush (*Artemisia* spp.) ecosystems are highly imperiled throughout North America (Noss and Peters, 1995), largely due to agricultural conversion, energy development, livestock grazing, nonnative species invasions, and altered fire regimes (Knick and others, 2011; Chambers and others, 2016). There has been an estimated 45 percent loss in sagebrush area relative to its historical distribution (Miller and others, 2011), which once likely covered more than 1 million km^2 of the Western United States (Beetle, 1960; McArthur and Plummer, 1978). Roughly one-half of the sagebrush biome is located in the Central and Northern Basin and Range and adjacent Snake River Plain ecoregions, collectively referred to hereinafter as the “Great Basin” and comprising 506,000 km^2 of land dominated by arid and semi-arid shrublands interspersed with isolated mountain ranges (fig. 1). Much of the sagebrush biome in the Great Basin was historically dominated by big sagebrush (*A. tridentata*). Big sagebrush has three primary subspecies: Wyoming big sagebrush (*A. t.* ssp. *wyomingensis*), basin big sagebrush (*A. t.* ssp. *tridentata*), and mountain big sagebrush (*A. tridentata* ssp. *vaseyana*). Other widespread or dominant sagebrush species in the region include low sagebrush (*A. arbuscula*), silver sagebrush (*A. cana*), and black sagebrush (*A. nova*). Although fire is a natural process that plays an important ecological role in the Great Basin, it is now a primary threat to many sagebrush ecosystems in the region (Chambers and Wisdom, 2009; Baker, 2011; Miller and others, 2011), and numerous Federal and State agencies are focused on limiting future losses (Pellant and others, 2004; Wisdom and Chambers, 2009; Havlina and others, 2014; Doherty and others, 2016).

Figure 1. Big sagebrush (*Artemisia tridentata*) landscape, Great Basin, northern Nevada. Photograph by U.S. Geological Survey.

The threat of fire to sagebrush landscapes largely comes from interactions with nonnative ("exotic") annual grasses and forbs, especially cheatgrass (*Bromus tectorum*) (fig. 2), which can promote increased fire frequency and fire spread across extensive areas (Brooks and others, 2004; Balch and others, 2013; Pilliod and others, 2017). Historically, average fire return intervals in sagebrush landscapes likely ranged from a few decades (Miller and Heyerdahl, 2008) to hundreds of years (Baker, 2006; Bukowski and Baker, 2013). Post-fire recovery to mature sagebrush conditions after fire was probably a slow process that typically required several decades or more, similar to post-fire recovery trends observed in contemporary sagebrush stands without substantial invasion by nonnative species (Lesica and others, 2007; Ellsworth and others, 2016; Shinneman and McIlroy, 2016). Warmer and drier sagebrush landscapes, especially those dominated by Wyoming big sagebrush and basin big sagebrush, often have sparse perennial grass cover and low resistance to nonnative species invasion (Chambers, Bradley, and others, 2014; Chambers, Pyke, and others, 2014; Taylor and others, 2014; Brummer and others, 2016). As cheatgrass and other fire-prone annual species invade these ecosystems, they fill interspaces between native perennial plants (Reisner and others, 2013), senesce early in the growing season (Chambers and others, 2016), and provide contiguous swaths of dried, fine fuels that facilitate fire spread and increase ignition rates (Brooks and others, 2004; Pilliod and others, 2017). Following fires, exotic annuals establish more readily and competitively displace native perennials, further intensifying nonnative plant dominance and future fire risk (Chambers and others, 2016). These conditions can lead to a self-perpetuating “grass/fire cycle” (D’Antonio and Vitousek, 1992) characterized by greatly reduced fire-free intervals that promote further dominance and spread of invasive, annual plant species (Brooks and others, 2004; Brooks, 2008) and prevent reestablishment of the native sagebrush community (Laycock, 1991; Brooks and others, 2016) (fig. 3).

Figure 2. Cheatgrass (*Bromus tectorum*). Photograph by U.S. Geological Survey.

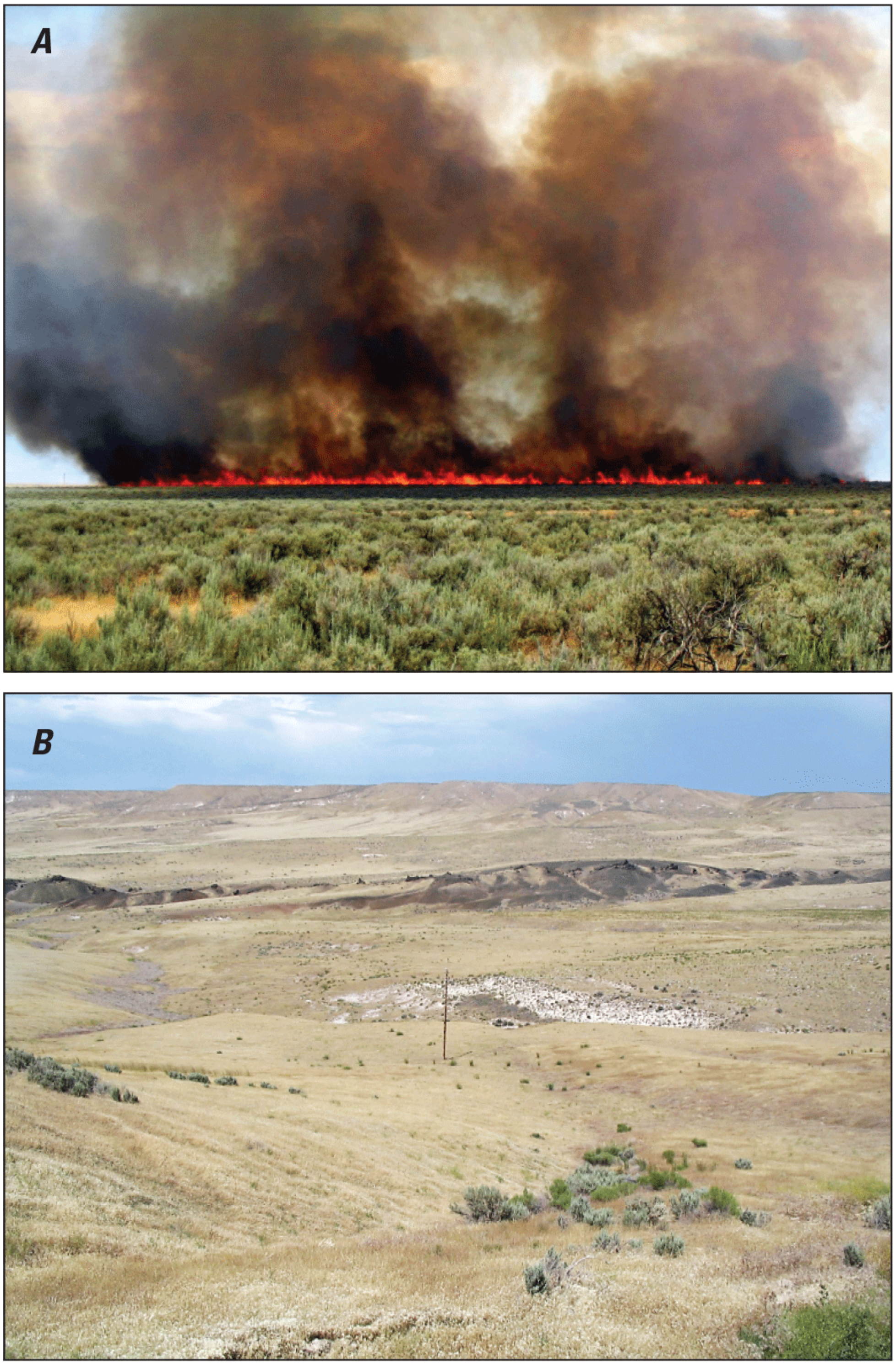

Figure 3. Examples (from southwestern Idaho) of ecological conversion via the grass/fire cycle: (a) fire burning in sagebrush landscape with dried cheatgrass fuels dominant in the understory, and (b) a landscape that formerly supported sagebrush-steppe but, after burning multiple times in recent decades, became dominated by cheatgrass and other fire-prone, annual species. Photographs by U.S. Geological Survey.

Protecting sagebrush ecosystems from the threat of the grass/fire cycle is critical for the myriad species they support. At least 350 plant and animal species depend on sagebrush ecosystems (Wisdom and others, 2005). The greater sage-grouse (*Centrocercus urophasianus*) (fig. 4) is a key sagebrush-obligate and potential umbrella species (Rowland and others, 2006) that is considered at risk throughout its range (Connelly and others, 2004, 2011). The steady loss and fragmentation of sagebrush habitat due to the grass/fire cycle, among other factors, is considered a primary threat to the species' remaining habitat, especially in the Great Basin (Miller and others, 2011; Balch and others, 2013; Brooks and others, 2015; Coates and others, 2016). Indeed, during 2015–17 alone, more than 1.3 million ha (about 3.3 million acres) of greater sage-grouse habitat burned in the U.S., and over two-thirds of that area was within the Great Basin (U.S. Department of the Interior, 2017). Loss of sagebrush habitat from increased wildfire activity has had negative effects on greater sage-grouse populations over the past 30 years, and may reduce the current population size by more than one-half over the next 30 years (Coates and others, 2015, 2016). Effects of exotic plant invasions and altered fire regimes on other sagebrush obligate and associated species are likely similar, but for most species the effects are largely unknown or relatively poorly studied (as reviewed by McAdoo and others, 2004; Litt and Pearson, 2013; Rottler and others, 2015).

Figure 4. Greater sage-grouse (*Centrocercus urophasianus*). Photograph by Tom Koerner, U.S. Fish and Wildlife Service.

In response to wildfire threats to sagebrush-dependent wildlife and other rangeland resources in the Great Basin, land management agencies rely heavily on a variety of pre-fire fuel treatments, fire suppression, and post-fire rehabilitation and restoration strategies aimed at increasing resistance to invasion by annual grasses and resilience from future wildfire disturbances. Implementing networks of linear fuel breaks has become a particularly strategic pre-fire management tool intended to enhance fire suppression effectiveness and limit ecological damage from unwanted wildfire (Green, 1977; Ager and others, 2013; Maestas, Pellant, and others, 2016; U.S. Department of the Interior, 2016a). A "fuel break" is defined by the National Wildfire Coordinating Group (2018) as “a natural or manmade change in fuel characteristics which affects fire behavior so that fires burning into them can be more readily controlled.” Land management agencies are increasingly planning and utilizing linear fuel break networks across much of the Great Basin to conserve sagebrush and greater sage-grouse habitat (Moriarty and others, 2016). However, despite the potential for fuel breaks to help slow the loss of sagebrush caused by fire, relatively little scientific information is available to assess either their effectiveness (that is, to control wildfire) or their ecological effects (that is, on plant and wildlife communities), especially in arid and semi-arid landscapes. In the only other review of fuel breaks for sagebrush ecosystems that has been compiled, the authors state, “Fuel break effectiveness continues to be a subject of much debate yet relatively little research has been conducted evaluating their role in constraining wildfire size and frequency” (Maestas, Pellant, and others, 2016, p. 4). Similarly, there is insufficient research regarding the effects of fuel breaks on rangeland ecosystems in general and the effects on wildlife populations specifically.

Objectives and Approach

This report is intended to serve as an initial assessment of fuel breaks in sagebrush landscapes of the Great Basin, including their potential effectiveness in altering fire behavior and reducing area burned (by facilitating fire suppression and containment), their ecological costs and benefits, and the need for further science. To accomplish these objectives, we examined prior studies, agency databases, and other scientific evidence for three crucial questions:

1. How effective are fuel breaks in reducing or slowing the spread of wildfire in arid and semi-arid shrubland ecosystems?
2. How do fuel breaks affect sagebrush plant communities?
3. What are the effects of fuel breaks on the greater sage-grouse, other sagebrush obligates, and sagebrush-associated wildlife species?

Before addressing these questions, we provide an overview of recent federal policies and management directives aimed at protecting remaining sagebrush and greater sage-grouse habitat, and discuss how the potential use of fuel breaks to help achieve these objectives also underscores the need for better scientific understanding of their potential effects. We then describe the fuel conditions, fire behavior, and fire trends in the Great Basin to set an operational context for the different types and designs of linear fuel breaks commonly used in the region. In light of the information provided, we close this report by summarizing what is known and not known about fuel breaks, and suggest how scientific inquiry and management actions can improve our understanding of fuel breaks and their effects in sagebrush landscapes.

The primary geographic focus of this review encompasses the sagebrush-dominated landscapes of the Great Basin (fig. 5), with an ecological focus on greater sage-grouse habitat and the sagebrush steppe and shrubland communities that are typically dominated by big sagebrush or low sagebrush. However, many of our findings are applicable to sagebrush ecosystems throughout the western half of the greater sage-grouse range, particularly where sagebrush community composition and climate conditions are similar to that of the Great Basin. These findings also may be pertinent to other shrubland ecosystems (for example, salt-desert shrublands, mountain shrublands) that are typically adjacent to, or intermixed with, sagebrush.

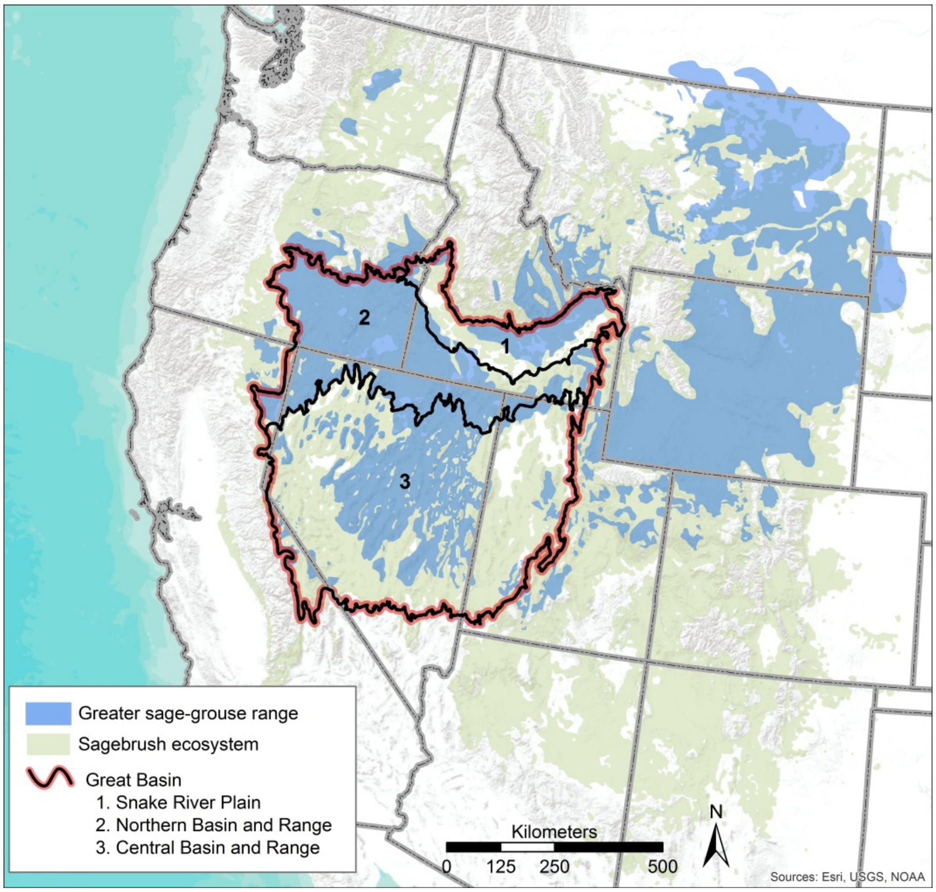

Figure 5. Location of the sagebrush ecosystem and distribution of greater sage-grouse in the Western United States. The Great Basin consists of the Central Basin and Range, Northern Basin and Range, and Snake River Plain ecoregions (Level III; U.S. Environmental Protection Agency, 2013). Sagebrush ecosystem data from U.S. Geological Survey (2018b); greater sage-grouse distribution data from U.S. Geological Survey (2018a).

To accomplish these objectives, we reviewed the scientific literature directly related to fuel breaks, but also considered research pertaining to the effects of other types of fuel treatments on sagebrush communities, as well as from other anthropogenic disturbances (especially linear landscape features, such as roads). Assessments of fuel break effects also were considered within an operational understanding of sagebrush ecosystem dynamics, including plant community function, disturbance ecology, fire behavior, nonnative species invasions, and wildlife population dynamics and habitat needs. We considered articles in peer-reviewed science publications, but also examined “gray” literature (for example, graduate theses and agency reports). Our objective did not include analytical review approaches (for example, a “meta-analysis”), largely due to the current paucity of data and quantitative research regarding the effects of linear fuel breaks in sagebrush ecosystems. Additionally, we assessed the utility of relevant agency databases that contain information on fuel treatment effects and effectiveness (for example, the Fuel Treatment Effectiveness Monitoring database) to help guide strategic fuel break plans moving forward.

Fuel Breaks to Protect Greater Sage-Grouse Habitat—Policy, Management, and Science Directives

In light of recent decisions regarding the legal status of the greater sage-grouse, rangeland fire suppression and sagebrush conservation have become dominant land management priorities in the Great Basin, and fuel breaks have been identified as an important strategy to help achieve these goals. The greater sage-grouse was first considered for listing under the U.S. Endangered Species Act (ESA) by the U.S. Fish and Wildlife Service (USFWS) in 2005. Listing for the greater sage-grouse was determined not to be warranted, but the official decision document recognized fire as significant threat, especially in the western part of the species’ range (U.S. Fish and Wildlife Service, 2005). A subsequent 2010 decision by the USFWS concluded that listing under the ESA was warranted but precluded by higher priorities, and it again emphasized the increasing role of fire in threatening greater sage-grouse habitat (U.S. Fish and Wildlife Service, 2010). Under court-order in 2015, the USFWS determined that the greater sage-grouse did not warrant protection under the ESA and would be removed from the candidate list (U.S. Fish and Wildlife Service, 2015). The agency cited the effectiveness of ongoing conservation partnerships that were benefitting greater sage-grouse over 90 percent of its 7-million-hectare range.

Despite this legal outcome, land management agencies were tasked with implementing policies that would conserve and benefit sagebrush ecosystems, in large part to ensure continued protection of greater sage-grouse habitat. In January 2015, U.S. Department of the Interior Secretarial Order 3336 called for a “science-based strategy to reduce the threat of large-scale rangeland fire to habitat for the greater sage-grouse and the sagebrush-steppe ecosystem” (U.S. Department of the Interior, 2015a, Section 6a). Two companion reports were subsequently published to reinforce and facilitate the Secretarial Order: “The Integrated Rangeland Fire Management Strategy” was intended to specifically identify effective actions to prevent and suppress rangeland fire, and to restore fire-affected sagebrush landscapes, while the “Actionable Science Plan” (hereinafter, IRFMS-ASP) identified key science needs and research priorities that would promote more efficient and effective use of specifically identified management strategies (U.S. Department of the Interior, 2015b, 2016a). Fuel breaks were identified as a key strategy in these documents.

Although the IRFMS-ASP suggested that the design of fuel breaks should use existing spatial information to help protect sagebrush focal areas and greater sage-grouse priority habitats (U.S. Department of the Interior, 2016a), it also pointed out that little is known about the effects of fuel breaks on greater sage-grouse populations, habitat use, and movement across the landscape. Moreover, the IRFMS-ASP outlined other potential negative effects of fuel breaks that are poorly studied, including spread of invasive plants, effects on other sagebrush-obligate species, increased habitat fragmentation, expanded access for off-highway vehicles, and increased potential for human-caused ignitions. Given such knowledge deficiencies, two of the eight “Fire Science Needs” that are described and prioritized in the IRFMS-ASP identify fuel breaks as high priority areas for scientific research. Specifically, Fire Science Need #5 stresses the need to determine how to minimize the potential deleterious ecological consequences of fuel breaks, similar fuel treatments, and resulting landscape patterns that are ostensibly designed to benefit greater sage-grouse and their habitats by reducing wildfire spread (U.S. Department of the Interior, 2016a, p. 21). Fire Science Need #8 seeks to determine the characteristics of fuel breaks that are effective in preventing fire spread or intensity, including through “…synthesis of the literature, critical evaluation of techniques and plant materials used in fire breaks (species, structure, placement, and native versus nonnative species), and economic tradeoffs” (U.S. Department of the Interior, 2016a, p. 26). The IRFMS-ASP additionally recommended various complementary steps designed to encourage assessment, research, and monitoring to determine the effectiveness of different types of fuel breaks in changing fire behavior, their potential ecological effects, and prospects for long-term maintenance.

Fire Regimes, Patterns, and Trends in the Great Basin

Recent studies have demonstrated that fire regimes across large portions of the Western United States have changed over the past several decades, with longer fire seasons, more area burned, and shorter fire return intervals on average over time (for example, Westerling and others, 2006; Littell and others, 2009; Dennison and others, 2014). Although fire has always been an integral natural process in most ecosystems and fire regimes are dynamic over time, anthropogenic factors such as changing climate, land use effects (for example, grazing, fire suppression), and nonnative species invasions are likely increasing fire activity in some ecosystems and pushing them beyond their historical ranges of variability (for example, Westerling and others, 2006; Abatzoglou and Kolden, 2013; Higuera and others, 2015). Of the major ecosystem types in the Western United States, sagebrush ecosystems have among the most clearly altered fire regimes due to these human-induced factors (Keane and others, 2008; Abatzoglou and Kolden, 2011; Balch and others, 2013; Bukowski and Baker, 2013).

A recent report by the U.S. Geological Survey (Brooks and others, 2015) documented that about 8.4 million ha burned in the western portion of the greater sage-grouse range (which is largely located in the Great Basin) over a recent 30-year period (1984–2013). Roughly 88 percent of that burned area was in sagebrush vegetation types. During that same 30-year period, about 1.2 million ha burned two or more times, and the vast majority (about 85 percent) of this "recurrent fire" area was also in sagebrush vegetation types, including cheatgrass invaded areas. Moreover, the annual area burned by fires in the western portion of the greater sage-grouse range has likely increased over the 30-year period, in large part driven by trends in the Snake River Plain, where recurrent fire is contributing to average fire return intervals of less than 7.5 years in some areas. The report also demonstrated that fire sizes have been increasing over the 30-year period in portions of the Great Basin (see also Balch and others, 2013), with “mega-fires” greater than 40,000 ha not uncommon, and with some individual fires exceeding 200,000 ha (fig. 6). Finally, the primary fire season in the Great Basin, which typically starts in May and often extends into September (as defined by the start dates of large fires), is the longest in the Snake River Plain, and there is statistical evidence that it has lengthened over the past 30 years in the southern portion of the Great Basin (Brooks and others, 2015). Although the fire area patterns and trends outlined in Brooks and others (2015) were derived from the best available data on large fires (>405 ha, which comprise about 95 percent of total area burned), small fires (comprising about 5 percent of the area burned) are not included, and some large fires are potentially missing from the earlier portion of the 30-year record (Short, 2015).

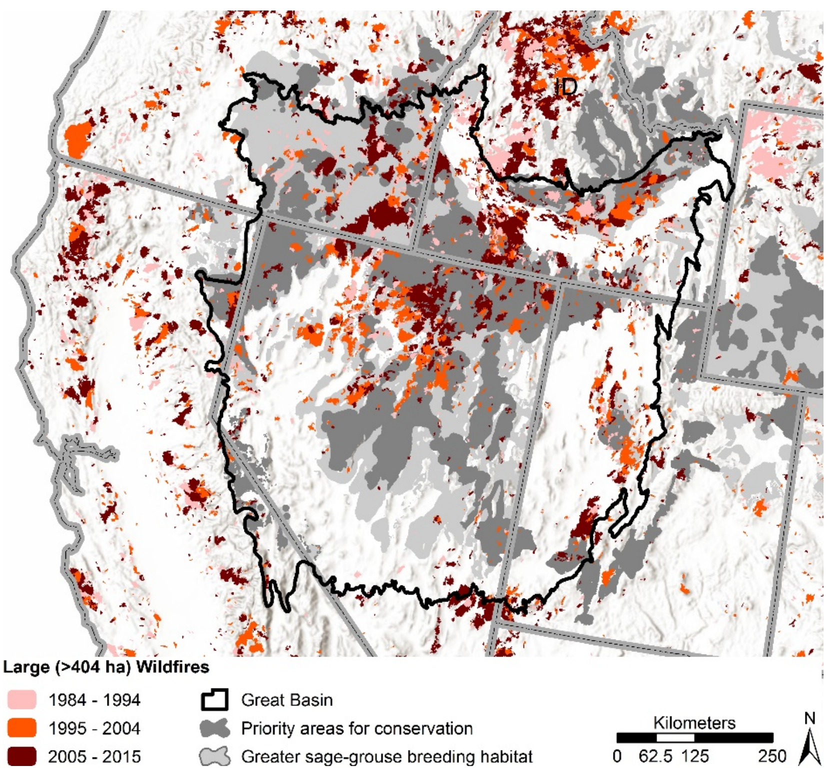

Figure 6. Large fires in and around the Great Basin, 1984–2015. Data from Monitoring Trends in Burn Severity (2018).

The number and extent of wildfires in the Great Basin (or any region) are influenced by ignition sources, climate and fire weather, fuel availability, and topography (DeBano and others, 1998). Historically, these factors contributed to infrequent occurrences of large fires in sagebrush landscapes of the Great Basin (Bukowski and Baker, 2013). However, fire trends of the past several decades have been influenced by human-altered fire regimes, largely due to interactions among ignition sources, invasive plants, and climate variability. Within the Great Basin, lightning accounted for 58 percent of all fires and 84 percent of area burned between 1992 and 2015 (Short, 2017) (fig. 7). However, human-caused ignitions in the U.S. generally are increasing the number of wildfires, the area burned, and the fire season length (Balch and others, 2017). In the Great Basin, the area burned by both lightning and human-caused fires is enhanced by the widespread availability of herbaceous fine fuels, especially in areas with substantial cover of nonnative annual grasses that dry early in the fire season and accumulate as litter over several years (Balch and others, 2013; Pilliod and others, 2017). In a recent remote-sensing mapping effort in the northern Great Basin, about 82 percent of the area in lower-elevation (<2,000 m) rangelands had some cheatgrass cover, about 33 percent had greater than 10 percent cover, and some areas (especially in the Snake River Plain) were at or near 100 percent cover (Boyte and Wylie, 2016) (fig. 8). Cheatgrass-dominated areas have been shown to be approximately two to four times more likely to burn compared to other rangeland community types (Balch and others, 2013).

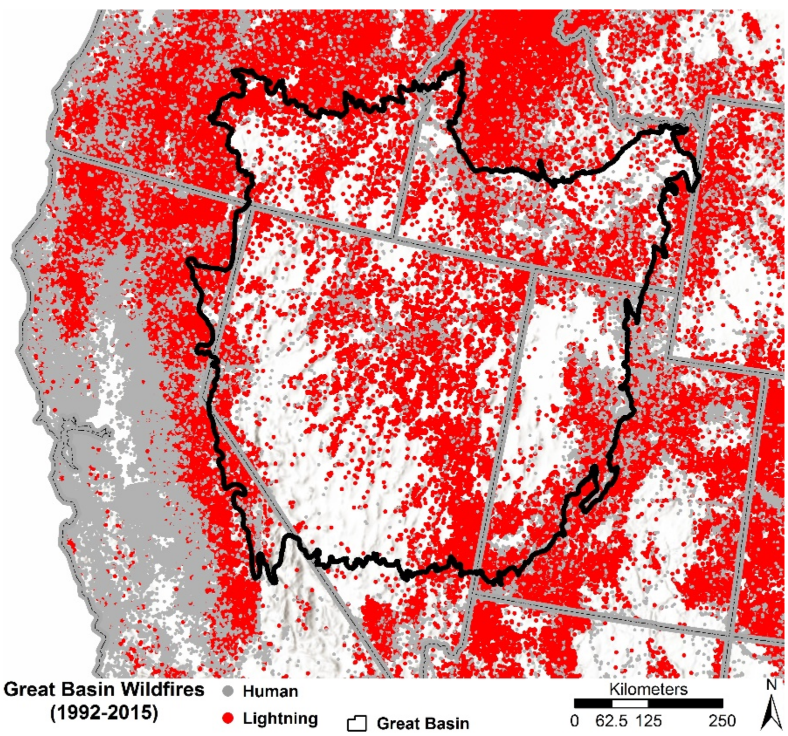

Figure 7. Wildfire ignitions by source (human and lightning) in and around the Great Basin, 1992–2015. Data from Short (2017).

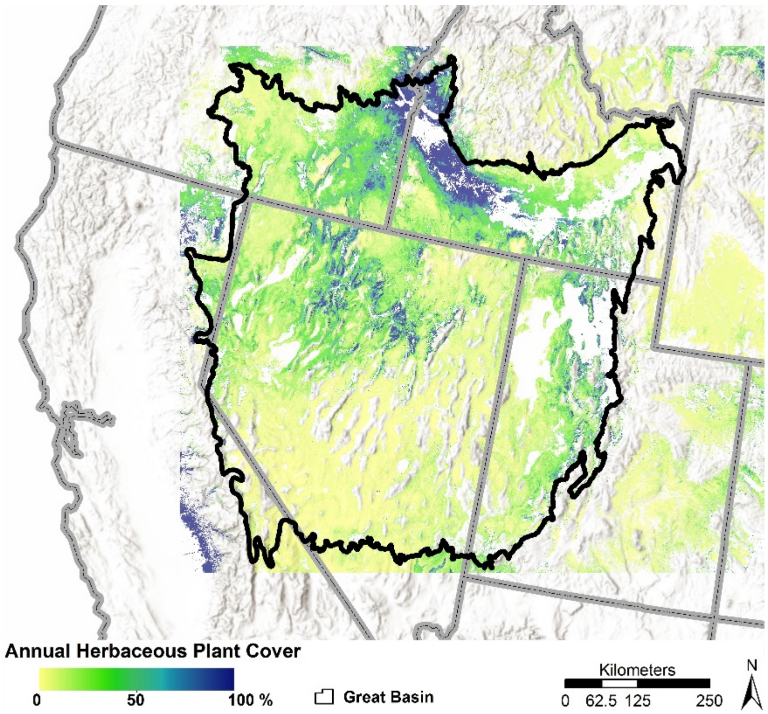

Figure 8. Near-real-time cover (June 19, 2017) of annual herbaceous grasses in the Great Basin. Data from Boyte and Wylie (2017).

Climate and fire weather influence fuel and fire dynamics but act at different spatial and temporal scales. Fire weather includes precipitation, wind speed, temperature, and relative humidity at temporal- and spatial-scales relevant to behavior of individual fires (Schroeder and Buck, 1970; Brown, 1982; Wright, 2013). Climate is a long-term phenomenon of annual cycles of precipitation and temperature that drive plant growth and phenology, fuel accumulation, desiccation of biomass (vegetation and litter), and lightning patterns that influence fire patterns and trends across broad spatial- and temporal-scales (Westerling and others, 2003; Minnich, 2006; Pilliod and others, 2017). Though climate conditions vary along elevational and latitudinal gradients, most of the Great Basin is typified by cold winters and warm-to-hot summers that receive relatively little precipitation. Because summer conditions are typically hot and dry enough to support fire in the Great Basin, there is no strong connection between contemporaneous moisture deficits alone, fuel drying, and wildfire activity in the region (Westerling and others, 2003; Davies and Nafus, 2013). Rather, fuel loading is typically the limiting factor that drives fire activity in Great Basin shrublands, and studies have demonstrated that higher precipitation during the winter and early growing season results in greater amounts (cover, biomass) of grasses and forbs, including nonnative species such as cheatgrass (Pilliod and others, 2017). Moreover, these fuel loads increase fire risk over several years because of nonnative forb and cheatgrass litter accumulation (Pilliod and others, 2017), resulting in the well-established phenomenon of increased fire activity (more fires and area burned) 1–3 years following above-normal moisture (Billings, 1994; Knapp, 1998; Westerling and others, 2003; Littell and others, 2009; Abatzoglou and Kolden, 2013; Balch and others, 2013). Modeled projections suggest that future climate could enhance both fuel production and fire-weather conditions, potentially making these ecosystems even more fire-prone in coming decades (Stavros and others, 2014; Barbero and others, 2015; Liu and Wimberly, 2016).

Fuel Break Objectives, Types, and Design Considerations

Within the fire environment (fig. 9), fire weather and topography cannot be altered, but fuels can be modified. A *fuel treatment* is a type of pre-suppression activity intended to manipulate or reduce fuels and modify fire behavior in an effort to mitigate potential negative wildfire impacts. The types and spatial pattern of fuel treatments can vary depending on the fire regime, fire management objective, and values at risk (Ager and others, 2013). A "fuel break" is a type of fuel treatment that involves the removal or modification of vegetation in strategically placed strips or blocks of land, specifically to disrupt fuel continuity and reduce fuel loads and accumulation. Fuel breaks target removal or control of plants with low-moisture or high volatile oil content that are more likely to carry fire, increase fire residence time, promote longer flame lengths, or encourage spotting (Weatherspoon and Skinner, 1996; Agee and others, 2000; Maestas, Pellant, and others, 2016). The strategic spatial configurations of fuel breaks are intended to enhance firefighter access, improve response times, provide safe and strategic anchor points for wildland firefighting activities (for example, back-burning), and compartmentalize wildfires to constrain their growth (Green, 1977; Maestes, Pellant, and others, 2016). A key point among these objectives is that fuel breaks are designed to facilitate fire suppression operations, and are not intended to stop fire activity unaided (though they occasionally do). Indeed, after interviewing 15 experienced fire managers in the northern Great Basin, Moriarty and others (2016) found wide agreement that the purpose of fuel breaks is to "…allow firefighters to actively engage in fire suppression in a safe, strategic manner without committing exhaustive resources to control or contain the spread of wildfire." Limited systematic analysis of fuel break effectiveness in forest and chaparral ecosystems also suggests that the main way in which fuel breaks effectively help to constrain fire size is by facilitating fire suppression activities (for example, Syphard and others, 2011a, 2011b).

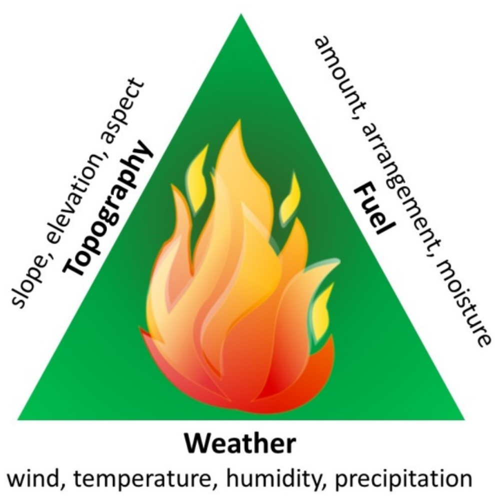

Figure 9. Fire environment triangle. Once combustion is sustained, the fire environment (weather, topography, and fuels), influences the growth and behavior of a fire. Within the fire environment the three factors are interrelated and vary with both space and time.

The three main types of linear fuel breaks used in the Great Basin include green strips, brown strip*s*, and mowed linear fuel breaks, and these are often employed along with other treatments, including modifying existing roadbeds, herbicide use, or targeted grazing. Linear fuel breaks are often dispersed among other broad-scale treatments designed to disrupt fire spread and help facilitate fire containment, including use of prescribed fire or thinning and removal of piñon (*Pinus* spp.) and juniper (*Juniperus* spp.) trees. In the following section, we describe the three primary types of linear fuel breaks used in the Great Basin. We later discusss potential ecological effects and limitations of each fuel break type in more detail, as we address fuel break effectiveness and effects on plant and animal communities.

Green Strips

The goal of constructing a green strip is to replace more flammable and contiguous plant communities (particularly those dominated by exotic annual grasses, such as cheatgrass) with perennial plants that retain moisture later into the growing season, often by using plants that grow as widely spaced, low-statured individuals that result in large, bare interspaces (fig. 10). Green strips are typically constructed in widths of 30–90 m along both sides of a road, although they can be wider and may result in a combined width of 180 m or more when including the road (Pellant, 1990, 1994, 2000; St. John and Ogle, 2009). In green strips, vegetation is typically first removed or altered with a plow, harrow, or chain, and often in combination with application of a broadly effective herbicide (for example, glyphosate) to control existing vegetation, with additional herbicide treatments (for example, Imazapic) to reduce invasive annual grasses (Maestas, Pellant, and others, 2016). New species are then sown into the prepared strips, with ideal seeded species having relatively deep roots, forming persistent stands that provide some competitive pressure against exotic annual invasion, and having relatively inexpensive seeds that germinate reliably. Not many species have these criteria, and they include the nonnative perennial crested wheatgrass (*Agropyron cristatum*, also *A. desertortum* and their varieties and hybrids) and the subshrub/semi-evergreen forage kochia (*Bassia prostrata*), as well as a few others (Monsen, 1994; Pellant, 1994; St. John and Ogle, 2009) (fig. 10). These vegetation type conversions are designed to result in reduced fuel loads, discontinuous fuels, and less-flammable vegetation that can slow rates of spread and wildfire intensity (Davison and Smith, 1997). Replacing cheatgrass or other annual species with more fire resistant vegetation breaks the continuity of fuels across the landscape, reducing the rate of spread and aiding in suppression success (Pellant, 1994). Early in the fire season, the increased fuel moisture of the vegetation alone can delay or limit burning (Monsen, 1994). Additionally, increasing the proportion of plants with higher moisture content during peak fire season can reduce the potential for ignition and rate of spread (Pellant, 1994).

Figure 10. Great Basin green strips. (a) Forage kochia (*Bassia prostrata*) in southwestern Idaho; (b) forage kochia; and (c) crested wheatgrass (*Agropyron cristatum*). Photographs by U.S. Geological Survey.

If established under ideal conditions, green strips may require relatively little maintenance, especially if planted species are drought resistant, tolerant of grazing, able to survive fire, or have competitive advantages over more fire-prone species. However, in many cases, the ability of a green strip to alter fire behavior generally diminishes over time without regular maintenance, and the treated areas may be prone to litter accumulation or invasion by annual species (Monsen, 1994; Gray and Muir, 2013; Meastas and others, 2016a). Thus, the effectiveness of a green strip to alter fire behavior can reduce over time without maintenance, and they typically need to be mowed or grazed to reduce the buildup of fine fuel between the desired plants (Monsen, 1994; Maestas, Pellant, and others, 2016).

Brown Strips

The two-fold objective of a brown strip (fig. 11) is firstly to limit fire starts within the fuel break and secondly to provide a place for firefighters to engage in suppression activities. A brown strip is typically installed along major thoroughfares (for example, paved highways) using a harrow or plow to completely remove vegetation (that is, all fuels) down to bare mineral soil, typically in widths of 3–6 m (and sometimes wider). Brown strips are the most simplistic of the linear fuel breaks in regards to potential fire behavior, because they are devoid of vegetation and thus cannot burn. Brown strips function as anchor points for direct-attack fire suppression or as a line for indirect attack tactics (for example, burnout operations) ahead of the approaching fire front. However, because of the narrow width that brown strips are typically constructed, they are breached under higher intensity fire events where flame length or spotting exceed the width (Green, 1977; Wilson, 1988; Pellant, 2000). Moreover, the effectiveness of a brown strip is short-lived (for example, single fire season) without continued maintenance (for example, re-disking or herbicides), as they are prone to weedy plant invasion (Pellant, 1990).

Figure 11. Brown strip that stopped a fire that started along an adjacent highway. Photograph by Bureau of Land Management.

Mowed Linear Fuel Breaks

The primary goal of creating a mowed fuel break is not to reduce the total fuel load but rather to compact and limit the vertical extent of the fuel bed, which results in lower flame lengths and reduced rates of spread. Effectively, mowing redistributes fuel loadings by reducing vegetation to 15–30 cm in height and by leaving the cut plant material on site (Maestas, Pellant, and others, 2016) (fig. 12). Mowed fuel breaks are typically at least 30–90 m wide and constructed along both sides of a road (they may be substantially wider, depending on fuel conditions and fire suppression needs). Mowed fuel breaks are the preferred method of treatment within patches of intact sagebrush because they are relatively easy to implement and, if wide enough, can help to disrupt large, wind-driven fires and limit wildfire spread (Maestas, Pellant, and others, 2016). However, reducing the canopy cover can increase herbaceous plants in the short-term, necessitating further intervention (Davies and others, 2011; 2012a), and treated areas require regular mowing or targeted grazing to maintain the desired fuel height (Schmelzer and others, 2014).

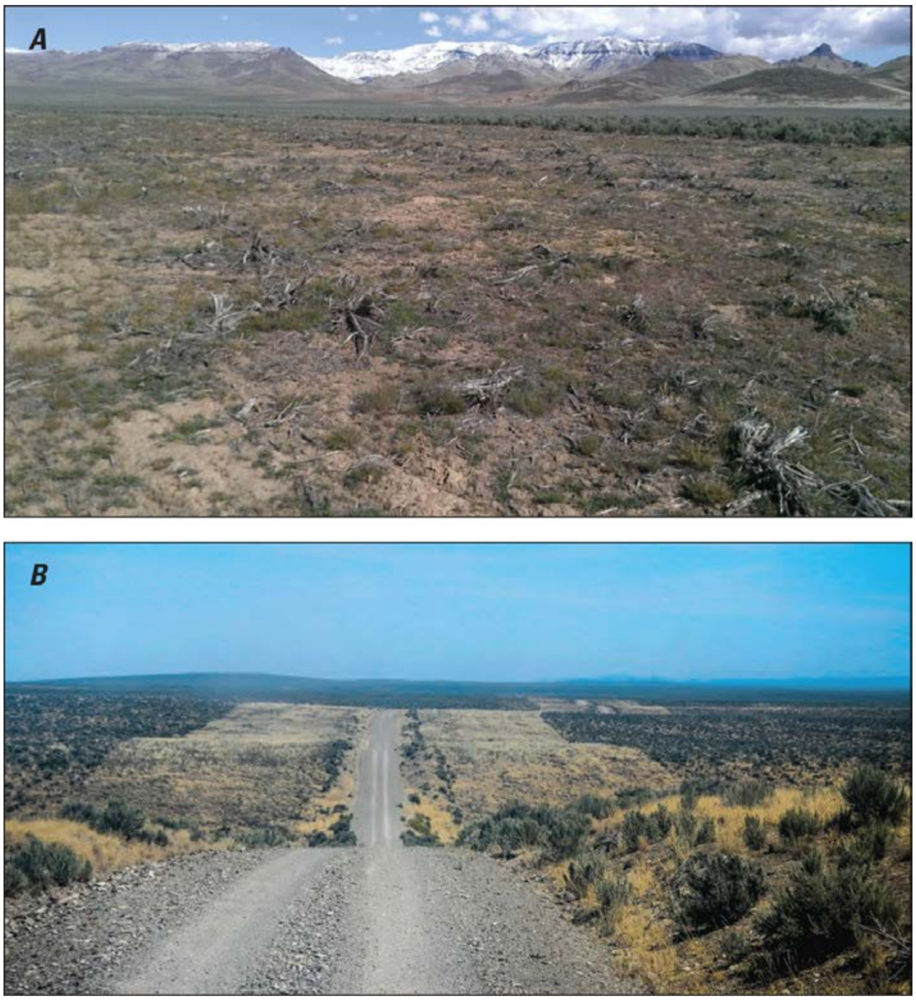

Figure 12. Example of (a) recent mowing in northern Nevada and (b) mowed linear fuel breaks along both sides of a gravel road in southwestern Idaho. Photographs by (a) Bureau of Land Management and (*b*) U.S. Geological Survey.

Other Fuel Break Treatments

One additional category of fuel break treatment worth mentioning that is relatively new, but now in limited use, is targeted grazing. In this approach, fuel reduction is accomplished by prescribing livestock utilization to specific levels and grass heights (Diamond and others, 2009; Schmelzer and others, 2014). Appreciable logistic challenges lie in the contractual constraints of grazing permits, economic costs or benefits to permittees, practical challenges of concentrating livestock into linear features, and other issues. Targeted grazing has occurred primarily on degraded sites with low resistance to invasion and existing high cover of exotic annual grass–that is, sites and landscapes that may have little ecological value to lose (Strand and others, 2014). Because of its rather limited and novel usage, the effects of targeted grazing are not explored further in this report.

Fuel Break Spatial Design and Strategic Placement Considerations

In addition to directly altering fuel characteristics within the break itself, the ability of fuel breaks to limit fire ignition and spread also depends on spatial design, particularly the width of individual breaks and the configuration of a fuel break system (that is, a network of individual fuel breaks) on the landscape. Depending on the type, individual fuel breaks are often constructed in widths ranging from just a few meters to over 100 m along roadsides, typically along roadways that provide reliable firefighter access (fig. 13). However, linear fuel breaks are also sometimes constructed along other human features on the landscape that may require protection or that can facilitate access (for example, power-lines, fences, and housing developments). Although wider fuel breaks are generally considered better for effectively altering fire behavior under more extreme conditions, it is not practical or realistic to create excessively wide fuel breaks that extend over many linear kilometers. As a result, different analytical approaches have been used to suggest efficient widths and shapes of individual fuel breaks. For instance, Finney (2001) used predicted fire shape and rates of spread to determine that the width and length of a rectangular fuel treatment unit could be considered optimized when the resulting shape caused the portion of the fire burning through the unit and the portion burning around it to bypass the unit at the same rate. Wilson (1988) used grassland fire experiments to determine that the flame length of an approaching fire can be used as a rough approximation for the necessary width of a brown strip to stop the spread of fire via flame contact. Using a more operational approach, federal agencies have recently promoted fuel break widths of about 90 m on both sides of a road, using both flame length considerations and the need for establishing enough fire-free space to provide adequate safety zones for firefighting activities (U.S. Department of the Interior, 2016b) (fig. 14).

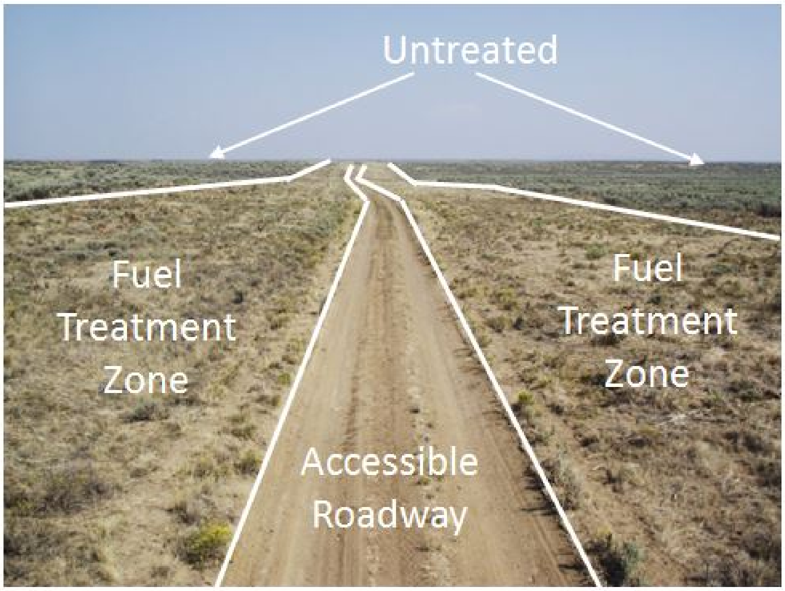

Figure 13. Conventional fuel break design along an accessible roadway. Photograph by Bureau of Land Management.

Figure 14. Example of how fuel type impacts flame lengths, with approximately 30-foot-long flames in sagebrush stands versus approximately 7-foot-long flames along mowed roadside. Figure by U.S. Department of the Interior (2016b).

Other spatial and strategic placement considerations include positioning fuel breaks on the landscape to most effectively influence patterns of fire ignition, probability, intensity and spread (based on prevailing wind direction). Simulation studies in both North American forests and Mediterranean woodland-shrublands have shown that optimizing the spatial pattern of fuel treatments is more effective at limiting fire spread than random or non-strategic placement (Finney, 2001; Duguy and others, 2007; Parisien and others, 2007; Schmidt and others, 2008; Bar-Massada and others, 2011; Oliveira and others, 2016). There has been relatively little spatially explicit fire behavior or fire-connectivity modeling done to help plan more effective fuel break networks in non-forest landscapes. Gray and Dickson (2016) used circuit theory simulations on rangelands in the Kaibab Plateau in Arizona to test the effectiveness of green strips to reduce overall fire spread between patches of cheatgrass within a landscape of piñon-juniper and sagebrush. Their models suggested that strategic placement of green strips at locales where fire is most likely to spread to surrounding areas, representing just 1 percent of the study area landscape, could decrease overall area burned. Recently, federal agencies and their partners have also been using landscape simulation models to help design fuel treatments more effectively across large landscapes in the Great Basin, to demonstrate the utility of modeling to improve the targeting of fuels reduction projects, and to minimize potential impacts on greater sage-grouse habitat (Rideout and others, 2017). However, we are aware of only a few such modeling studies for the Great Basin (Welch and others, 2015; Opperman and others, 2016) and, to our knowledge, these have not undergone external, scientific peer-review. Experimentally testing various fuel break designs that are supported by modeling analysis is a logical next step to ensure their efficacy. In the only such study we are aware of in the Great Basin, the Bureau of Land Management (G. Dustin, written commun., April 20, 2017) tested a spatially strategic fuel break configuration as suggested by Finney (2001), and results indicated that rate of fire spread and flame length could be effectively reduced by using parallel, overlapping disc lines in a cheatgrass dominated landscape in northern Utah. However, such designs are not likely to be practical in intact sagebrush habitat, due to wildlife habitat fragmentation concerns (as discussed later).

When it is not feasible to complete spatially explicit fire behavior simulations for local planning, nationally produced maps of fire and fuels data may still provide useful information for the placement of fuel breaks within landscapes. Here, we highlight a few examples of national-scale datasets, some with fairly comprehensive (gridded) spatial coverage. Fire data from the Monitoring and Trends in Burning Severity program (Monitoring and Trends in Burning Severity, 2018) uses a consistent methodology to provide fire perimeter and severity information for all fires 405 ha (1,000 acres) or larger that have burned since 1984 (Eidenshink and others, 2007). This comprehensive and spatially explicit dataset can help to target fuel break locations; for instance, by identifying fire spatial patterns and temporal trends that indicate changing fire extent or frequency. The smallest fires–especially those less than 40 ha (about 100 acres)–are often only reported as point locations (that is, not fire boundaries) in other available fire datasets, and are less reliable due to missing or inaccurate information and redundancy errors (Brooks and others, 2015). However, there are now relatively comprehensive datasets that contain fires of all sizes and that attempt to reconcile problematic small fire records (Short, 2017; Welty and others, 2017). Small fires are more numerous than large fires and, although they account for only about 5 percent of area burned over time (Eidenshink and others, 2007), may be particularly relevant for locating areas with high rates of ignition and for assessing fuel break effectiveness (for example, to determine if fuel breaks influenced fire size).

The LANDFIRE program (LANDFIRE, 2018) is a source of gridded geospatial information available for the entire United States. It can be used to assess potential fire threats based on disturbance history, vegetation type, and fuel characteristics (Rollins, 2009; Ryan and Opperman, 2013). The LANDFIRE program provides fuel model grids for predicting fire behavior (for example, spread and intensity) and has recently offered dynamic fine fuel measurements for the Great Basin and Southwest based on current fire season herbaceous cover (currently available as provisional data [LANDFIRE, 2017]). These dynamic fuels data are meant to better reflect the seasonal and inter-annual variability in fine fuel loadings that are common in desert and semi-desert ecosystems (Gray and others, 2014; Pilliod and others, 2017).

Other highly pertinent products include mapped analyses of wildfire likelihood, intensity, and risk (using comprehensive fire and fuels data for the conterminous United States). These analyses are intended to inform evaluations of wildfire risk or prioritization of fuels management needs across large landscapes. Short and others (2016) developed mapped estimates of annual likelihood of a fire burning (that is, "burn probability," fig. 15a) and associated intensity (under current landscape conditions and fire management practices, fig. 15b) by simulating tens of thousands of hypothetical contemporary fire seasons (Finney and others, 2011). Recently, Chambers and others (2017) combined the fire probability maps developed by Short and others (2016) with greater sage-grouse breeding habitat probability and resilience/resistance maps to indicate where sagebrush and greater sage-grouse habitats are at highest risk from fire across the sagebrush biome (fig. 16). More specifically for Great Basin rangelands, Pilliod and others (2017a) developed a model of wildfire risk on the basis of established relationships between seasonal precipitation data and wildfire characteristics in Major Land Resource Areas. Finally, there are myriad other fire-relevant datasets that contain dynamic (fuel moisture and fire danger rating), static (fuel models), and historical (ignition location/source) information of varying geographic coverage, resolution, and utility (U.S. Forest Service, 2018) that could also aid fuel break design, but assessing each of these is beyond the scope of this review.

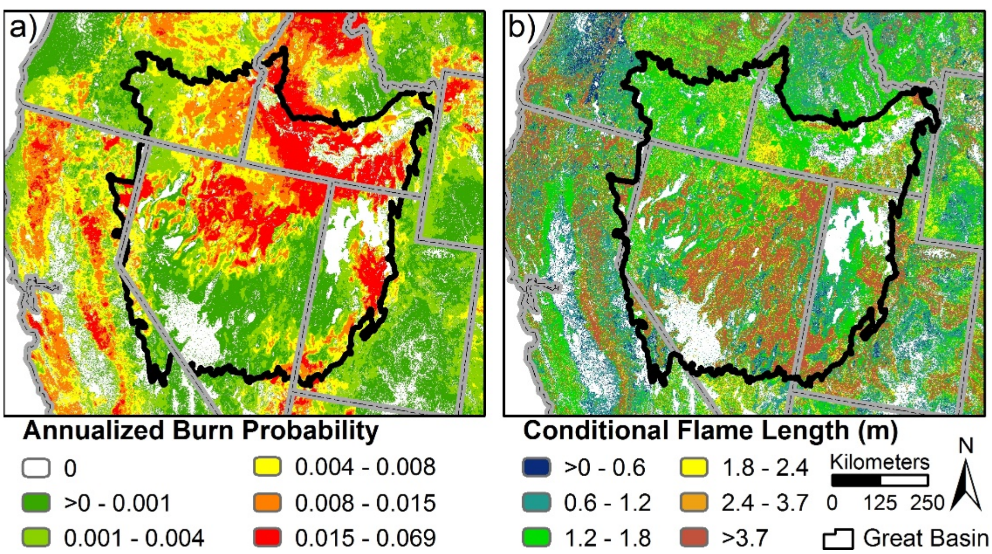

Figure 15. Nationally available maps of (a) simulated fire probability and (b) intensity for the Great Basin. Data from Short and others (2016).

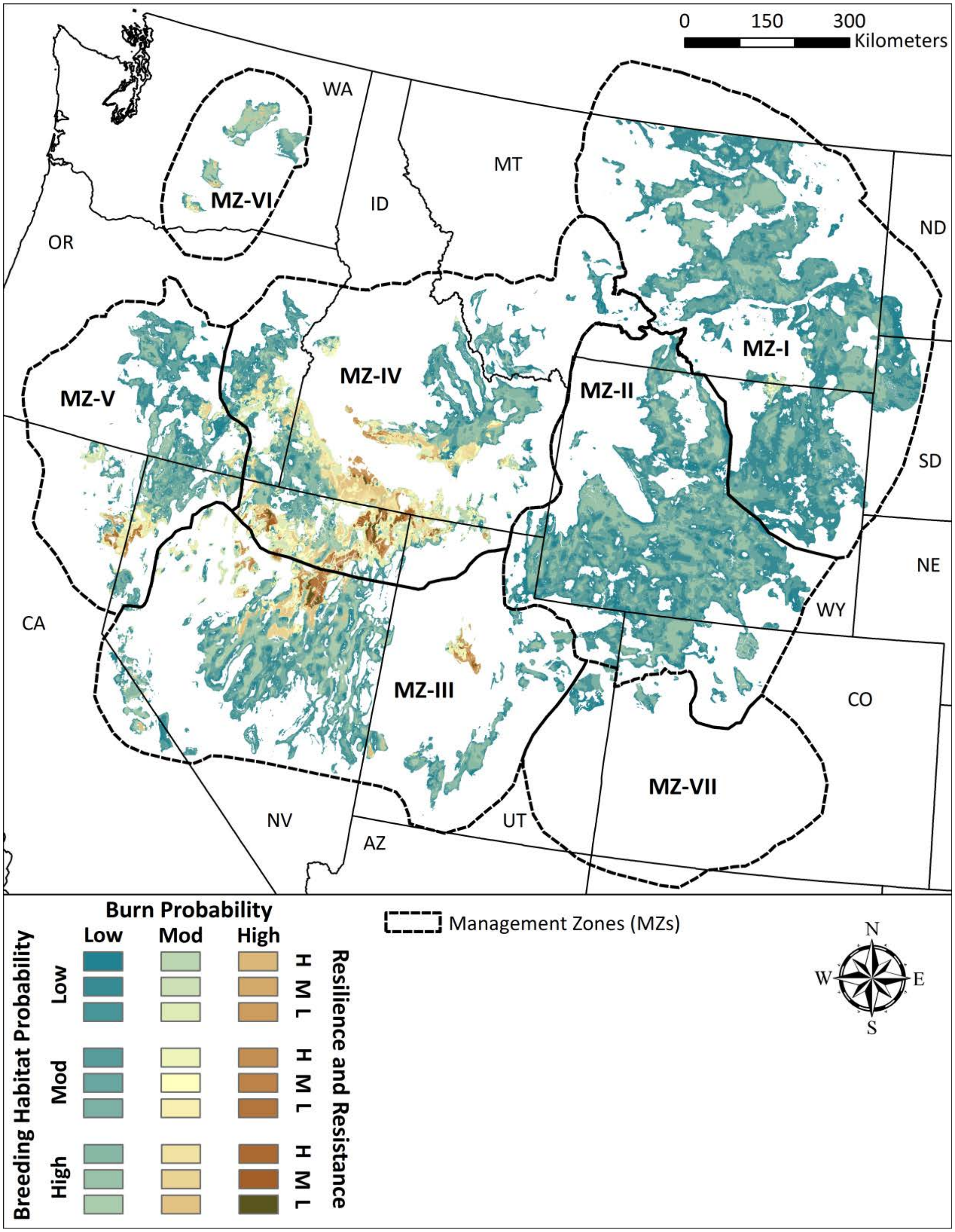

Figure 16. Wildland fire risk map for greater sage-grouse, in which Chambers and others (2017, fig. A10.1) depicted 27 different combinations of burn probability and threats to sagebrush ecosystems, using greater sage-grouse breeding habitat probability (Doherty and others, 2016), resilience and resistance classifications (Maestas, Campbell, and others, 2016), and large wildland fire probability (Short and others, 2016). Effectively, wildfire risk increases as colors on the map transition from blue-green to brown, reflecting combinations of increasingly greater fire probability, lower resistance and resilience, and more valuable greater sage-grouse habitat.

Using Wildfire Simulation to Model Fuel Treatment Effects on Fire Behavior

Modeling is another important tool that can be used to design and maintain fuel breaks, by projecting their effectiveness in altering fire behavior and assessing their ability to provide utility and safety for firefighting activities. Fire modeling systems (that is, "fire simulators") are important tools in fuels management because they can be used to predict the effect of fuel treatments on potential fire behavior, including flame length, rate of spread, fireline intensity, fire growth, and burn patterns that affect the ability to safely use suppression activities (Miller and Landres, 2004; Varner and Keyes, 2009). Fire modeling systems exist for both stand- (that is, typically <40 ha) and landscape-level assessments (>40 ha).

Commonly used stand-level (or point-based) fire systems are non-spatial models that give a “snapshot” of potential fire behavior for a given fire environment (that is, with uniform fuel, topography, and fire weather conditions in time and space), as specified by the modeler. Thus, model users can alter inputs (for example, by changing fuel types or wind speed) among simulations to compare fire behavior under different environments (as in fig. 17). Recently, such models have been used by land management agencies in the Great Basin to assess the likely effectiveness of treated fuel conditions in proposed fuel break projects for Great Basin rangelands (U.S. Department of the Interior, 2016b). Behave Plus (Andrews, 2014) is the most frequently used non-spatial fire behavior system among fire and fuel management professionals (Miller and Landres, 2004).

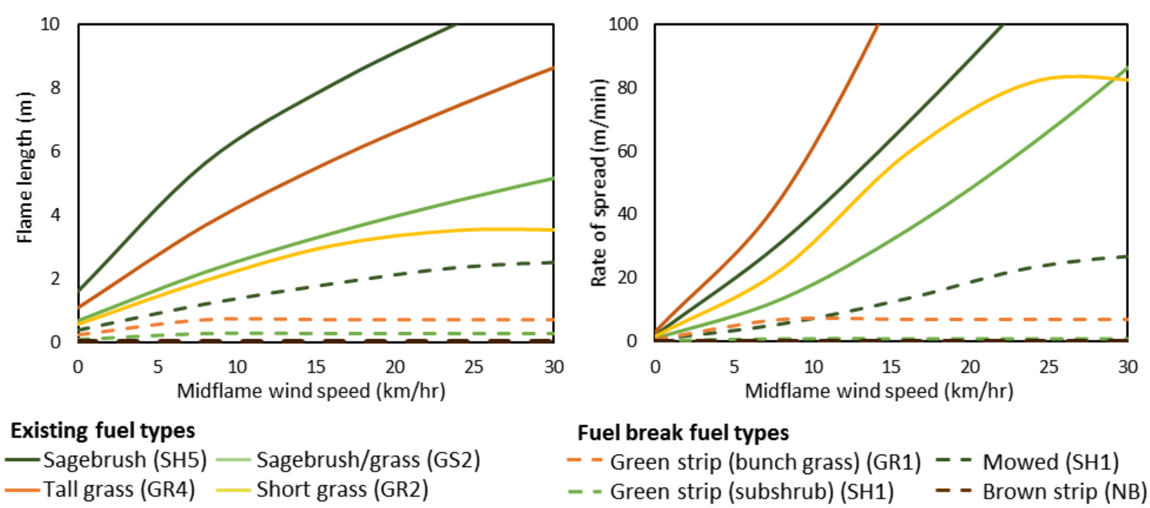

Figure 17. Predicted flame length and rate of spread for common existing and treated fuel types within the Great Basin using BehavePlus (Andrews, 2014), a stand-scale, fire-behavior model. Fuel models are from Scott and Burgan (2005), where GR is "grass," GS is "grass shrub," SH is "shrub," and NB is "non-burnable." Fuel moisture levels were different for mowed and green strip ("subshrub") modeling with the same fuel model. These simulations indicate that fairly rapid fire movement could still occur across mowed fuel breaks, although more resolute modeling is needed (see appendix 1 for model parameters).

In contrast to stand-level models, landscape-level wildfire systems simulate the spread of fire across a landscape under variable fire environments. In the fuel management context, these models are used to determine the effectiveness of landscape fuel treatments in reducing fire size (that is, reduce rate of spread), changing intensity (that is, flame length), and predicting fire likelihood (that is, burn probability) for known and random ignitions. To assess fire behavior for pre-determined ignition points, FARSITE is among the most widely used, typically to simulate the growth of a single wildfire over time under heterogeneous fuels and terrain, as well as under dynamic fire-weather conditions (Finney, 2004). To project potential fire behavior of multiple fires across landscape scales, FlamMap (Finney, 2006) and its derivatives (for example, the large fire simulator, FSim; Finney and others, 2011) are among the most commonly used wildfire simulation systems for management and planning purposes (for example, determining where to apply fuel treatments). In FlamMap, for example, fire-weather is held constant for any particular model run and the spread of one to many fires is simulated across spatially heterogeneous landscapes and fuel conditions.

However, it is worth pointing out that these landscape-scale fire behavior models have largely been developed and used for forested landscapes, and they have rarely been used in the sagebrush ecosystems of the Great Basin or other dryland landscapes (and mostly for non-research purposes). Moreover, inputs for landscape-scale fire behavior models may not adequately capture the influence of cheatgrass and other nonnative annuals that drive seasonal and interannual variability of fine-fuel loadings and continuity that greatly influence ignition rates, fire probability, and rates of spread (compare Gray and Dickson [2016]); and LANDFIRE dynamic fine fuel measurements [LANDFIRE, 2017]). Moreover, whether using spatial or non-spatial fire behavior models, the inputs required to represent fuel conditions are generally derived from standard *fuel models* (Anderson, 1982; Scott and Burgan, 2005) that specify surface fuel attributes (for example, fuel loading) among different fuel types. These standard fuel models are derived from a priori fuel type classifications that may not adequately capture key fuel attributes found in the Great Basin, particularly in fuel break treatments (for example, forage kochia monocultures or recently mowed sagebrush), and custom fuel models may need to be developed to obtain more accurate fire behavior predictions (in the sense of Keane [2015]).

Question 1. How Effective Are Fuel Breaks in Reducing or Slowing the Spread of Wildfire in Arid and Semi-Arid Shrubland Ecosystems?

Historically, most empirical evidence for the effectiveness of fuel breaks has been largely anecdotal, based on previous wildland firefighting experience or occasional agency reports for specific projects. Despite the extensive use of fuel breaks in sagebrush landscapes, especially since the 1990s, the IRFMS-ASP points out that “no specific research within the sagebrush ecosystem has been conducted to evaluate their effectiveness” (U.S. Department of the Interior, 2016a, p. 25). Moreover, the IRFMS-ASP also suggests that fires often occur 10 or more years after a fuel break is constructed, when effectiveness may have reduced if lack of maintenance resulted in conversion to vegetation types that more readily carry fire. Moreover, fire managers acknowledge that, under extreme fire weather conditions, fuel breaks are unlikely to adequately reduce fireline intensity, flame length, or rate of spread (Moriarty and others, 2016). These factors make it challenging to assess the relative effectiveness of properly maintained fuel breaks under different fire environments.

Examples of effectiveness of fuel breaks in the Great Basin have been reported in various agency publications to highlight the success of fuel treatments. For example, the combination of wildfire suppression efforts and a 60-m wide green strip stopped a wildfire along 10 of 11 km of the contact zone (the breach was along a rocky ridge surrounded by pockets of sagebrush) near Grasmere, Idaho in 1988 (Pellant, 1994). Similarly, green strips adjacent to a highway contributed to limiting a wildfire in 1990 near Mountain Home, Idaho to 6 ha relative to the 10-year average of about 725 ha for that location (Pellant, 1994). Forage kochia green strips in Utah and Nevada reduced flame lengths and even stopped fires completely in places (Harrison and others, 2002). However, over-reliance on hand-picked examples of success underscores the difficulty in accurately assessing fuel break effectiveness, as such cases represent anecdotal reporting with a lack proper study controls. Even studies that have used simulation modeling to assess fuel break influence on fire dynamics tend to lack empirical validation of results.

Despite these individual reports and studies, consistent record-keeping and monitoring of fuel treatment effectiveness has not historically been a priority for fire and land management agencies. Until recently, there was no central repository to store information specifically regarding the efficacy of fuel breaks. This has been partially remedied by the Fuel Treatment Effectiveness Monitoring (FTEM) program. The FTEM was initiated in 2006 with the goal of demonstrating the utility of hazardous fuels reductions by verifying that fuel treatments encountered by wildfire worked as intended. The FTEM database has become the primary source of information for qualitatively assessing the effectiveness of fuel treatments to alter fire behavior. Initially, the FTEM included only voluntary reporting of treatment effects on U.S. Forest Service lands, but reporting became mandatory for the U.S. Forest Service in 2011 and for the Department of the Interior in 2012. For each treatment burned in a wildfire, two "yes/no" questions are required in the FTEM: (1) "Did the fire behavior change as a result of the treatment?" and (2) "Did the treatment contribute to the control of the fire?" Using FTEM data, Moriarty and others (2016) found that of the 58,000 ha of fuel treatments reported by the Bureau of Land Management in Oregon, Idaho, and Nevada, 97 percent of the treatment area was considered to have altered fire behavior, and 95 percent aided in the control of the fire. Although these findings are encouraging, a "yes" response in the FTEM database is relatively subjective. For example, what criteria constitute a significant change in fire behavior? Moreover, although the FTEM database provides fields for supplying important additional information, many records lack adequate descriptions of the fuel treatment, how fire behavior was changed, or the specific fire-environment. Generally, more recent FTEM records contain more of this critical information than older records, and they are often cross-linked to other databases containing fuel treatment details. However, based on our assessment of recent fire and known fuel break locations extracted from the Land Treatment Digital Library (Pilliod and Welty, 2013), it is not clear that all fire interactions with fuel breaks are entered into the FTEM. We found that between 2012 and 2016 there were 114 fires that intersected (that is, burned through) mapped linear fuel breaks in the Great Basin, and many of these incidents do not match locational information provided in the FTEM (see example landscape in fig. 18). Thus, we do not know how the behavior of these fires may have been affected by fuel breaks.

Additionally, agencies within the U.S. Department of the Interior lack a single comprehensive database for storing fuel treatment locations, their spatial extent, or conditions over time (that is, by monitoring species composition, cover, biomass). Thus, it is uncertain how many fuel breaks currently exist in the Great Basin, let alone their spatial configurations or fuel loadings. This further confounds our ability to systematically determine where and when fuel breaks work. Recently developed agency-wide databases (for example, the Land Treatment Digital Library [LTDL]) are intended to remedy these previous record-keeping deficiencies, but they are still not entirely inclusive, in large part because older records are incomplete or missing, and many record entries lack critical information. Based on our assessment of available records of linearly shaped treatments that contained information indicating a fuels reduction focus (for example, “green strip,” “fuel break”), we estimate that there are at least 10,000 linear kilometers of fuel breaks already in the Great Basin, and about 130,000 ha contained within them (table 1). Small (generally <1 km) linear treatments, non-linear fuel treatments, and linear features with no associated treatment information were not included in table 1, and it is likely some fuel breaks remain unmapped. Thus, undoubtedly, additional fuel breaks exist for which we lack records entirely or that have not yet been properly entered into agency databases.

In short, anecdotal evidence, sporadic project monitoring, and limited record-keeping indicate that fuel treatments do accomplish their intended goals under certain conditions. However, a history of incomplete and insufficient record-keeping has resulted in a lack of systematically collected data on fuel treatments in general, and fuel breaks specifically, that would allow us to readily and objectively analyze how often and under what conditions linear fuel breaks are effective. We simply lack spatially and temporally comprehensive datasets on fuel breaks, including locations, treatment types, maintenance history, fire environments (for example, fire-weather conditions, fuel loadings), and firefighting response (for example, whether or not used for suppression activities, and in what manner) to accomplish such an analysis at this time. However, as agency-wide databases continue to be compiled and improved, such analyses may become prudent, at least for portions of the Great Basin with consistent record keeping.

Table 1. Known and likely linear fuel break distance and area in the Great Basin by Bureau of Land Management (BLM) district office.

[Values are approximate, based on incomplete mapping and database entries, and probably underestimate actual totals. **Data sources:** Land Treatment Digital Library: Pilliod and Welty (2013), and (2) The Vegetation Treatment Area database (Bureau of Land Management, 2010). Data accessed: October 13, 2017. This is an initial assessment that will eventually be reconciled with other agency databases, especially the National Fire Plan Operations and Reporting System (NFPORS). See appendix 2 for methods.]

BLM District Office	State	Hectares	Kilometers
Battle Mountain	Nevada	19,803	567
Boise	Idaho	3,518	2,431
Burns	Oregon	7,996	777
Carson City	Nevada	1,639	65
Color Country	Utah	18,656	250
Elko	Nevada	18,355	1,124
Ely	Nevada	6,106	318
Idaho Falls	Idaho	8,657	702
Lakeview	Oregon	1,814	774
Northern California	California/ Nevada	7	2
Prineville	Oregon	867	59
Twin Falls	Idaho	11,190	962
Vale	Oregon	17,467	649
West	Utah	7,511	570
Winnemucca	Nevada	10,356	1,273
Totals		**133,942**	**10,523**

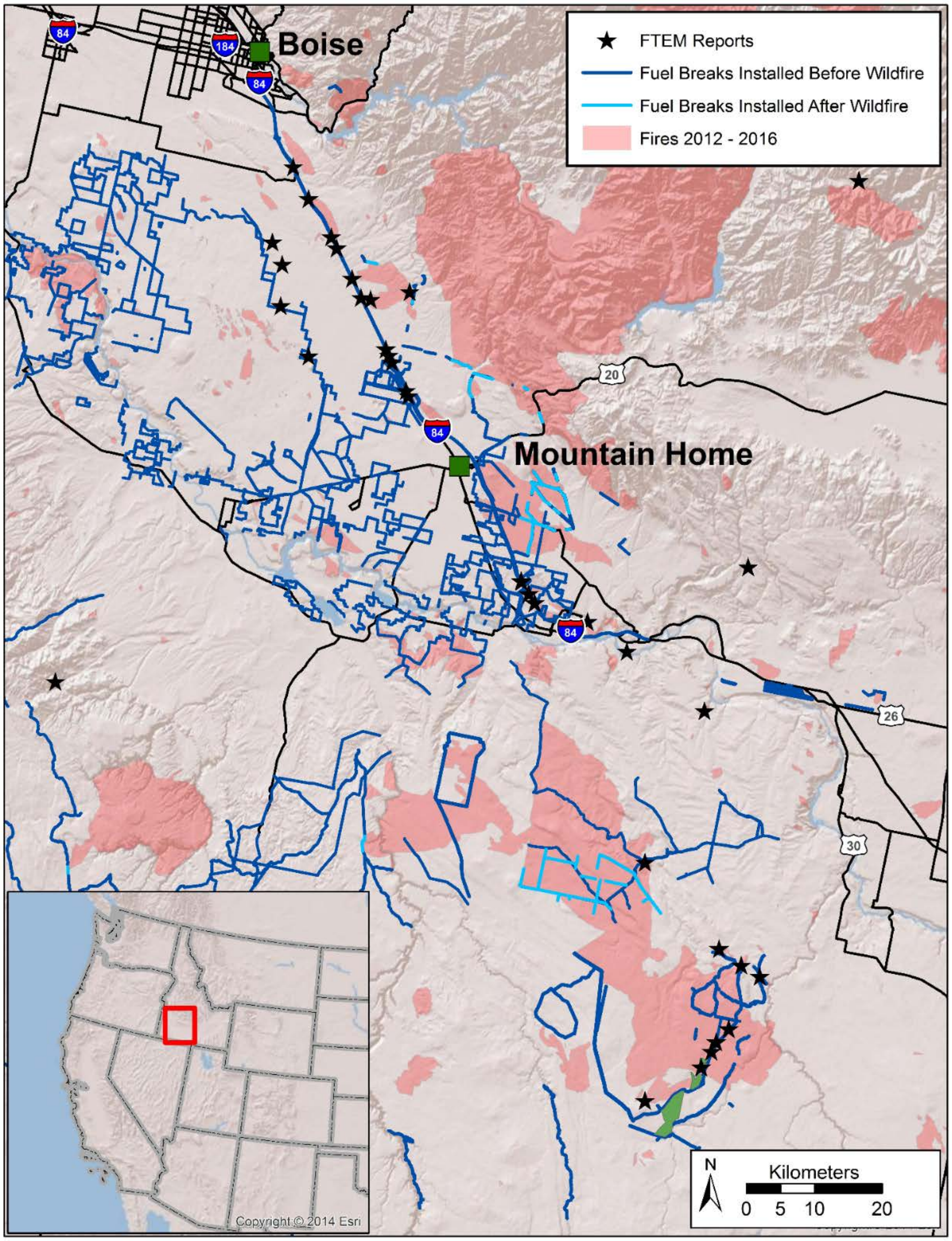

Figure 18. Reports of fire interaction with fuel treatments recorded in the Fuels Treatment Effectiveness and Monitoring (FTEM) database. All existing FTEM records are shown relative to fires that burned from 2012–16 (after reporting to the FTEM became mandatory for the Bureau of Land Management) for a portion of the northern Great Basin. In this landscape, it is apparent that many FTEM records are for smaller fires that started along roads, while some larger fires clearly intersected existing fuel breaks but were not reported in the FTEM. The multiple FTEM records in and around the large fire at the bottom of the map include some fuel breaks, as well as other fuel treatment types, but they lacked comments to describe how they contributed to suppression objectives or changed fire behavior. Fire data compiled by U.S. Geological Survey (2017) (see table 1 for fuel treatment data sources).

Question 2. How Do Fuel Breaks Affect Sagebrush Plant Communities?

Different types of fuel breaks can affect plant communities directly through modification or conversion of the existing plant community, and indirectly through the spread of invasive species and changes in soil conditions. Here, we characterize potential plant and soil responses to fuel break treatments across plant community types and their climates over short, intermediate, and long time periods. We also discuss implications for different management and maintenance scenarios with an emphasis on fuel breaks in intact shrublands versus herbaceous annual grasslands, and across landscapes with varying resistance to cheatgrass and resilience to disturbance. We focus on dominant plant communities and do not cover sensitive species because we assume fuel breaks would be diverted around them. The limited empirical evidence for plant community responses to green stripping, brown stripping, and mowing treatments is then reviewed. A summary of key effects of fuel breaks on plant communities are in table 2.

Table 2. Summary of potential fuel break effects on plant communities.

Fuel break type	Maintained?	Vegetation condition	Wildfire potential	Risks
Green strip	Yes	Widely spaced and more fire resistant species; typically nonnatives introduced through seeding	Shortened period for combustibility; in some cases less contiguous fuels (for example, forage kochia [*Bassia prostrata*])	Stand failure (maladaptation), risks of emigration or invasive spread of seeded species into surrounding landscape
	No	Potential attrition of desirable species and gain of undesirable species (for example, annual grasses, invasive forbs)	Fine fuels accumulate, enhancing ignition and fire spread	Fuel break becomes invaded (or re-invaded), affecting surrounding landscape
Brown strip	Yes	Bare soil	Does not burn	Herbicide risks, soil erosion
	No	High potential for annual species invasion	Increased ignition and rates of spread	Increased fire hazard, spread of exotic species
Mowing	Yes	Reduced height (15–30 cm)	Reduced flame height	“Bushout” could increase fuel continuity, potential for exotic invasion
	No	Height is regained	Flame height reduction lost; potential for enhanced ignition and fire spread	Initial condition regained, potential for exotic invasion

We note that while the effects of fuel breaks have been evaluated in Mediterranean-like climates of California (for example, Syphard and others, 2011a, 2011b), there are limits to the transferability of the information into sagebrush steppe. Merriam and others (2006) and Potts and Stephens (2009) reported increases in bare soil and exotic plant abundances (up to 40 percent increases or more) on fuel breaks applied across a diverse array of habitats in California, a number of which have similar exotic species and winter-wet conditions that have favored conversion of native shrublands to nonnative annual grass communities in the Great Basin. However, there are limitations to transferring information from habitats such as chaparral to sagebrush ecosystems, due to different life forms and growing season patterns (for example, there is generally less grass cover in chaparral). Similarly, there is substantial literature on cutting, masticating, and prescribed burning of piñon-juniper, oak woodland, and chaparral habitats; however, these systems contain much greater biomass in standing woody species than most sagebrush sites. The removal of larger and more dominant trees or shrubs would be expected to result in greater resource release (for example, that could be exploited by exotic annuals) than in sagebrush ecosystems.

Plant-Community Trajectories and Their Relationship to Soil Resources within Fuel Breaks

Plant community responses within fuel break treatments are determined by both biophysical setting (for example, elevation, topography, precipitation, temperature, and soils) and type of fuel break installed. Fuel breaks in the Great Basin are applied in many different plant communities over a wide range of elevations that receive different amounts of precipitation annually, from less than 120 mm at lower elevations to greater than 500 mm at upper elevations. Precipitation combined with soil properties strongly influences vegetation communities, as well as what can grow successfully in a fuel break and how a fuel break might need to be maintained. For example, the warm and dry salt desert at the lowest elevations support shrub species with unique adaptations to salt, drought, or toxic minerals, including shadscale (*Atriplex confertifolia*), greasewood (*Sarcobatus* spp.), and winterfat (*Krascheninnikovia lanata*). Middle elevations support sagebrush steppe, a cold desert perennial grassland characterized by the presence of shrubs, particularly sagebrush. Mountain shrub communities, characterized by big sagebrush (*A. t. subsp. vaseyana*), snowberry (*Symphoricarpos* spp.), and curlleaf mountain mahogany (*Cercocarpus lediofolius*), dominate at higher elevations where precipitation is less limiting but still inadequate to support coniferous forests. These plant communities all are structurally heterogeneous in their intact condition, co-dominated by woody (shrub) or herbaceous perennials interspersed with bare soil “canopy gaps” that provide discontinuity in wildfire fuels. Invasion of these canopy interspaces by nonnative annual grasses can lead to nearly complete replacement of perennials by a homogeneous canopy of annuals, that results in a high-continuity, fine-textured fuel bed that senesces with low water content for about 80–90 percent of each year (Brooks and others, 2004; Germino and others, 2016).

Although there are few reports of sagebrush or other rangeland vegetation response to fuel break treatments, the treatments used to create fuel breaks are similar to treatments that are commonly applied as part of rehabilitation or restoration actions to large tracts of land in the Great Basin (Pyke and others, 2014). Thus, the general paradigms and concepts currently used to predict vegetation change (see section, “General Concepts for Plant Community Responses”) are usually also applicable to fuel breaks, although scale and edge effects are anticipated to be relatively important landscape factors for fuel breaks due to their extensive linear configurations. However, one of the most basic concepts represented by classic plant succession models (that is, an orderly transition from early, to mid, and late series of species assemblages) may have only marginal or sometimes no utility in explaining or predicting the plant communities of interest after a treatment. For example, the species that successfully establish after a fuel break treatment are likely to be the species that will persist into the mid- to long-term, unless invasions by nonnative plant species, grazing, or subsequent fire cause further change. Notable exceptions are when ruderal species become established and undergo apparent seral replacement, such as replacement of Russian thistle (*Salsola kali*) by exotic annual mustards (for example, *Sisymbrium altissimum*) and then cheatgrass (Piemeisel, 1951). Also, native species such as Sandberg bluegrass (*Poa secunda*) or 6-weeks fescue (*Vulpia* spp.) can have a clear early-successional, colonizing role compared to other native species.

Below, we extend state-and-transition and resistance and resilience concepts (as described in section, “General Concepts for Plant Community Responses”) to the three types of fuel breaks, focusing on the relationship of the potential plant community outcomes of the treatments to wildfire risk, site conditions, and soil stability. Traditional state-and-transition models (STMs) do not account for the surrounding landscape of a subject site; however, edge effects and species immigration and emigration (that is, invasion of fuel breaks, or invasion of species seeded onto fuel breaks) are primary concerns for fuel breaks. By design, fuel breaks have a high perimeter-to-area ratio, and movement of species from or to the surrounding landscape is of primary concern. Generally, the habitat fragmentation and edge-effect impacts of fuel breaks on the plant communities surrounding them will usually relate to spread of seeded/planted or volunteer (invasive) species from fuel breaks into the surrounding landscape (Gray and Muir, 2013). For instance, any resulting increase of exotic annuals on the fuel break “strip” would likely increase the potential for invasion of the surrounding landscape.

General Concepts for Plant Community Responses

State-and-transition theory and the resistance and resilience paradigm are two alternative theories to classic plant successional models that are considered more effective constructs for understanding changes in plant communities following disturbance, invasion, or treatment (Allen-Diaz and Bartolome, 1998). Vegetation changes at mid-elevations, specifically within sagebrush steppe, have become an archetype for state-and-transition concepts and modeling (Laycock, 1991). State-and-transition models (STMs) for Wyoming big sagebrush communities generally suggest that exotic annual grasses, such as cheatgrass or medusahead (*Taeniatherum caput-medusae*), invade and promote wildfire occurrence in ways that further favor their dominance and inhibit perennials. The outcome of this grass/fire cycle is that perennial communities are converted to (that is, transition to) annual grasslands, which is an alternative, stable state from which it is difficult or impossible to redirect the plant community towards the native perennial state (Bagchi and others, 2013; Chambers, Miller, and others, 2014). Additional alternative states can include near monocultures of introduced perennial grasses (for example, crested wheatgrass) that are seeded to stabilize soils and preempt exotic annual grasses (Hull and Klomp, 1966; Marlette and Anderson, 1986; Hulet and others, 2010). Transitions among the three dominant states (that is, native mixed woody/herbaceous, exotic annual, or introduced perennial) are caused by disturbances such as fire, grazing, or management actions or treatments. These state changes are relevant to fuel breaks because the intent is to leverage them to render the treatment area in a stable state that has consistently lower hazardous fuels. The National Resources and Conservation Service Ecological Site Descriptions (Natural Resources Conservation Service, 2018) contain STMs for plant communities.

A more contemporary view of invasion and recovery in these ecosystems uses the terms "resistance" to annual grass invasion and "resilience" from disturbance (Chambers, Bradley, and others, 2014). These concepts are operationalized in the Fire and Invasives Assessment Tool (FIAT), which is used to prioritize areas for land treatments. Areas with moderate amounts of resistance and resilience are deemed most suited for treatment, because they can benefit from intervention and they have enough growth potential to respond to treatment.

Communities with high resilience have a high recovery potential for many or most native plant species (for example, Seefeldt and others, 2007), which limits the available space and soil resources that exotic annuals require for invasion and thus confers resistance to invasion. From a rehabilitation or restoration perspective, areas with high resistance and resilience have a high likelihood of recovering without intervention.

Conversely, at the lowest elevations, in or near salt desert or low-elevation Wyoming big sagebrush, resistance and resilience is low with longer recovery rates, making restoration challenging in these sites. At the core of the resistance and resilience concepts is the hypothesis that re-sprouting perennial grasses can quickly control soil water and nutrients and competitively displace exotic annuals, and that minimum temperatures also inhibit exotic annuals at higher elevations. An emphasis of the resistance and resilience concept is placed on promoting deep-rooted perennial bunchgrasses, as they are considered to confer greater drought resilience and thus longer-term competition against exotic annuals than shallow-rooted and ephemeral grasses such as Sandberg bluegrass (Reisner and others, 2013). Healthy stands of perennial bunchgrasses have smaller and less connected bare-soil patches that are considered to minimize available microsites for annual grass invasion, particularly if biological soil crusts cover the interspaces and inhibit establishment of annual grasses (Deines and others, 2007; Reisner and others, 2015). Biological crusts prevented germination of exotic annuals in a greenhouse study (Serpe and others, 2006). These biological and physical elements that increase resistance to annual grass invasion also confer soil stability, which is another major management concern in the affected plant communities. Erosion following disturbances such as fires or stand failure can be extensive, strongly affecting ecosystem properties and feeding back on annual grass communities (Germino and others, 2016). The duration of bare soil exposure is a major factor affecting erosion risks.

Potential Effects of Green Strips on Plant Communities and Soils

The purpose of green stripping is to provide vegetation that will likely prevent the growth and spread of annual invasive vegetation (for example, *Bromus* spp.) but will also consist of relatively shorter vegetation with higher moisture content and, thus, reduced fuel loading (Davison and Smith, 1997). Plants that are commonly used for green stripping include crested wheatgrass and forage kochia (Monsen, 1994; Pellant, 1994), although other species are also used (Davison and Smith, 1997; Harrison and others, 2002; St. John and Ogle, 2009; Maestas, Pellant, and others, 2016). Crested wheatgrass has shown some effectiveness as a strong competitor against less desirable nonnative annual forbs and grasses, such as halogeton (*Halogeton glomeratus*) and cheatgrass, and is considered fire tolerant (Hulet and others, 2010; Nafus and others, 2016; Svejcar and others, 2017). However, its ability to spread into nearby undisturbed sagebrush environments and outcompete native bunchgrasses and other desirable plants has been a subject of valid concern (Pyke, 1990; Bakker and Wilson, 2004). To prevent further disruption of the surrounding sagebrush ecosystems, crested wheatgrass may require careful management and frequent maintenance (Hansen and Wilson, 2006) because, once established, it can be difficult to remove (Hulet and others, 2010; McAdoo and others, 2017; Svejcar and others, 2017). Forage kochia is also used widely in fuel break construction (as well as in a few experimental applications) (Graham, 2013). This medium-sized sub-shrub, originally from central Asia and Europe (McArthur and others, 1990) typically contains greater moisture content than crested wheatgrass, which helps to better prevent the spread of fire compared to crested wheatgrass (Graham, 2013). Forage kochia has relatively high resilience after wildfire and various case studies suggest it can be competitive with cheatgrass (McArthur and others, 1990; Harrison and others, 2002). However, forage kochia is subjected to similar concerns as crested wheatgrass regarding potential spread into adjacent sagebrush environment. Kochia has the potential to spread at least 700 m beyond the original planting areas (Gray and Muir, 2013) and might hinder attempts to maintain or recreate proper functioning of original sagebrush communities (Graham, 2013).

Areas converted to green strips are usually first treated with herbicides to remove competition and then drill seeded with vigorous species known to confer high resistance and resilience as described above. Establishment success varies with method of seeding, generally increasing if applied with a rangeland drill than by aerial broadcast, and is typically greater at higher elevations receiving more precipitation (Knutson and others, 2014) (fig. 19). In contrast, seeding crested wheatgrass or forage kochia on sites having only 200 mm/year of precipitation did not appear to increase those species on treatment areas, while many-fold increases were evident in areas receiving 400 mm/year. Unsuccessful seedings in sites that are less resistant or resilient are more likely to degrade into annual grasslands, whereas successful seedings typically lead to monospecific stands (Beyers, 2004), which could influence stand development over time. For instance, although both crested wheatgrass and forage kochia are moderately deep rooted and likely use soil nutrients thoroughly, resulting in relatively vigorous growth, these species generally do not use soil water as efficiently as a diverse stand of woody and herbaceous perennials in native communities (for example, Kulmatiski and others, 2006). This dynamic can result in greater duration of available soil water and potentially facilitate invasion by exotic tap-rooted forbs (for example, Hill and others, 2006; Prevéy and others, 2010) or cheatgrass. Invasion of interspaces maybe somewhat controllable if biological soil crusts can become established (Serpe and others, 2006), and research is underway to determine how to facilitate that process (Condon and Pyke, 2016). It is likely that dispersal of moss and other biological soil crust life forms (from persisting remnant patches) can be used to “seed” areas like green strips in the near future (Bowker, 2007), increasing abundance of soil crusts and stabilizing the soils and plant community to the desirable seeded plant species.

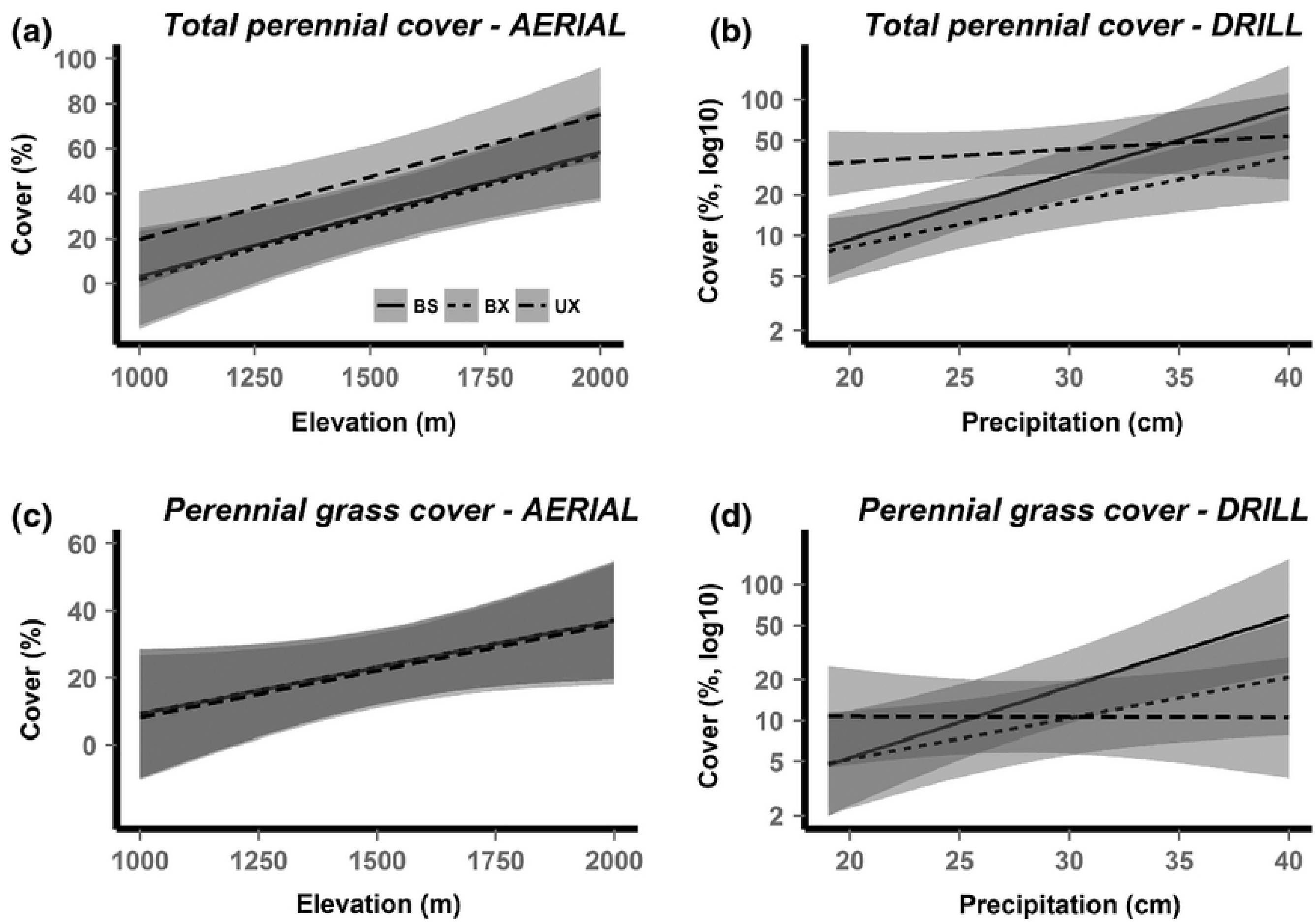

Figure 19. Cover of all (*a, b*) perennial life forms and (*c, d*) perennial grasses in burned-seeded (BS), burned-unseeded (BX), and unburned (UX) treatments at aerial and drill projects. Other significant model covariates not shown were held constant at intermediate values (precipitation: 28 cm; age: 12 years; elevation: 1,400 m; heat load: 0.94). Shaded bands are 95-percent confidence intervals, and darker areas represent overlap. Used with permission from Knutson and others (2014).

Potential Effects of Brown Strips on Plant Communities and Soils

Brown strips involve the removal of above-ground biomass and exposure of bare mineral and organic soil, leading to high potential for recolonization by early seral native or naturalized perennial species following treatment (for example, Sandberg bluegrass, crested wheatgrass) or invasion of exotic, annual herbs with propagules present. Recolonization and invasion potential necessitates intensive annual (or more frequent) treatments to eliminate plant cover. Any trace vegetation that evades herbicide treatment would likely have abundant soil moisture and nutrients available, due to the lack of plant community usage. If brown strips are maintained, there will be no carbon inputs by plants and soil carbon would likely decrease. Brown stripping likely reduces biological crusts, eliminating the nitrogen fixation potential they confer and increasing the potential for physical crusts (reducing aeolian but increasing water erosion), as could be verified with slaking tests. There can be a substantial time lag between the construction of brown strips and the formation of physical crusts, as days to months may elapse for wetting and drying cycles to trigger the crusting. In the interlude, soils may be unconsolidated and aeolian erosion risks are greatly increased. Disturbances to crusts from hooves or intense precipitation events could reduce transient soil aggregation and increase the availability of erodible soil. Depending on the orientation of brown strips to water and wind flow (that is, runways), erosion risks could increase, although catastrophic wind erosion (>about 2 cm of surface soil removal) generally requires much larger disturbance area-to-perimeter ratio than fuel breaks typically provide (Miller and others, 2012). In the absence of plant transpiration, soil water storage would increase, as evaporation is the only means for water loss until potential saturation and runoff occur (on slopes). If brown strip maintenance ceases, recolonization will be affected by which species' seeds are present; however, species that are expected to be favored include those that have seeds capable of anchoring onto hard soil crusts (for example, awns on cheatgrass seeds, Hoover and Germino [2012]), that rapidly germinate and grow (for example, many exotics, in addition to cheatgrass, such as burr buttercup [*Ranunculus testiculatus*] or mustards [for example, *Sisymbrium altissimum*]), or that can capitalize on abundant deeper water supplies (for example, tap-rooted forbs that include many exotics, such as skeletonweed [*Chondrilla juncea*], thistles [for example, *Cirsium* spp.], and knapweeds [*Centaurea* spp.]). The rate at which plant community cover and height develop, as herbs recolonize or invade unmaintained brown strips, will vary considerably with climate, weather, and species involved.

Even with diligent brown strip implementation and maintenance, vegetation could evade initial or follow-up treatments in several ways. The timing of treatments relative to weather patterns is an important determinant of post-treatment plant emergence. Blading the soil surface may not remove all meristems of perennials, and some seed may remain. Without herbicides, plant establishment is expected in spring or fall following blading, contingent on sufficient rain and suitable temperatures. Pre-emergent herbicides such as imazapic can be expected to reduce or eliminate new seedling establishment following blading for about 1 year (Owen and others, 2011). Herbicide applications in the years following fire is expected to be most effective if timed to precede germination events, and predicting these and applying treatments in a timely fashion can be challenging (allowing for some residual cover of early seral species).

Potential Effects of Mowed Fuel Breaks on Plant Communities and Soils

Mowing Wyoming big sagebrush reduced sagebrush cover, density, canopy volume, and height for at least 20 years in a study by Davies and others (2009), and Pyke and others (2014) found that woody biomass was reduced by at least 85 percent for 3 years in sites with high resistance and resilience. The degree of transformation achieved by mowing will be determined in part by the large height differences that exist for adult or mature plants between the different plant communities of interest. Shrubs in salt desert communities can range from about 1.5 m for four-winged saltbush (*Atriplex canescens*) and greasewood (*Sarcobatus vermiculatus*) to less than 50 cm for winterfat or shadscale. Low sagebrush species are generally less than 30 cm high, yet Wyoming and mountain big sagebrush can vary from about 30 cm to nearly 1 m high, and basin big sagebrush is frequently 1 to about 3 m high. In big sagebrush communities, some bunchgrass species are generally small statured (for example, <about 15 cm for Sandberg bluegrass, with most foliage just a few cm above ground), while others may have leaf heights greater than 1 m (for example, Great Basin Wildrye [*Leymus cinereus*]). After mowing vegetation to 15–30 cm in height (to reduce flame length), grasses, herbs, and some shrubs will have meristems at the soil surface that are able to resprout and regain height loss the following growing period, provided that moisture and temperature conditions are adequate.

Clipping off apical meristems (that is, the top of the plant, which regulates primary growth) causes many woody species to “bush out” due to release of apical dominance controlled by hormone interactions. Sagebrush and several other woody species have meristems above ground and their regrowth potential following clipping will depend strongly on how clipping height relates to the location of their meristems. Regrowth of sagebrush heights can vary substantially depending on subspecies, topographic position, and elevation effects. Basin big sagebrush, for example, grows much taller and faster than does Wyoming big sagebrush (McArthur and Welch, 1982), and topographic depressions are often more fertile and support greater growth. Basin big sagebrush in their fifth to seventh year following planting grew 10–15 (mean=12) cm/yr compared to 6–11 (mean=8) cm/yr in mountain and Wyoming big sagebrush in a deep-soil site (26.8 cm/yr of precipitation; 1,700 m in elevation; McArthur and Welch, 1982). However, no meristems would typically be left following mowing of basin big sagebrush to 15 cm height. Low sagebrush species may be intermixed with big sagebrush communities, and have slower vertical growth following mowing compared with big sagebrush subspecies. Based on these annual growth rates, sagebrush plants could be expected to vary from no recovery to nearly a doubling of height in the year following mowing, depending on the species and its initial height. For instance, minimal growth following mowing would be expected for a low-sagebrush whose height is near or below cutting blades, and for a mature basin big sagebrush that has a thick singular trunk and no meristems above blade height. In contrast, if younger basin big sagebrush are cut, then large incremental growth would be expected (about 12 cm added to a mowed base height of 15 cm).

Mowing of herbaceous vegetation or other woody species that have meristems at or near ground (for example, rabbitbrush [*Chrysothamnus* and *Ericameria* spp.]) will initially reduce the standing litter, but regrowth to pre-cutting heights or potentially more (due to compensatory responses) is expected in the year following treatment (provided weather is suitable). Additionally, soil disturbance and release of resources after mowing in sagebrush stands may further benefit herbaceous species in subsequent years. Pyke and others (2014) found mowing to have no beneficial reduction of herbaceous fuels, and instead observed at least a 36 percent increase in herbaceous fuels (biomass) over 3 years following mowing. Indeed, increased production of herbaceous plants following mechanical (and sometime chemical) removal of big sagebrush is also well-documented elsewhere (Hedrick and others, 1966; Wambolt and Payne, 1986; Swanson and others, 2016) and raises issues about the efficacy of mowing as a fuel treatment. Moreover, bunchgrasses must be present prior to mowing to reduce or prevent exotic grass invasion after mowing as demonstrated by a series of studies in eastern Oregon. Researchers found that mowing of degraded sagebrush steppe, where exotic annual grasses were already fairly dominant, did not increase perennial herbs, but did increase exotic annual grass and annual forb biomass production by as much as 7–9 times, respectively, by the third post-treatment year (Davies and others, 2011; see also Davies and others, 2012a; Davies and Bates, 2014). In northern and central Nevada, mowing resulted in more cover of litter, perennial grasses, cheatgrass, and exotic forbs over a span of 1–10 years (Swanson and others, 2016). However, mowing high-elevation mountain big sagebrush where exotic annual grasses were less common enhanced native herbs, including desirable bunchgrasses, but did not increase exotic annuals (Davies and others, 2012b).

Erosion risks would be minimal for mowed fuel breaks, and the soil fertility and hydrology effects of mowing are likely substantially different than for brown strips. Mowing often causes foliar shoots that have relatively high nutrients, such as nitrogen, to be deposited to soil (depending on phenology of species at the time of clipping), especially compared to leaves that drop to soil after normal translocation of nutrients into the plant. Thus, we can hypothesize that litter resulting from mowing would have greater decomposition rates than normally senesced foliage. Unlike brown strips, the mowed plant community would continue to use available soil moisture and nutrients, providing resistance to annual invasion; though the ratio of soil resources per remaining leaf unit area would likely increase. However, in a study by Davies and others (2009), sagebrush leaves that evaded cutting did not have enhanced foliar nutrition.

Plant Community Responses Adjacent to Fuel Breaks

Fuel breaks also may influence surrounding, untreated plant communities by providing a seed source of species that were seeded into or inadvertently colonized fuel breaks, as well as potential indirect effects of altered microclimates, wind velocity, soil movement and deposition, surface and soil-water hydrology, and snow deposition patterns. For example, both crested wheatgrass and forage kochia have been reported to emigrate from areas they were seeded into the surrounding landscapes (Marlette and Anderson, 1986; Gray and Muir, 2013). However, that process may take considerable time to develop and may not occur everywhere, as in a recent study that found forage kochia did not disperse outside of treated areas for the first 5 years after seeding (Satterwhite, 2016). In 24 fuel breaks (similar to unmaintained brown strips) across California, blading (bulldozing) in chaparral habitat resulted in increased nonnative cover (relative), density, and richness, especially 0–20 m from brown strip edges (Merriam and others, 2006). Grazing and time both also influenced these invasion rates. Careful research investigating where and why fuel breaks become corridors for weed invasion in surrounding landscapes is needed.

The low vegetation cover or low vertical height of linear fuel breaks may result in unintended climatological effects within the fuel breaks and this could have ecological consequences for surrounding plant communities. In areas that have winter snow accumulation and significant wind, redistribution of snow off of fuel breaks and into the surrounding taller vegetation would be likely. Sagebrush and other tall perennials also affect radiation regimes, which feedback to affect snow retention and soil microclimate. Greater bare soil exposure could result in warmer soils (with less canopy shading of soil) and could impact species like cheatgrass that are active in early spring and late fall. The increases in soil moisture that would accompany vegetation reduction on fuel breaks would likely increase effective water availability for surrounding, un-treated vegetation along and outside the treatment boundaries, potentially enhancing plants outside the border of fuel breaks.

Question 3. What Are the Effects of Fuel Breaks on Greater Sage-Grouse, Other Sagebrush Obligates, and Sagebrush-Associated Wildlife Species?

Fuel breaks have the potential to directly affect populations of greater sage-grouse (hereinafter sage-grouse) and other sagebrush-associated species across multiple spatial-scales. In this section, we examine habitat needs and conservation requirements for sage-grouse and other key species relative to the potential for fuel breaks to directly modify habitat, fragment habitat, disrupt seasonal habitat use, impede movement of individuals between populations, influence predator-prey relationships, or cause other deleterious effects on species of concern.

Fuel breaks in shrublands may influence animals across multiple levels of biological organization (individuals, populations, and communities) and across a range of temporal and spatial scales. This ecological complexity often makes it difficult to understand fully the effects of habitat alterations. Some changes increase mortality conspicuously (for example, higher predation rates), whereas other habitat changes have negative effects on animals that are difficult to observe or measure. These subtle effects, such as lower fecundity resulting from increased stress or poor body condition, may result in responses at the population level that are not detectable for several years. Furthermore, habitat treatments may alter prey populations, such as rodents (McAdoo and others, 2006), which could result in delayed population response by their predators. Changes in predator populations can have ecological consequences far outside an area of disturbance because predators tend to be more wide-ranging than prey.

We begin our assessment of the potential effects of fuel breaks on wildlife by first examining several issues that may be common across different types of fuel breaks in sagebrush landscapes: habitat fragmentation and loss, edge effects, and linear features (see section, “General Concepts for Wildlife Considerations”). We then evaluate the empirical evidence for potential effects of green stripping, brown stripping, and mowing treatments on wildlife.

General Concepts for Wildlife Considerations

Wildlife habitats are characterized by the structure and composition of vegetation and various abiotic elements in a landscape, some of which have direct relation to fuel break design and function. Habitat structure has three dimensions, measured typically as two-dimensional ground cover and height. Habitat composition encompasses plant species richness or functional group diversity (for example, perennial grasses), as well as the relative amounts of cover types across a landscape. Cover types can be defined at the species level, functional group level, or broader ecological classes (for example, riparian, grassland, shrubland), depending on level of information needed or available. Abiotic elements, such as amount of rock and bare mineral soil (that is, usually measured as bare ground) and size of interspaces (that is, canopy gaps) among plants, are important components of habitats because they influence movements and cover. Subsurface aspects of habitats, particularly soils, influence burrowing animals as well as plant communities.

Most terrestrial vertebrates respond to habitat structure and composition because of the strong influence on development, growth, survival, and production (that is, number of offspring or fitness). Habitat structure and composition have direct influences on an animal's ability to find food, identify locations to reproduce and raise young, avoid predators, and shelter from stressful or life-threatening environmental conditions. Animals also are aware of the spatial and temporal (that is, diel or seasonal changes) characteristics of their habitats. Whether evaluating their environments from above, such as a bird, or from the ground, animals are adept at navigation and spatial recognition of the distribution of critical resources in their environments. In many cases, animals can perceive potential threats, using habitat resources to minimize those risks, but in anthropogenically modified landscapes, novel risks may not be recognized. Any rapid changes to the structure or composition of habitats can be stressful to animals and may reduce individual fitness and population viability.

Habitat Fragmentation and Loss

Although the effects of fuel breaks on wildlife habitat remains largely unstudied, there is a rich scientific literature on the effects of other anthropogenic landscape features that result in direct habitat loss and subdivide continuous habitats into smaller components, such as happens with development of roads, power-lines, agriculture, and housing (Wilcox and Murphy, 1985; Robinson and others, 1995; Hill and Caswell, 1999; Fahrig, 2002). Most research on the effects of habitat fragmentation in sagebrush shrublands has focused on passerine bird species, which tend to be negatively affected by reduction in the size of sagebrush patches or core habitat (Knick and Rotenberry, 1995; Knick and Rotenberry, 2002; Hethcoat and Chalfoun, 2015b). Core habitat, or habitat that is relatively large and contiguous, contains environmental conditions and resources needed to sustain an individual or a population. The requisite size of core habitat patches is relative for each species, but large patches of habitat that extend beyond individual home ranges (Knick and Rottenberry, 2002) tend to support higher abundance of individual species and greater diversity (Rodewald and Vitz, 2005). For example, the abundance of pygmy rabbits (*Brachylagus idahoensis*) in Utah increased significantly with distance into sagebrush stands, particularly greater than 100 m from the edge created by mechanical treatment (Pierce and others, 2011). Similarly, sage-grouse leks are more likely to be abandoned when contiguous patches of sagebrush are smaller (Wisdom and others, 2011), and entire populations have even disappeared where landscape cover of sagebrush falls below 65 percent (Aldridge and others, 2008). Occupied leks have approximately twice the amount of sagebrush habitat as those leks that have been extirpated (46 versus 24 percent, respectively) and 10 times the size of sagebrush patches (4,173 versus 481 ha, respectively; Wisdom and others, 2011).

Subdividing or fragmenting once-continuous sagebrush habitats may be problematic for some species (Coates and others, 2014a), but our lack of understanding of the mechanisms causing population-level effects (Fletcher and others, 2007) makes it difficult to adapt fuel breaks to minimize negative consequences. For example, Knick and Rotenberry (2002) assessed how landscape composition, configuration, and change influenced passerine bird population dynamics in sagebrush steppe and hypothesized that fragmentation (from any given cause) of otherwise intact native habitat might influence productivity through differences in breeding density, nesting success, or nest predation or parasitism. They concluded that fragmentation was important in determining the distribution of shrubland-obligate species like Brewer's sparrows (*Spizella breweri*), sage sparrows (*Amphispiza belli*), and sage thrashers (*Oreoscoptes montanus*), but the causal mechanisms were unresolved. In other cases, it has been shown that loss in the amount of habitat surrounding populations (for example, degraded habitat at larger spatial scales) is most influential in affecting fitness outcomes of wildlife species in sagebrush ecosystems (for example, for sagebrush-obligate songbirds; Hethcoat and Chalfoun 2015a), perhaps by disrupting meta-population dynamics. Loss of habitat from energy development has been correlated with increased nest predation of sagebrush-obligate songbirds, especially by rodent species that increased in abundance with loss of sagebrush (Hethcoat and Chalfoun, 2015b).

We suspect that habitat disturbances (such as fuel breaks) that subdivide the landscape into isolated patches will make it more difficult for animals to migrate seasonally among complimentary habitats (Harris and Reed, 2002), but the empirical evidence for sagebrush-associated wildlife is lacking. For less vagile animals, such as some small mammals and lizards, it is plausible that fuel break systems could have an isolating effect.

Edge Effects

Because fuel breaks typically create sharp transitions with surrounding habitats, they increase the amount of edge within a landscape, and thus also increase edge effects. Here, we define edge as the interface between two or more adjacent ecological communities or land cover types. Although fuel breaks are often built along existing roads, where edges already exist, there may still be increased edge effects caused by both the road improvement or widening (that often accompanies fuel break construction), as well as the addition of parallel edges adjacent to roads created by the fuel break treatment. Edge effects might resemble natural ecotones or have considerably different environmental characteristics (that is, atypical for a given landscape), including changes in species composition and relative abundance (Woodward and others, 2001; Rodewald and Vitz, 2005); changes in biotic interactions, such as predation (Winter and others, 2000; Vander Haegen and others, 2002), parasitism (Vander Haegen and Walker, 1999), and competition (Ingelfinger and Anderson, 2004); and changes in environmental gradients.

Changes in composition and relative abundance of wildlife species in fuel breaks may result from novel environmental conditions associated with edges or ecotones, but understanding these causal factors is difficult because of confounding effects of biotic interactions and fragmentation. Empirical data on the effects of edges on sagebrush-associated wildlife are lacking, although ecotones between sagebrush stands and sagebrush removal areas (that is, similar to fuel breaks) are thought to attract some species that forage in open habitats, but use adjacent shrubs as cover (McAdoo and others, 2004; Beck and others, 2012). Other species, such as pygmy rabbits (fig. 20), may avoid habitat edges if competitors (for example, cottontails [(*Sylvilagus* spp.] and jackrabbits [(*Lepus californicus*]) prefer these ecotones (Pierce and others, 2011).

Figure 20. Pygmy rabbit (*Brachylagus idahoensis*). Photograph by H. Ulmschneider (Bureau of Land Management) and R. Dixon (Idaho Fish and Game).

Biotic interactions, especially predator-prey, are better documented than other edge effects. Some predators prefer edges; thus, fuel breaks may increase vulnerability of grassland or low-cover species that colonize fuel breaks or species that are moving along or attempting to cross fuel breaks. For example, nesting probability of common ravens (*Corvus corax*) increases near edges, specifically where sagebrush shrubs interface areas dominated by crested wheatgrass or cheatgrass (Coates and others, 2014b; Howe and others, 2014). Edge not only positively influences breeding pairs of ravens but also influences occurrences of non-breeders that are often numerous and transient (Coates and others, 2015). Ravens use visual cues while hunting and edge-dominated areas may offer greater opportunity to detect their prey than those areas with contiguous stands of sagebrush. Edges likely provide ravens the opportunity to more readily locate and depredate nests of other bird species. In areas with ravens, a 1 percent decrease in shrub cover can increase the odds of predation by as much as 7.5 percent (Coates and Delehanty, 2010). Ravens are attracted to edge environments largely associated with lack of shrub canopy. Fuel breaks that intersect shrublands may result in increased ravens and other predators.

The attraction of edges to predators has consequences for prey. For example, greater sage-grouse nests located in fragmented habitats (that is, remnant patches of sagebrush within an agricultural matrix) were approximately nine times more likely to be depredated than those in contiguous habitats, and the majority of nests in fragments were depredated by ravens and other corvids (Vander Haegen and others, 2002). Similarly, increased habitat loss and creation of edges due to natural gas development has been associated with decreased nest survival and increased rodent nest predation rates on sagebrush songbirds (Hethcoat and Chalfoun, 2016a, 2016b). Studies have shown that ravens are important predators of eggs and nestlings of multiple species of birds (Andren, 1992; Luginbuhl and others, 2001), including sage-grouse in the Great Basin (Coates and others, 2008; Coates and Delehanty, 2010; Lockyer and others, 2013). Fuel breaks in nesting habitat might put sage-grouse at relatively higher risk of nest loss, which can influence population growth (Taylor and others, 2012). Sage-grouse tend to avoid nesting in sagebrush environments with relatively high densities of ravens (Dinkins and others, 2012), and raven abundance has been associated with changes in sage-grouse incubation patterns (Coates and Delehanty, 2008) and their nest survival, while other predators have been found to be less important within the Great Basin (Coates and Delehanty, 2010). Although corvids have been influential nest predators in the Great Basin, additional studies within and outside the Great Basin are needed to help clarify spatial variation in the impacts to prey communities. Lastly, increased edge has positive effects on other generalist predatory birds that likely impact sage-grouse adult and juvenile survival, particularly red-tailed hawks (*Buteo jamaicensis*) and Swainson's hawks (*B. swainsoni*), both of which are effective predators of adult and juvenile sage-grouse (Conover and Roberts, 2017).

Although the line between fuel break and surrounding vegetation may be sharp, environmental changes of a fuel break are likely to extend into the surrounding vegetation. This environmental gradient will have varying effects on animals depending on their environmental tolerances, but with decreasing effects with distance from edge. Compared to the core of surrounding habitats, conditions at the edge are usually warmer, drier, windier, and have more diel and seasonal variability. Thus, these disturbances can influence the remaining native vegetation by altering resource availability and species composition; particularly at the edge between cover types (Saunders and others, 1991). Within sagebrush ecosystems, surrounding habitats that are immediately adjacent to fuel breaks, likely often consist of less shrub canopy cover than those areas located within contiguous core habitat. Total shrub cover is one of the most critical microhabitat factors related to nest site selection and survival across sage-grouse range (Connelly, Reese, and others, 2000; Connelly, Schroeder, and others, 2000b; Connelly and others, 2004) and most notably within the Great Basin (Kolada and others, 2009; Lockyer and others, 2015; Gibson and others, 2016), where the large majority of fuel breaks have been proposed. Overstory shrub cover is also important for pygmy rabbits (Larrucea and Brussard, 2008; Lawes and others, 2013), black-tailed jack rabbits (*Lepus californicus*) (Johnson and Anderson, 1984), ground squirrels (Yensen and others, 1992; Steenhof and others, 2006), and several passerine birds (Baker and others, 1976; Knick and Rotenberry, 1995; Chalfoun and Martin, 2007). These edge effects could be resulting in "functional" habitat loss (Aldridge and Boyce, 2007), where otherwise suitable sagebrush habitat adjacent to roads (or proposed fuel breaks) are avoided, as has been shown for greater sage-grouse (Aldridge and Boyce, 2007). Functional habitat loss for sage-grouse in otherwise suitable sagebrush habitats may extend out at least as far as about 2 km in winter habitat (Carpenter and others, 2010).

Linear Features

A feature of fuel breaks that is different from other forms of wildlife habitat alteration is their linearity. Few natural features in the environment are as linear as those that are anthropogenic (for example, transmission lines, fences, roads, fuel breaks). Variation likely exists in how wildlife perceive these linear features within a landscape compared to natural irregularly shaped features and, as such, these linear features are likely to have different consequences among species. Sage-grouse showed strong avoidance of edges in Canada during nesting (Aldridge and Boyce, 2007). Others have shown that, while on the ground, sage-grouse tend to move along topographic features and to avoid areas without sagebrush cover (Dunn and Braun, 1986). These behaviors are fairly typical of wildlife in general, which often spend time in close proximity to, or avoid crossing (including flying over), non-vegetated areas (for example, brown strips), and will instead attempt to cross in areas that offer at least some protective cover (Richard and Armstrong, 2010). As such, we suspect that some species might move unusually long distances as they attempt to locate an area to transit the fuel break. If fuel breaks reduce successful dispersal, there could be consequences for colonization of new habitats, metapopulation dynamics, or gene flow.

Within sagebrush ecosystems, newly created fuel breaks might impose travel corridors allowing terrestrial predators to readily access sagebrush habitats and operate at much larger spatial scales. For example, mammalian predators, including coyotes (*Canis latrans*), badgers (*Taxidea taxus*), and red foxes (*Vulpes vulpes*), have been shown to use anthropogenic corridors as travel routes while hunting, presumably improving functional response by easier access to prey (Crête and Larivière, 2003; Frey and Conover, 2006). However, potential mechanisms of such effects also have been debated (Larivière, 2003), despite the observed increased in predation rates along edges. Badgers in British Columbia displayed a preference for both roads and general linear corridors (Apps and others, 2002). Common ravens (*Corvus corax*) and other predatory birds are attracted to roads and cleared linear right of ways within shrublands (White and Tanner-White, 1988; Knight and Kawashima, 1993; Coates and others, 2014b; Howe and others, 2014). In sagebrush habitats in Idaho, raven occurrence declines exponentially with the distance from transmission lines and roads (Howe and others, 2014). The authors indicate that ravens were often observed flying over roads, particularly in the early morning hours, presumably searching for prey. Reports of similar observations of ravens flying along linear networks have been reported elsewhere (Bui and others, 2010). Direct removal of overstory shrubs within fuel breaks likely helps to increase movement speeds of predators traveling from one point to another. This effect may diminish in green strips when seeded species reach maturity.

Many fuel breaks are associated with roads, and there is considerable empirical evidence that roads have negative effects on wildlife through vehicle collisions, noise, pollutants, and habitat alteration. For example, Ingelfinger and Anderson (2004) examined how unpaved roads constructed for natural gas extraction in Wyoming big sagebrush habitats influenced passerine birds. They found that Brewer’s sparrow and sage sparrow numbers within sagebrush stands were reduced by 39–60 percent within 100 m of the road despite little traffic (<12 vehicles per day). They concluded that the bird responses were unrelated to vehicles and were likely caused by edge effects, habitat fragmentation, and arrival of other passerine species along the road corridor. Some animals are attracted to roads, such as snakes using road surfaces for thermoregulation, which can further increase probability of vehicle-related mortality. In southeastern Idaho, a road survey through sagebrush steppe revealed that most road mortality was associated with gophersnakes (*Pituophis catenifer*) and rattlesnakes (*Crotalus oreganus*), especially where roadsides were dominated by nonnative grasses (Jochimsen and others, 2014). Horned Larks (*Eremophila alpestris*) are attracted to roadways where they forage on windblown seeds that collect on dirt roads (Ingelfinger and Anderson, 2004).

Roads and other linear right of way features can have varying effects on sage-grouse populations (Manier and others, 2014). These linear features may simply be avoided by both greater and Gunnison sage-grouse (*Centrocercus minimus*) (Aldridge and Boyce, 2007; Carpenter and others, 2010; Aldridge and others, 2012), or are thought to alter productivity and survival of local sage-grouse populations, and even result in local extirpations of leks, as has been observed along Interstate-80 in Wyoming (Connelly and others, 2004). However, smaller, less-frequently used trails may be selected by brooding greater sage-grouse during the summer, possibly for the abundance of succulent invasive forbs that are associated with these disturbed sites (Aldridge and Boyce, 2007). If fuel breaks similarly provide succulent food resources, sage-grouse could be drawn into these habitats, possibly increasing predation risk.

Although one study of very coarse road density did not support impacts to sage-grouse range-wide persistence (Aldridge and others, 2008), roads did correlate with lek extirpations (Wisdom and others, 2011). Other studies done at local scales have demonstrated negative associations with roads and both greater and Gunnison sage-grouse avoidance or productivity (Braun, 1986; Lyon and Anderson, 2003; Holloran, 2005; Aldridge and Boyce, 2007; Aldridge and others, 2012; Kirol and others, 2015). Perhaps the discrepancy between these studies was the differences in data collection, where the range-wide presence analyses (Aldridge and others, 2008) was unable to consider numerous secondary roads and underrepresented total road density.

Potential Effects of Green Strips on Wildlife

The sowing of nonnative species into green strips will influence wildlife habitats and use. A study of crested wheatgrass seedings, for example, revealed that these areas supported fewer nesting bird species and lower densities of birds, mammals, and reptiles compared with intact stands of sagebrush (Reynolds and Trost, 1980). Also, some species seeded into green strips may act as an attractant to wildlife because of higher moisture content, chemical composition, or other characters. Butterflies, for example, could take advantage of seeded areas that provide abundant (or even unique) nectar resources (McIver and Macke, 2014). Other sown species, however, may be unpalatable or undesirable to pollinators or grazers. Forage kochia, for example, has been found to share similar dietary characteristics as sagebrush for sage-grouse, but a recent study indicated that sage-grouse do not tend to consume forage kochia and instead continue to eat the native sagebrush (Graham, 2013). Herbicide use to create green strips could potentially have negative effects on wildlife, though very little research has evaluated these consequences (Freemark and Boutin, 1995).

Green strips may also create ecological traps (Remes, 2000; Bock and Jones, 2004) that reduce survival for some wildlife if they are attracted to an area for food resources but in the process get exposed to higher rates of mortality. For example, sage-grouse maybe drawn into these more risky open areas to seek potential food resources, as long as they have suitable escape cover provided by near-by patches of sagebrush (Dahlgren and others, 2006; Aldridge and Boyce, 2008). However, predators, like burrowing owls (*Athene cunicularia*) and badgers, are also attracted to these open areas within shrublands because their prey (for example, deer mice [*Peromyscus maniculatus*], ground squirrels) favor these open habitats (Rich, 1986; Holbrook, Arkle, and others, 2016).

Potential Effects of Brown Strips on Wildlife

We found no literature examining the effects of brown strips on wildlife species or habitats. However, brown strips essentially involve the removal of wildlife habitat, and many of the same dynamics discussed elsewhere in this section (for example, the addition of edge effects) are likely applicable to brown stripping.

Potential Effects of Mowing Fuel Breaks on Wildlife

Mowing in shrublands may be an attractive fuel break alternative from a wildlife perspective because it reduces fuel loads and height without significantly changing plant species composition, unless exotic annuals are present (Davies and others, 2011, 2012a; Swanson and others, 2016). As such, some wildlife species could benefit from mowing treatments, especially those species that prefer disturbed areas, early successional vegetation, open grasslands, or habitat mosaics. For example, Beck and others (2012) reviewed the literature for the effects of mechanical treatments (as well as herbicide applications and prescribed burning) to identify whether these treatments are beneficial for greater sage-grouse, elk (*Cervus Canadensis*), mule deer (*Odocoileus hemionus*), and pronghorn (Antilocapra americana) in sagebrush habitats. They found some evidence that small-scale treatments (≤60-m width) in mountain big sagebrush may create suitable foraging conditions for brooding sage-grouse. Mowing Wyoming big sagebrush may also increase nutritional quality of remaining sagebrush, suggesting some benefits to wildlife (Davies and others, 2009). However, across the Great Basin, butterfly richness and abundance did not increase for 4 years after mowing or herbicide in Wyoming big sagebrush habitats, with the exception of Becker's white (*Pontia beckerii*), which were lower in mowed plots for at least 4 years relative to other plot types (McIver and Macke, 2014). Similarly, mowing Wyoming big sagebrush stands in north-central Wyoming resulted in no detectable effects on ants, beetles, or grasshoppers relative to reference sites (Hess and Beck, 2014). Mowing can also have direct and indirect consequences for wildlife. If mowing or removal of vegetation takes place during sensitive times of nesting or brood-rearing for birds (grouse, songbirds, ducks, etc.) or denning mammals, mechanical equipment could result in direct mortality. Indirectly, removal of existing vegetation creates a structurally less diverse vegetation community, which is a direct habitat loss for some species that use the shrub structure, negatively affecting wildlife, as was the case for nesting shrub-obligate songbirds when habitat was mowed (Carlisle, 2017). Other types of sagebrush removal techniques, such as use of a Dixie Harrow, have been shown to reduce pygmy rabbit abundance (based on fecal pellet counts) while increasing cottontail and black-tailed jackrabbit abundance (Pierce and others, 2011; fig. 21).

Mowing may also have direct and indirect adverse impacts on sage-grouse and other wildlife. Mowing in sage-grouse winter habitat may be particularly harmful to some populations (Eng and Schladweiler, 1972; Beck, 1977). Noise associated with mowing in sensitive sage-grouse areas also is likely to share similar detrimental effects as other types of noise on sage-grouse populations (Blickley and others, 2012). Timing of mowing in relation to sensitive areas for sage-grouse merits further investigation.

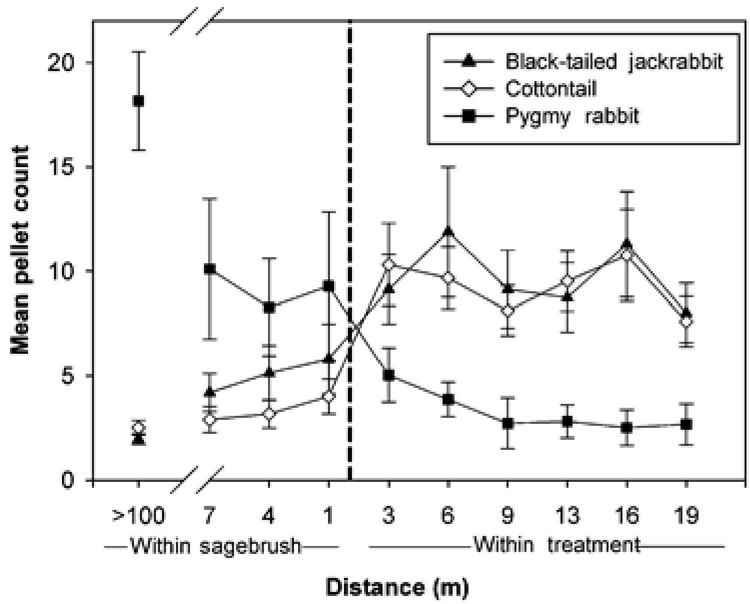

Figure 21. Mean pellet counts (±95-percent confidence interval) by leporid species in control areas in sagebrush near habitat edge, and in mechanically treated areas devoid of sagebrush. (Fig. 4 from Pierce and others [2011], used by permission of John Wiley & Sons, Inc.).

Conclusions and Recommendations

Fuel breaks serve as an important strategy for fire and land management agencies to reduce the risks and negative ecological impacts of wildfire in the Great Basin. Indeed, the Bureau of Land Management has currently identified and prioritized locations for a region-wide network of fuel breaks aimed at collectively minimizing future loss of remaining high priority habitat for sage-grouse that will include both existing, planned, and future fuel break projects (fig. 22). Ideally, these projects will be designed to help minimize future loss of key sagebrush habitat from wildfire, and to reverse recent trends in which hundreds of thousands of hectares of sagebrush habitat are degraded or destroyed each fire season (on average). However, these projects could also add thousands of kilometers of new fuel breaks to the region over the next decade or two, directly altering hundreds of thousands of hectares through habitat conversion, and indirectly affecting sagebrush plant and animal communities through creation of new edge effects and habitat fragmentation.

Enhancing the record-keeping, monitoring, and scientific assessment capacities of the Bureau of Land Management and its science partners (for example, U.S. Geological Survey, university researchers) will be critical for designing, implementing, and maintaining an effective fuel break system into the future. Various types of scientific investigation are likely to be instructive, including retrospective (“space for time”) studies of the ecological effects of existing fuel breaks (both maintained and unmaintained); study designs that incorporate comparative analysis of pre- and post-treatment conditions for planned fuel breaks; and modeling exercises that identify opportunities to minimize ecological costs, while maximizing wildland fire suppression potential to protect important natural resources and wildlife habitat. Importantly, it should be recognized that implementation of fuel break systems by land managers is a grand experiment that is not feasible for researchers to replicate or emulate at the appropriate scales; thus, integrating scientific assessment in the form of adaptive management of fuel breaks may also be a key path forward. Finally, we acknowledge that there are other aspects of fuel breaks not addressed in this report that may also be considered, including the potential for increased human impacts (for example, greater ignition rates) in remote areas, as a result of improving roads for fuel break construction and access.

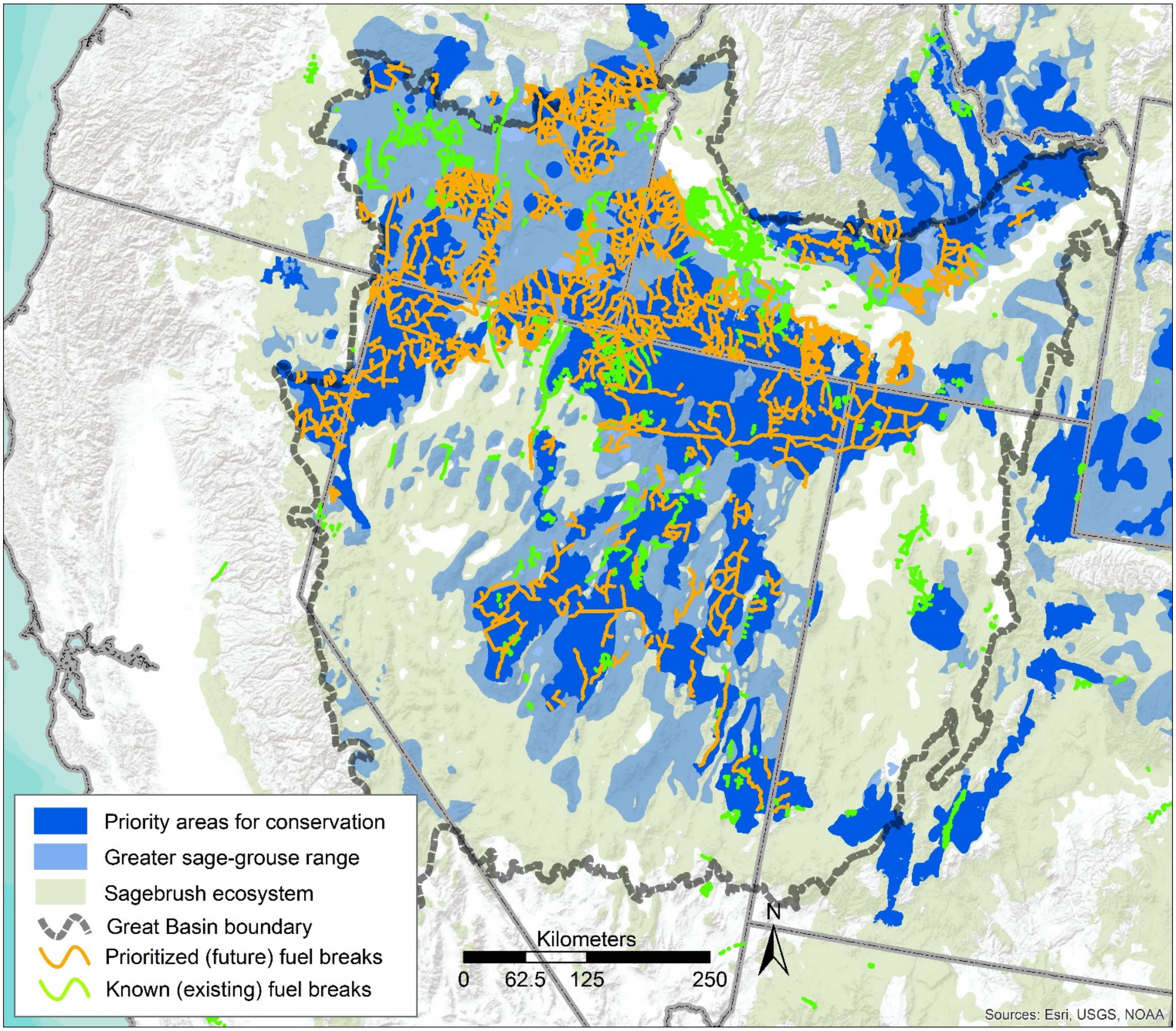

Figure 22. Existing and prioritized locations for future fuel breaks in the Great Basin relative to sage-grouse (*Centrocercus urophasian*us) habitat and priority areas for conservation. Existing fuel breaks include known linear fuel breaks (data sources as described in table 1) based on treatment information and mapped locations. Numerous unmapped fuel breaks also likely exist. Priority future locations for fuel breaks are based on conservation values derived from the Fire and Invasives Assessment Tool (Bureau of Land Management, 2017) that provides the BLM and other agencies a framework to prioritize wildfire management and conservation of sage-grouse habitat. Implementation of priority fuel breaks will require further agency planning and review, and includes both new fuel break construction as well as maintenance and enhancement of existing fuel breaks. Sagebrush ecosystem data taken from U.S. Geological Survey (2018b); greater sage-grouse distribution data taken from U.S. Geological Survey (2018a).

Fuel Break Effectiveness at Reducing Wildfire Impacts

Using wildfire simulation systems and other modeling tools to better plan the spatial configuration of landscape scale treatments would enhance strategic planning efforts to mitigate wildfire spread across the Great Basin and to use fuel breaks most effectively. Although modeling systems already exist to assist with this effort, there is concern within land and fire management communities about the lack of standard surface fuel models (that is, representing more precise fuel conditions) to characterize vegetation types typical of the Great Basin, as well as a lack of data available to validate modeled outputs of potential fire behavior. These concerns are not unique to the Great Basin; as with any application of models and fire behavior systems, fire behavior outputs are probabilistic representations of very complex phenomena which are subject to sources of errors not limited to input data, applicability of use, and model accuracy (Albini, 1976; Alexander and Cruz, 2013a, 2013b). These sources of error can lead to both under- and over prediction of potential fire behavior. However, with careful calibration of both input data and the simulation parameters, an experienced user can minimize these errors (Varner and Keyes, 2009). Various data sources can be used to better fit standard fuel models or develop custom fuel models for use in fire behavior simulations (for example, Stebleton and Bunting, 2009; Bourne and Bunting, 2011). New techniques are also available to obtain dynamic fuel conditions across large regions (for example, Li and others, 2017; Anderson and others, 2018) that could help to quantify fuel parameters for spatially-explicit, landscape-scale, modeling applications. Moreover, other vegetation-based models are being developed to aid in planning and predicting how rangeland fuel loadings might change over time under different climate and management scenarios (for example, the Rangeland Vegetation Simulator; Reeves and Frid, 2016).

The ability of agencies to weigh the potential costs and benefits of implementing extensive networks of fuel breaks would also aid in their efficient and strategic use. Modeling can help to locate fuel breaks where ecological costs may be minimized while simultaneously maximizing wildland fire suppression efforts to protect human development, important natural resources, and wildlife habitat (for example, Bar-Massada and others, 2011; Gray and Dickson, 2016; Opperman and others, 2016). However, these analytical models would benefit from more consistent record-keeping and enhanced information regarding fuel break conditions, ecological effects, and effectiveness over time and space. Although the FTEM program is a step in the right direction regarding effectiveness, there is a need for more quantitative monitoring of fuel break ability to alter fire behavior. For instance, the effectiveness of linear fuel breaks to aid in fire suppression and therefore limit fire size could be assessed across the Great Basin using different metrics of success (for example, containment). Mapped wildfire data coupled with documented wildfire suppression tactics, fuel treatment locations and maintenance history, and fire environment conditions (for example, fuels within and outside of fuel breaks, fire weather) could be used to better assess if fuel breaks aided suppression efforts and, if so, whether they were useful in controlling fire spread or meeting other fire management objectives (for example, reducing severity). Such information could also be valuable to evaluate and determine optimal and cost effective fuel break maintenance strategies. However, the lack of well-mapped historical linear fuel breaks makes a retrospective analysis difficult for many applications. Additionally, better and more comprehensive information is needed from programs that specifically and systematically monitor fuel and other vegetation conditions in fuel breaks over time, as well as their ecological effects (as described below).

Fuel Break Design Considerations for Plant Communities

In plant communities, the effectiveness and potential collateral impacts of fuel breaks mainly depend on (1) the spread of nonnative species that are seeded onto breaks or which invade the breaks, and (2) if and how fuel breaks are maintained. The impact of fuel breaks will depend on the condition, resistance, and resilience of the land converted into a fuel break, as well as in the surrounding landscape. With such little research done on fuel break impacts on plant communities, and yet with expansion of fuel breaks underway, it is vitally important to learn from the actual implementation of fuel breaks. This opportunity to learn would only be possible with carefully designed experiments and (or) comprehensive monitoring that includes species composition and biomass measured before and in the years after implementation of fuel breaks. Monitoring and analyses will be most effective if done for both the direct area on the ground converted to fuel breaks, as well as at different distances from the edge of fuel breaks into surrounding landscapes.

It is also worth pointing out that native species that do not contribute substantially to fuel accumulation and are more drought tolerant than nonnative wheatgrasses (Frank, 1994) may also have potential utility within fuel breaks in some cases. For instance, although Sandberg bluegrass senesces early in the growing season, it is drought and fire tolerant, low-statured, and competitive with cheatgrass (Howard, 1997; Goergen and others, 2011), and it has recently been used in fuel breaks in sensitive species habitats in the northern Great Basin (fig. 23; Mark Williams, Bureau of Land Management, oral commun.).

Figure 23. Native species fuel break, northern Nevada. Photograph by Bureau of Land Management.

Fuel Break Design Considerations for Wildlife

Managing the effects of fuel breaks on wildlife might build upon historic literature of sagebrush removal for purposes of forage production for domestic grazers. In 1976, the Conservation Committee of The Wilson Ornithological Society reviewed available data on the effects of reducing sagebrush on birds and came to the following conclusion: “Sagebrush alteration should be confined to relatively small areas of 16 ha, preferably less. These should be in irregular strips which would give a maximum amount of edge for wildlife and maintain habitat diversity, and be aesthetically most pleasing. Such strips should be alternated with undisturbed strips of sagebrush about twice as wide, or more, and preferably at right angles to the prevailing wind and/or the slope of the land” (p. 169, Baker and others, 1976). Such well-intentioned recommendations to maintain the integrity of sagebrush habitat could be modified to be consistent with the science some 40 years later, especially given our improved understanding of invasive and generalist species that capitalize on habitat disturbance and edge, and the ecological benefits of protecting contiguous tracts of habitat from the irreversible impacts of wildfire.

The width of fuel breaks is an important aspect of their design when considering potential effects on wildlife. Some of the earliest work on passerine birds recommended herbicide treatments of no more than 30 m to avoid negative effects on sagebrush-dependent species such as the Brewer’s sparrows (Best, 1972). Others recommended mechanical or chemical removal of sagebrush in 100-m-wide strips with untreated strips 100–200 m wide to provide sufficient nesting habitat for sagebrush-dependent species such as sage thrashers (Castrale, 1982). Castrale (1982) also recommended retaining scattered shrubs in treated strips “because they are frequently used by all species as perches” (p. 951). McAdoo also suggested retaining at least 10 percent shrub cover in treated areas to maintain bird diversity (McAdoo and others, 1989). More recently, studies suggest treatments less than 60 m wide may be beneficial to wildlife, such as brood rearing sage-grouse, by creating attractive foraging conditions (Pyle and Crawford, 1996; Dahlgren and others, 2006). In a recent review, however, Beck and others (2012, p. 452) stated that “relying on dogmatic beliefs rather than the best available data to support management programs is premature at best for some species and irresponsible at worst for sage-grouse and possibly other species, especially given the stressors currently affecting sagebrush steppe habitats” and “more research is needed to understand the associations between sagebrush wildlife and patch size of treatments better.” For instance, recent studies that demonstrate lower songbird nest survival with a decrease of surrounding habitat (Hethcoat and Chalfoun, 2015a) suggest a likely a trade-off between implementing effective fuel breaks and habitat loss for some wildlife.

The risks of a no-action alternative are unknown, but there is mounting evidence that both fire and conversion of shrublands to invasive grasslands following repeated fires can have strong effects on animal communities, including insects (Ostoja and others, 2009; Holbrook, Pilliod, and others, 2016), mammals (Ostoja and Schupp, 2009; Holbrook, Arkle, and others, 2016; Holmes and Robinson, 2016), birds (Knick and others, 2005; Earnst and others, 2009), and reptiles (Hall and others, 2009). Hence, efforts to protect intact sagebrush may have long-term benefits to sagebrush-associated wildlife even if fuel breaks have mixed effects for individual species and populations at local scales. The lack of correlative or, more importantly, experimental studies, that assess the effects of different types of fuel breaks for most wildlife species (and at relevant spatial scales) is a severe limitation for design and implementation recommendations for fuel breaks in sagebrush ecosystems landscape. A conservative “first do no harm” approach may be warranted to restrict fuel break implementation until this research is completed, but we also recognize that, by waiting, it may be too late to act given current trends in wildfire across the Great Basin.

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Glossary

All definitions (except 'Sagebrush Focal Area') obtained from the National Wildfire Coordinating Group (2018).

Fine Fuels: Fast-drying dead or live fuels, generally characterized by a comparatively high surface area-to-volume ratio, which are less than 1/4-inch in diameter and have a timelag of 1 hour or less. These fuels (grass, leaves, needles, etc.) ignite readily and are consumed rapidly by fire when dry.

Fire Regime: Description of the patterns of fire occurrences, frequency, size, severity, and sometimes vegetation and fire effects as well, in a given area or ecosystem. A fire regime is a generalization based on fire histories at individual sites. Fire regimes can often be described as cycles because some parts of the histories usually get repeated, and the repetitions can be counted and measured, such as fire return interval.

Fireline Intensity: (1) The product of the available heat of combustion per unit of ground and the rate of spread of the fire, interpreted as the heat released per unit of time for each unit length of fire edge. The primary unit is Btu per second per foot (Btu/sec/ft) of fire front. (2) The rate of heat release per unit time per unit length of fire front. Numerically, it is the product of the heat yield, the quantity of fuel consumed in the fire front, and the rate of spread.

Fireline: The part of a containment or control line that is scraped or dug to mineral soil.

Fire Weather: Weather conditions which influence fire ignition, behavior, and suppression.

Fuel Bed: An array of fuels usually constructed with specific loading, depth, and particle size to meet experimental requirements; also, commonly used to describe the fuel composition.

Fuel Break: A natural or manmade change in fuel characteristics which affects fire behavior so that fires burning into them can be more readily controlled.

Fuel Loading: The amount of fuel present expressed quantitatively in terms of weight of fuel per unit area. This may be available fuel (consumable fuel) or total fuel and is usually dry weight.

Fuel Moisture Content: The quantity of moisture in fuel expressed as a percentage of the weight when thoroughly dried at 212 °F.

Fuel Type: An identifiable association of fuel elements of distinctive species, form, size, arrangement, or other characteristics that will cause a predictable rate of spread or resistance to control under specified weather conditions.

Sagebrush Focal Area or "SFA": The U.S. Fish and Wildlife Service has identified important landscape blocks with high breeding-population densities of greater sage-grouse (*Centrocercus urophasianus*), existing high quality sagebrush habitat, and a preponderance of Federal ownership or protected area that serves to anchor the conservation value of the landscape.

Spotting: Behavior of a fire producing sparks or embers that are carried by the wind and which start new fires beyond the zone of direct ignition by the main fire.

Appendix 1. Behave Plus Modeling Parameters

BehavePlus (version 5.0.5, Heinsch and Andrews, 2010; Andrews, 2014) was used to model potential flame lengths and rates of spread for existing and treated fuel types within the sagebrush ecosystem of the Great Basin (fig. 17). Fuel model selection (Scott and Burgan, 2005) and description for each fuel type is shown in table 1-1. All runs were completed assuming: (1) a 15 percent slope; (2) 29 °C (85 °F) air temperature; and (3) very low dead and live fuel moisture conditions[1] as defined by Scott and Burgan (2005). For each model run, midflame wind speed was stepped by 8 km/hr (5 mi/hr) increments.

Table 1-1. Fuel model section for each fuel type modeled with BehavePlus (Heinsch and Andrews, 2010; Andrews, 2014).

Fuel type	Fuel model type	Fuel model	Fuel model description
Sagebrush	Shrub	SH5	Heavy shrub load about 1.2–1.8 m (4–6 ft) tall
Sagebrush/grass	Grass-shrub	GS2	Shrubs are 03.–0.9 m (1–3 ft) tall with moderate grass load
Tall grass	Grass	GR4	Moderately coarse continuous grass about 60 cm (2 ft) tall
Short grass	Grass	GR2	Moderately coarse continuous grass about 30 cm (1 ft) tall
Green strip (bunch grass)	Grass	GR1	Grass is short and patchy
Green strip (subshrub)	Shrub	SH1	Low shrub fuel load about 30 cm (1 ft) tall and some grass may be present
Mowed	Shrub	SH1	Low shrub fuel load about 30 cm (1 ft) tall and some grass may be present
Brown strip	Non-burnable	NB	Insufficient wildland fuel to carry wildland fire under any condition

1

[1] For all scenarios, fuel moisture was set to 3, 4, 5, 30 and 60 percent for 1-hr, 10-hr, 100-hr, live herbaceous and live woody, respectively, with the exception of green strips. For the green strip model runs, live fuel moistures were low or two-thirds cured (that is, 60 and 90 percent for live herbaceous and live woody, respectively).

Appendix 2. Methods to Map and Quantify Linear Fuel Breaks (Distance and Area) in the Great Basin

Data Sources

Data were acquired from the LTDL, a legacy database of Bureau of Land Management (BLM) land treatments entered by USGS personnel, and the Vegetation Treatment Method (VTRT), a spatial record of treatments uploaded to the VTRT by BLM field offices. The LTDL and VTRT data sources were accessed on October 13, 2017, and are available at Pilliod and Welty (2013) and by contacting the BLM, respectively. These data sources are incomplete (especially pertaining to older treatments), contain duplicate records, and typically have inconsistent and non-standardized field entries for past treatment records making identification of linear fuel breaks within these datasets difficult. Thus, we used a series of automated and manual steps to conservatively identify and measure linear fuel breaks, as described below.

This is an initial assessment of fuel breaks in the Great Basin that will eventually be reconciled with other agency databases, particularly the National Fire Plan Operations and Reporting System (NFPORS) and additional information from BLM state offices.

Identifying Fuel Breaks

First, a query function was developed to search records in both the LTDL and VTRT for terms that would identify potential fuel breaks. For example, "green strip" fuel breaks were searched using many possible variations of the term (for example, "greenstrip", "green strip", etc.). These records were then flagged and standardized in a newly created field identifying them as "green strip" record. The same process of looking for variations on terms was used to identify other types of fuel breaks (for example, mowed or brown strip), as well as other potential treatment terms (and their variations) that could be potentially later verified as linear fuel breaks (for example, kochia, WUI, fuel break, fuelbreak, highway, tumbleweed) after review of descriptive fields (that is, those fields describing a fuel treatment).

Second, all identified potential records of fuel breaks were then manually assessed in the associated spatial data layers for each database using a GIS. This process was also used to display and search for long, narrow, linear features about 1 km or longer that, based on other available attributes or descriptions, were likely to be fuel breaks. While this process was somewhat subjective, nearly all additional linear fuel breaks identified using this process were apparent based on combinations of their physical features, treatment names, and treatment descriptions. For example, a treatment labeled "prescribed fire" that was long, narrow, and along a roadway would be included in the linear fuel break dataset.

Third, incorrectly identified records were removed from the initial list of potential fuel breaks (obtained from both the VTRT and LTDL), based on additional key word searches and information identified during a manual scanning of the attribute fields that suggested the primary treatment (for example, monitoring, erosion control, or fire rehabilitation) was not fuel related.

Fourth, the two resulting linear fuel break datasets derived from the VTRT and LTDL (via the process descried above) were merged into a single master linear fuel break database. The name, treatment type, treatment year, and all other relevant fields (for example, treatment descriptions) were brought into common fields created for the merge. To identify and remove duplicate entries between the two original databases, a customized python script was developed that identified features that intersected spatially and occurred in the same year and binned them into a single group for analysis. These features were examined, and if determined to be true duplicates, only one version of the record was kept. A similar process was used to identify multiple treatment entries for a given fuel break over time; such that fuel break boundaries were merged (dissolved) into one spatial record that retained the original information on the number, types, and dates of fuel breaks treatments over time. Finally, using a second visual inspection of the dataset, we removed all records that were not linear in nature (<1 km long).

Calculating the Linear Distance and Area of Linear Fuel Breaks

To calculate the total area by BLM district office treated as linear fuel breaks, we dissolved all fuel breaks into a single multipart feature and used ArcGIS Calculated Geometry to calculate the area in hectares. This value represents the estimated area of land that has been treated, not the number of actual treatment area, as some treatments overlap or represent maintenance of existing treatments. Thus, this value likely underestimates the true total land area and the actual area treated by district, due to both missing records and the repeated treatments within a given area being combined in this analysis. To calculate the linear distance treated of linear fuel breaks, the same multipart feature was used. However, because some fuel breaks consisted of treatments occurring along both sides of a road or highway (and even the median, if one existed), we used ET GeoWizards Aggregate Polygons tool (ET Spatial Techniques, 2016; http://www.ian-ko.com) in GIS with a 100 m buffer to aggregate separate polygons into a single polygon unit. The GeoWizards Calculate Centerline tool was then used to create a centerline for all remaining polygons. The total distance (in kilometers) of these centerlines was then calculated using ArcGIS Calculate Geometry to derive the total length of each line by BLM district office. This value represents the estimated length of land within positively identified linear fuel breaks and not the total number of treatment kilometers, as some treatments overlap or were maintained via two or more treatments over time. Moreover, many linear kilometers are likely not accounted for due to missing records (especially older treatments) in the two databases assessed (VTRT and LTDL).

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