Chapter M. Conifer Expansion

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Executive Summary

Coniferous trees, principally juniper (*Juniperus* spp.) and pinyon pine (*Pinus* spp.), have increased considerably in cover and density in the western United States since European settlement with wide ranging consequences for sagebrush (*Artemisia* spp.) ecosystems. A continuum of vegetation types exists across the region, from conifer-encroached shrublands to persistent pinyon-juniper woodlands and savannas. This chapter focuses on the issue of conifer expansion into sagebrush shrublands and ensuing woodland succession, not the infill of persistent woodlands and savannas. Detrimental effects of conifer expansion on sagebrush ecosystem vegetation composition and productivity, wildlife, water and nutrient cycles, carbon storage, resilience to fire, and resistance to cheatgrass (*Bromus tectorum*) invasion are well-documented. Unprecedented partnerships have formed in recent years to address conifer expansion impacts across ownerships in the sagebrush biome. While significant conifer reduction has occurred in some strategic priority areas, the overall proportion of conifer being reduced through management and wildfire across the region remains relatively small.

Conifer removal is one of the few restoration practices known to be effective for restoring and maintaining a variety of sagebrush ecosystem functions and sagebrush-dependent plant and animal communities, but the degree of efficacy varies by treatment method, pretreatment site type and ecological conditions, location of treatment, follow up treatments, and posttreatment management. Understanding ecological site and stand characteristics is critical when evaluating conifer cover changes and determining appropriate management responses. Carefully crafted management prescriptions across the spectrum of woodland to shrubland—based on ecological site potential and historical stand conditions and dynamics—are needed to address all species habitat requirements at a whole watershed scale in the appropriate places on the landscape. A nuanced and holistic approach is likely necessary to balance multispecies habitat needs across the spectrum of woodland to shrubland.

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Introduction

Around the world, native trees are expanding into previously grass- and shrub-dominated systems (Nackley and others, 2017) contributing to the loss of rangelands (fig. M1). In the sagebrush (*Artemisia* spp.) biome, coniferous trees—principally pinyon pine (*Pinus* spp.) and juniper (*Juniperus* spp.; hereafter pinyon-juniper)—have increased considerably in both cover and density and are considered a persistent threat to sagebrush communities in some areas in the sagebrush biome (Chambers and others, 2017a). The focus of this chapter is primarily on factors that have contributed and continue to contribute to conifer expansion into sagebrush shrublands and ensuing woodland succession, along with the impacts of this expansion on ecosystem processes and wildlife. This chapter does not focus on the infill of persistent woodlands and savannas.

Nature and Extent of Conifer Expansion

Pinyon-juniper woodlands occur over an extensive area (greater than [>] 40 million hectares [ha; >100 million acres]) and in a wide variety of environmental conditions across the western United States, but three fundamentally different vegetation types have been described based on canopy structure and understory conditions: (1) persistent woodlands, (2) savannas, and (3) wooded shrublands (fig. M3; Romme and others, 2009). Increases in tree cover and density have resulted in both infill of persistent woodlands, savannas, and wooded shrublands leading to stand closure, as well as tree expansion into sagebrush ecosystems that historically did not support trees. This expansion has resulted in land cover type conversion from shrubland to woodland. Conifer expansion has been especially pronounced in the Great Basin where tree-ring analyses suggest a twofold to sixfold increase in woodlands since European settlement (Miller, R.F., and others, 2008). However, the extent of pinyon-juniper increase varies across the biome and effects are more localized in some ecoregions such as the Colorado Plateau (see reviews in Romme and others, 2009; Miller and others, 2019).

In parts of the sagebrush biome, other conifer species are also expanding locally, such as fir (*Abies* spp.) and ponderosa pine (*Pinus ponderosa*). Ninety percent of tree expansion is estimated to have occurred in sagebrush ecosystems (Miller and others, 2011). Although it is difficult to quantify expansion without site-specific data, remotely-sensed,

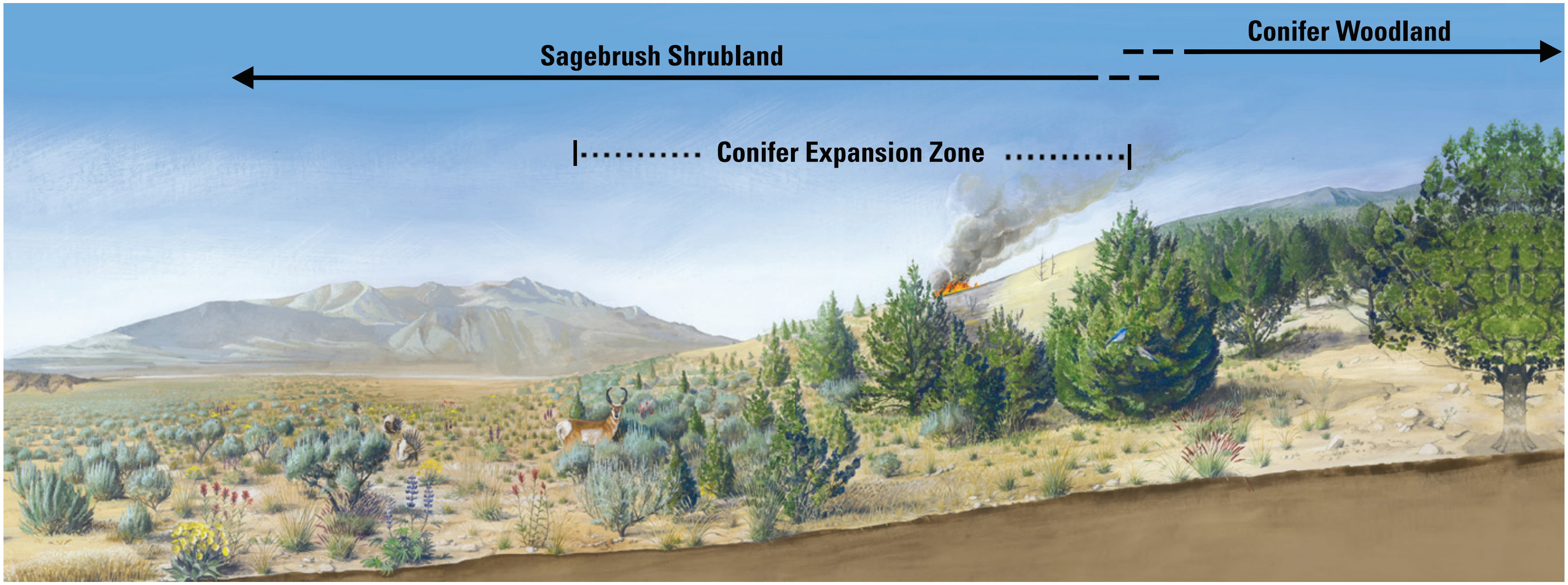

Figure M1. Illustration depicting the shrubland-to-woodland continuum. Adapted from Johnson and others (2019a, b).

high-resolution mapping of conifer cover within the occupied range of greater sage-grouse (*Centrocercus urophasianus*) found that over one-quarter of mapped lands already support >1 percent tree cover (Falkowski and others, 2017). Newly-available land cover data showing tree cover change in recent decades (1999–2018) further illustrate the patterns and extent of tree cover increase in the sagebrush biome (fig. M2).

Woodland succession progresses in predictable phases that have been described with associated changes in understory vegetation composition (fig. M4): Phase I—trees are present, but shrubs and herbs are the dominant vegetation influencing ecological processes on the site; Phase II—trees are codominant with shrubs and herbaceous vegetation, and all three vegetation layers influence ecological processes; and Phase III—trees are the dominant vegetation on the site and the primary plant layer influencing ecological processes on the site (from Miller and others, 2005). A study in the Great Basin found most sites were still in Phases I and II, but 75 percent of affected shrublands were expected to transition to Phase III woodlands in the next 30–50 years (Miller, R.F., and others, 2008).

Remotely sensed data provides insight into the extent and nature of recent conifer expansion and infill. Across the Great Basin, between 2000 and 2016, the amount of area considered forested by pinyon-juniper (>10 percent tree cover) increased >4,600 square kilometers [1.1 million acres] at an overall rate of 0.46 percent per year (Filippelli and others, 2020). Widespread infilling also occurred over this time period with 80 percent of documented increases in pinyon-juniper aboveground biomass because of infilling of existing woodlands (Filippelli and

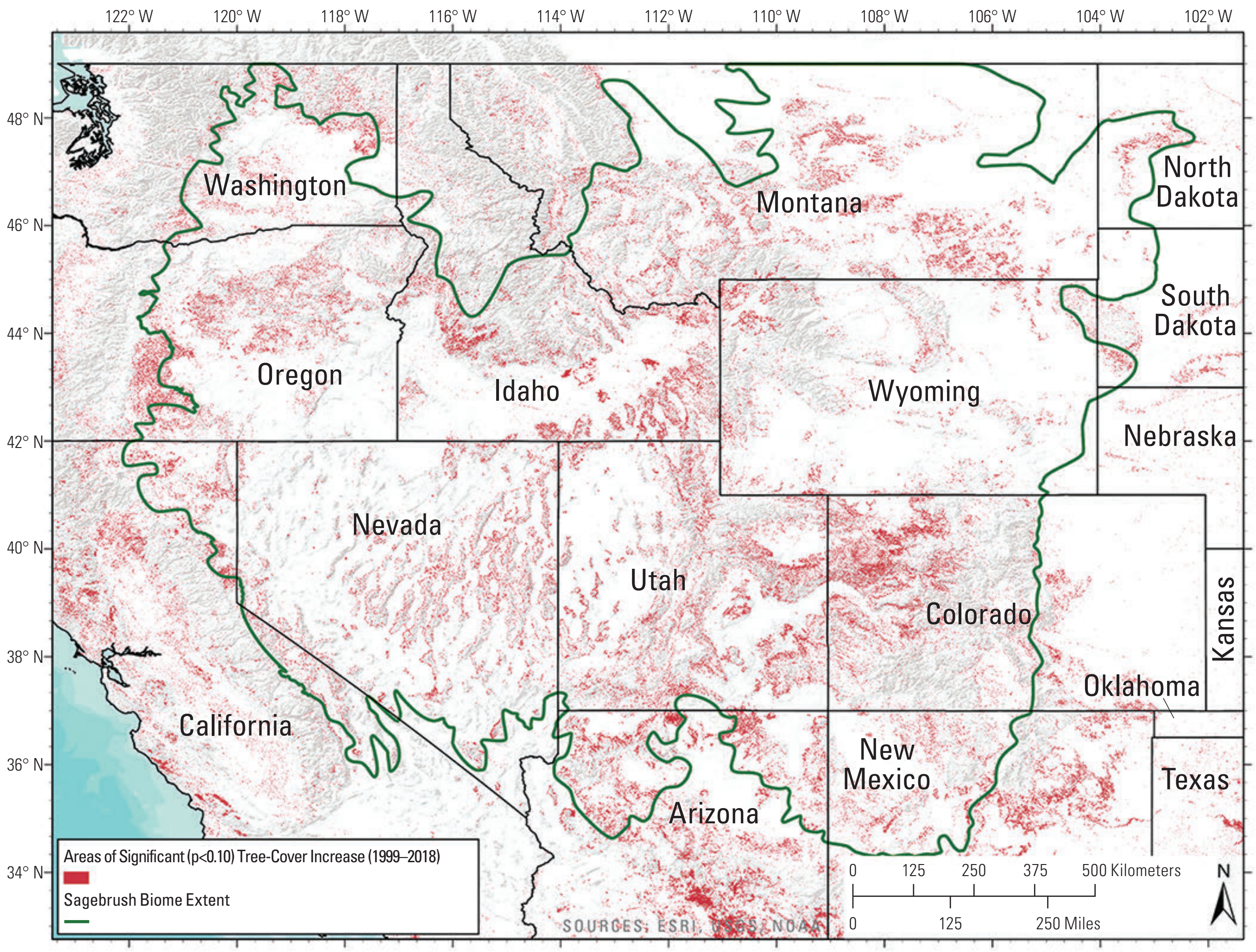

Figure M2. Rangelands experiencing a significant (p<0.10) increase in tree cover in the western United States (1999–2018). Tree cover includes all tree species, not just conifers. Tree cover from Jones, M.O., and others (2018) assessed for rangelands only, as defined by Reeves and Mitchell (2011). p, probability; <, less than.

others, 2020). Mapping of conifer cover within the occupied range of greater sage-grouse range revealed over one-quarter of mapped lands already support >1 percent tree cover (Falkowski and others, 2017). Newly available land-cover data showing changes to tree cover in recent decades (1999–2018) across rangelands in the sagebrush biome further illustrate the patterns and extent of tree-cover increase (fig. M2; Jones, M.O., and others, 2018).

Causes of recent increases in tree expansion and infill are not fully understood but are often attributed to climate, grazing, and reduced fire occurrence (Miller and others, 2019). However, there is no scientific consensus on the relative importance of these factors (Baker, 2011; Miller and others, 2011). Because a continuum of vegetation types exist across the biome—from sagebrush shrublands to pinyon-juniper woodlands and conifer forests—understanding ecological site and stand characteristics is critical when evaluating conifer cover changes and determining appropriate management responses (Floyd and Romme, 2012; Miller and others, 2014a).

Impact on Sagebrush Communities, Ecosystem Processes, and Wildlife Communities

Where conifer and sagebrush communities interface, an increasing dominance of trees results in the decline of perennial grasses (Tausch and West, 1995; Schaefer and others, 2003; Roundy and others, 2014a), perennial forbs (Bates, 2005; Dhaemers, 2006), and herbaceous productivity and species richness (Miller and others, 2000). Declines occur particularly on warm and dry sites and on sites with shallow, root-restrictive layers in the soil profile (Miller and others, 2005). Increasing woodland cover can affect snow distribution and soil water availability, which in turn shortens the growing season and the duration of water availability (Bates and others, 2000; Roundy and others, 2014a; Kormos and others, 2017). Conversion of shrubland to woodland has also been shown to influence infiltration, runoff, erosion, and sediment loads (Pierson and others, 2007, 2010; Petersen and Stringham 2009; Miller and others, 2013). Susceptibility to erosion

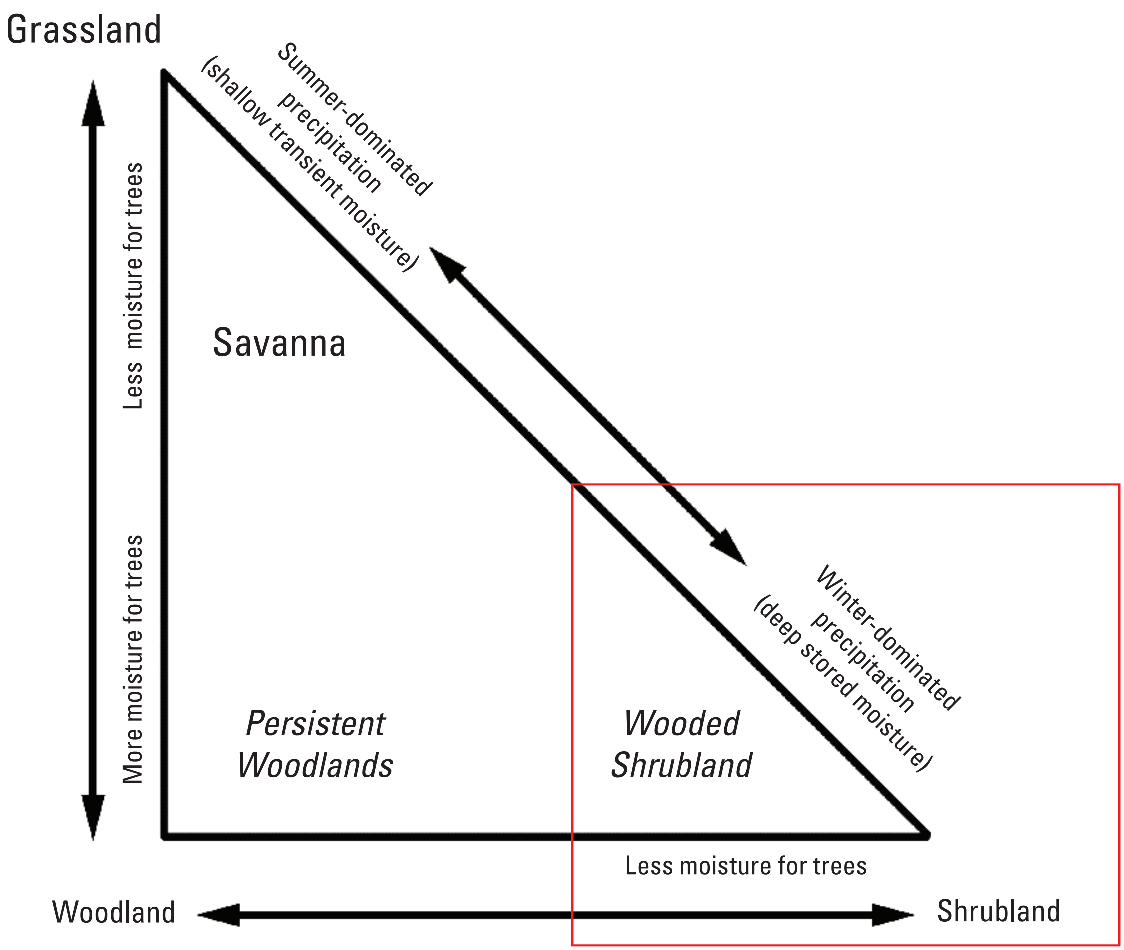

Figure M3. General framework for the pinyon (*Pinus* spp.)-juniper (*Juniperus* spp.) woodland-to-shrubland continuum along a gradient of soil moisture and seasonality of precipitation (adapted from Romme and others, 2009). Expansion of woodlands into former sagebrush (*Artemisia* spp.) shrublands (red box) is the primary focus of this strategy.

Phase I:

- Shrub and herbaceous dominance
- Active tree recruitment
- Low cone production

Phase II:

- Tree, shrub, and herbaceous codominance
- Active tree recruitment
- Cone production moderate to high
- Shrubs intact to thinning

Phase III:

- Tree dominance
- Shrubs >75% absent
- Herbaceous intact (cool-moist sites) to depleted (warm-dry sites) with increased loss on sites with shallow soils
- Limited tree recruitment
- Cone production low to none

Figure M4. Phases of woodland succession and observable field characteristics (adapted from Maestas and others, 2016). >, greater than; %, percent.

following tree increases varies with ecological site potential, as determined by climate, geomorphology, soil erodibility, and ground cover (Davenport and others, 1998).

The carbon cycle changes with increasing tree cover in shrub- and grass-dominated ecosystems. In sagebrush ecosystems, most carbon is stored below ground in the roots (Rau and others, 2011). Conifer expansion into sagebrush ecosystems increases aboveground carbon storage owing to the large increase in biomass, but effects on below-ground carbon storage are poorly understood (Rau and others, 2011). Because there is a larger part of the carbon pool above ground, it is susceptible to volatilization during high-intensity fires (Rau and others, 2009, 2011).

These alterations can reduce sagebrush ecosystem resilience to disturbances and resistance to invasive plants and increase susceptibility to shifts to novel ecosystem states (Chambers and others, 2007, 2014c; Miller and others, 2013). Increases in woody fuel loads because of conifer expansion and infilling heighten the risk of high-severity crown fires, especially during extreme fire weather. High-severity fires can increase the potential for ecosystem conversion to an alternative state dominated by invasive annual grasses (that is, cheatgrass [*Bromus tectorum*] and medusahead rye [*Taeniatherum caput-medusae*]) in warmer and drier areas with insufficient perennial herbaceous species to promote recovery (Miller and others, 2014a; Chambers and others, 2014c). Excessive soil loss on steeper slopes can also result in conversion to an eroded state that is largely irreversible (Chambers and others, 2014c). These state shifts can reduce ecosystem function at landscape scales by fragmenting intact sagebrush shrublands and impairing movements and reproductive processes necessary to sustain plants and wildlife.

Both sagebrush- and woodland-dependent wildlife are affected by increases in conifer cover in the sagebrush biome. Increases in conifer canopy cover result in nonlinear declines in sagebrush cover (Miller and others, 2000; Roundy and others, 2014a), which directly reduces the amount of available food and cover for sagebrush-dependent species. Even before direct habitat loss occurs, sage-grouse avoid or are negatively associated with conifer cover during all life stages (that is, nesting, brood-rearing, and wintering; Doherty and others, 2008, 2010a, 2016; Atamian and others, 2010; Casazza and others, 2011; Dinkins and others, 2014a; Walker and others, 2016; Severson and others, 2017a). Local sage-grouse distribution and demographic rates are impacted with low amounts of conifer present (approximately 1.5–4 percent canopy cover; Baruch-Mordo and others, 2013; Coates and others, 2017a).

No leks remained active when conifer canopy exceeded 4 percent within 1 kilometer (km; 0.6 mile [mi]) of the lek in an Oregon study (Baruch-Mordo and others, 2013). Also, most active leks averaged <1 percent conifer cover within 5 km (3.1 mi) in the western part of the range (Knick and others, 2013). Sage-grouse movement across conifer-expansion areas may be more rapid than across areas without conifer expansion. This may result in lower survival rates among sage-grouse because of potential increased exposure to predators (Prochazka and others, 2017). As a result, the perceived increased risk of predation may cause sage-grouse to avoid habitats with conifer expansion. Higher-elevation sites with early-phase woodland expansion (>2 percent conifer cover) that provide desirable food sources may function as ecological traps, likely because of increased predation from raptors (Coates and others, 2017a).

Phase III encroachment of Douglas-fir (*Pseudotsuga menziesii*) at high-elevation mountain big sagebrush sites in Montana and Idaho reduced availability of forage (big sagebrush [*A. tridentata*] and forbs and grasses), decreased cover quality, and increased predation risk for pygmy rabbits (*Brachyiagus idahoensis*) relative to reference plots (Woods and others, 2013). Shifts in small mammal species composition have been documented, including a decrease in sagebrush specialists (for example, Great Basin pocket mouse [*Perognathus mollipilosus*]) and an increase in woodland specialists (for example, pinyon mouse [*Peromyscus truei*]), in association with increasing conifer woodlands (Rickart and others, 2008).

Long-term trends in Breeding Bird Survey data across the region show similar patterns for songbirds, including decreases in sagebrush species and increases in woodland species, with the exception of pinyon jay (*Gymnorhinus cyanocephalus*; Sauer and others, 2017). Pinyon jay declines may reflect, in part, changes in habitat structure and quality and pinyon pine productivity (Boone and others, 2018) and mortality because of drought (Fair and others, 2018) in persistent pinyon-juniper woodlands and savannas. Big game, such as mule deer (*Odocoileus hemionus*), are likely affected by changes in forage availability and quality that occur as woodland succession advances. Experimental and observational research has shown that nutrition on winter range can limit mule deer survival and population growth (Baker and Hobbs, 1985; Peterson and Messmer, 2007; Bishop and others, 2009). Suppression of forage by trees may explain why the amount of pinyon-juniper in the annual mule deer home range was negatively related to—and explained 26 percent of variation in—ingesta-free body fat in female mule deer in New Mexico (Bender and others, 2007).

Impact on Human Resource Needs and Values

Domestic livestock grazing is the most widespread, contemporary human use of sagebrush ecosystems impacted by woodland expansion. Perennial herbaceous cover is halved when pinyon-juniper cover reaches 40 percent (fig. M5; Roundy and others, 2014a), directly reducing available forage for grazing. In a model of juniper encroachment impacts on ranch economics, McClain (2013) showed that transitioning from Phase I to Phase III reduced available forage, thereby limiting the number of livestock that could be sustained and reducing ranch income by a third. Removing juniper from shrublands increased the livestock carrying capacity by nearly 10 times, which provides added management flexibility as the forage base improves (Bates and others, 2005). Observational studies suggest potential interactive effects among livestock grazing, tree expansion, and livestock forage. In those studies that compared adjacent grazed and historically ungrazed areas, pinyon-juniper densities, canopy cover, or basal area were greater in the grazed than ungrazed pastures (Madany and West, 1983; Guenther and others, 2004; Soulé and others, 2004; Shinneman and Baker, 2009). Impacts of increasing conifer cover on other human resource needs and values, such as use by indigenous peoples and recreation, are likely occurring but not well studied.

Current Efforts to Address Conifer Expansion

Conifer removal has been occurring in the sagebrush biome for decades, but contemporary management differs in its primary objectives, approach, and scale. Early conifer removal efforts (for example, 1950s–70s) were largely done to increase forage production, improve watershed conditions, and enhance deer winter range (Johnson, 1967; Terrel and Spillett, 1975). Because little distinction was made between ecological sites that supported presettlement or newly expanded woodlands, and soil and understory vegetation conditions were seldom evaluated prior to treatment, the response to tree removal was not always positive (O'Rourke and Ogden, 1969; Clary, 1971, 1974). Current primary objectives for most treatments focus on fuels reduction, shrub- and grassland-dependent wildlife habitat, watershed function, and sagebrush ecosystem restoration (Miller and others, 2019). Treatment planning has become more nuanced and now often incorporates ecological site potential, ecosystem resilience to disturbance and resistance to invasive plants, and phase of woodland succession into management decisions (Tausch and others, 2009; Miller and others, 2014a).

In recent years, sage-grouse have become a primary driver of landscape-scale conifer removal in the sagebrush biome with the threat of sage-grouse species being listed under the Endangered Species Act of 1973 (16 U.S.C. 1531 et seq.) and heightened awareness of conifer expansion effects on sagebrush habitats (U.S. Department of the Interior, 2015c; Miller, R.F., and others, 2017). Research indicating sage-grouse sensitivity to very low levels of conifer (approximately 4 percent; Baruch-Mordo and others, 2013) spurred a shift in treatment approach to prioritizing tree removal in expansion areas that are still in early phases of succession (Phases I–II) over areas of dense woodlands (Phase III; fig. M6). Additional research on understory response to woodland succession and treatment further emphasized the benefits of targeting areas of early-phase tree expansion for preserving ecosystem resilience and resistance (Roundy and others, 2014a; Chambers and others, 2014c).

Owing to sage-grouse space requirements and limited restoration resources, clustered treatments across ownership boundaries that expand upon large sage-grouse strongholds have become preferred over small, scattered treatments that

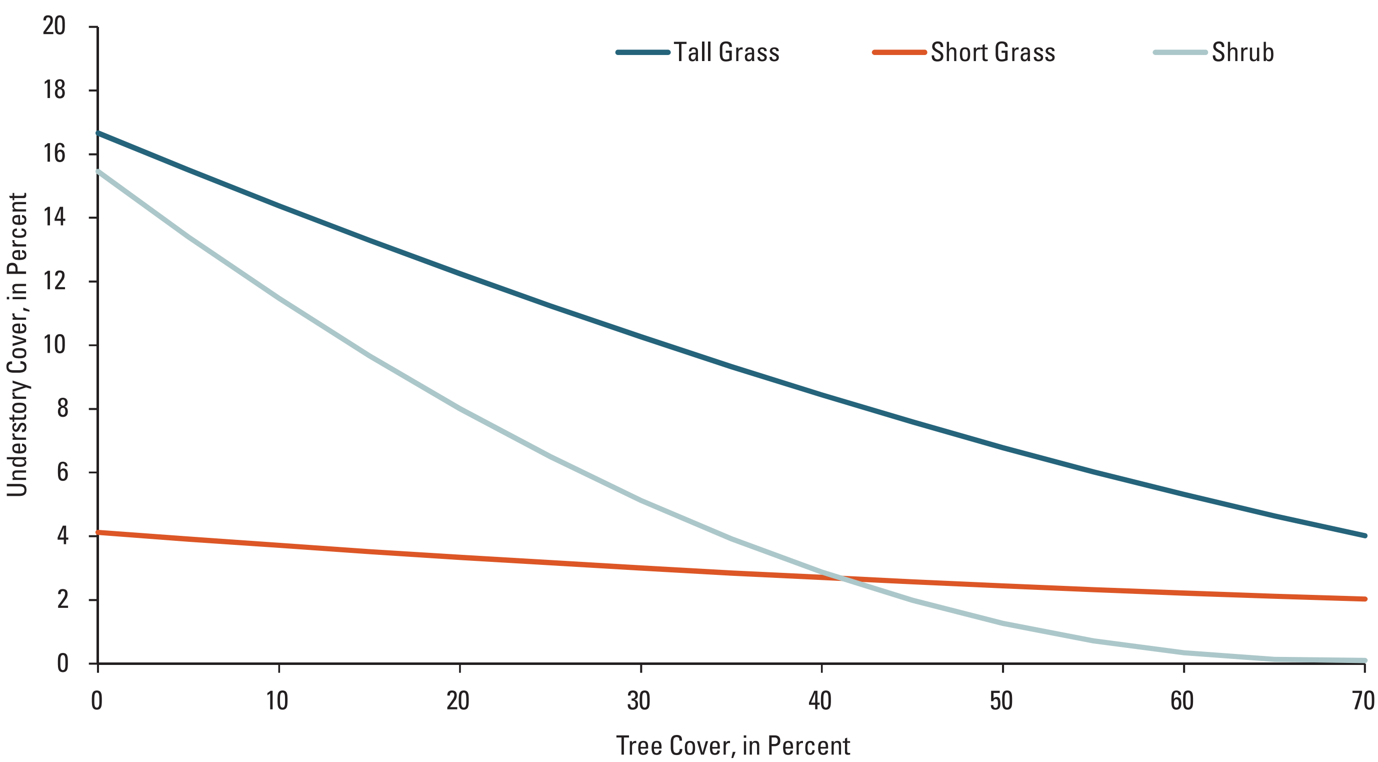

Figure M5. Effects of increasing tree cover on understory cover of shrubs and grasses on 11 sites measured across the Great Basin (Roundy and others, 2014a; adapted from Maestas and others, 2016).

may not yield habitat benefits (Severson and others, 2017b). Use of selective mechanical techniques for tree removal (for example, chainsaw cutting, mastication) has become more prevalent as a means to preserve understory shrubs and herbaceous vegetation and reduce opportunities for invasive annual grasses, although prescribed fire remains an important tool for long-term management under certain conditions (Boyd and others, 2017b).

Extensive efforts to address conifer expansion are ongoing across the sagebrush biome for a variety of land-management objectives. Some larger regional efforts highlighted here provide a glimpse of ongoing management. Since 2010, a diverse coalition of partners have greatly accelerated conifer removal efforts to improve sage-grouse habitat (U.S. Department of the Interior, 2015c). The U.S. Department of Agriculture, Natural Resources Conservation Service launched the Sage Grouse Initiative (SGI) to accelerate voluntary and incentive-based species recovery and proactive ecosystem restoration (Natural Resources Conservation Service, 2015). Private landowners through SGI treated over 250,000 hectares (ha; 617,000 acres) of conifer expansion between 2010 and 2017.

States, other Federal agencies, and private organizations are also involved in woodland management for sage-grouse, fuels reduction, and watershed improvement. For example, from 2005 to 2018, the State of Utah's Watershed Restoration Initiative partnership completed 459,120 hectares (ha; 1,134,472 acres) of pinyon-juniper removal, averaging roughly 35,000 ha (87,000 acres) of treatment per year (fig. M7; Utah Watershed Restoration Initiative, 2019). The U.S. Department of the Interior, Bureau of Land Management (BLM) has also accelerated treatments addressing conifer expansion on public lands mainly because of efforts related to sage-grouse habitat improvement and fire-risk reduction. From 2013 to 2017, the annual rate of conifer removal increased nearly fivefold (2013: 21,606 ha [53,390 acres]; 2017: 101,636 ha [251,137 acres]) for a total of 284,266 ha (702,412 acres) of conifer treated in the sagebrush biome. Conifer treatment continues to be a priority for BLM in the region.

A 2019 study used remote sensing to assess conifer reductions across a large part of the sage-grouse range. The study encompassed >45.7 million ha (>113 million acres) of sagebrush ecosystems over a 4–6-year period coinciding with large-scale treatment efforts (fig. M8; Reinhardt and others, 2020). Of the total estimated area experiencing conifer reduction during this timeframe, 87 percent occurred in three States in the Great Basin: Nevada, Oregon, and Utah. Over half (53 percent) of the conifer reduction occurred in Priority Areas for Conservation (PACs) or in sage-grouse strongholds where managers have been working to maintain large, intact sagebrush-dominated landscapes (U.S. Fish and Wildlife Service, 2013). Mapping confirmed that a diverse array of landowners and managers are contributing to conifer reduction, with the majority of treatments occurring on BLM-administered lands. Accelerated restoration efforts in recent years have raised some concerns about loss of tree cover. However, despite recent concerted efforts, conifer reductions owing to management and wildfire have occurred on just 1.6 percent of the total area supporting trees, providing critical context for local and regional discussions (Reinhardt and others, 2020). In fact, current conifer-reduction efforts may just be keeping pace with the estimated expansion rates of approximately 0.4 to 1.5 percent per year (Sankey and Germino, 2008).

Climate change and wildfire will likely continue affecting conifer expansion in some areas. The total area of pinyon-juniper land cover types that burned in the sagebrush biome increased significantly from 1984 through 2013, except in the Central Basin and Range (Board and others, 2018). More than one-third of the total conifer reduction in studied sage-grouse range was attributed to wildfire (fig. M8; Reinhardt and others, 2020). Climate change may also affect tree expansion as localized areas of die-off have resulted following recent drought in the Central Basin and Range (Greenwood and Weisberg, 2008; Flake, 2016; Flake and Weisberg, 2019) and the Southwest (Fair and others, 2018). Pinyon-juniper woodlands are projected to contract in warmer and drier parts of the biome (Rehfeldt and others, 2012). Conversely, increasing atmospheric carbon dioxide may accelerate growth rates of western juniper (*J. occidentalis*) where soil water and nutrients are not limiting, potentially increasing rate of woodland expansion in some areas (Knapp and others, 2001).

Efficacy of Tree Removal at Restoring Ecosystem Function and Plant and Animal Communities

The efficacy of conifer removal for restoring and maintaining sagebrush ecosystem function, and plant and animal communities, has been well-documented in recent years (see special issues summarized in McIver and others, 2014; Miller, R.H., and others, 2017; Miller and others, 2019). As with all restoration treatments, the degree of efficacy varies depending on treatment method (for example, mechanical, fire), pretreatment site type and ecological conditions, spatial location of treatment, follow up treatments (for example, seeding, weed control), and posttreatment management. However, some generalized outcomes emerging from the literature are summarized in this chapter.

Conifer removal generally results in more herbaceous cover and biomass; twofold to twentyfold increases have been recorded after tree removal (Young and others, 1985; Clary, 1987; Vaitkus and Eddleman, 1987; Rose and Eddleman, 1994; Bates and others, 2000; Stephens and others, 2016; Bates and others, 2017). Perennial herbaceous vegetation is critical to site resilience to disturbance and resistance to invasive annuals (Chambers and others, 2014c; Miller and others, 2014a; Roundy and others, 2014a). However, both perennial herbaceous vegetation and cheatgrass can increase in response to rises in available nutrients and water following conifer removal. Increases in cheatgrass can be

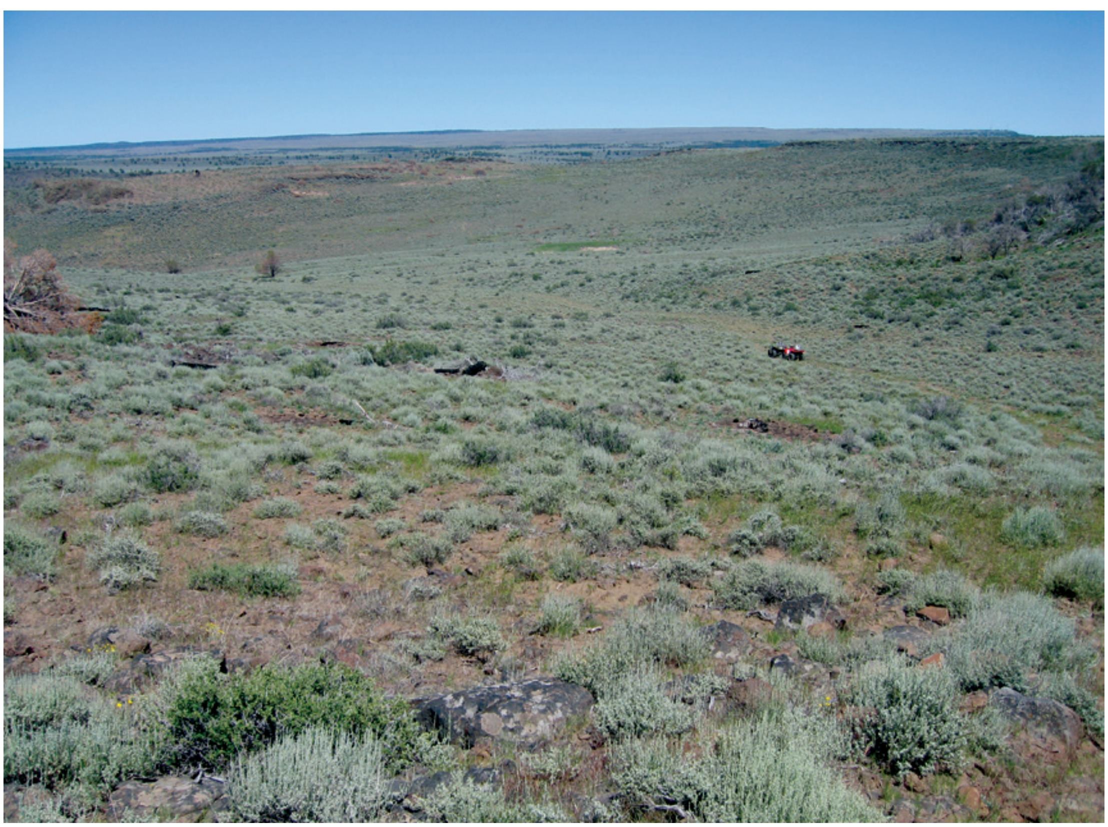

Figure M6. Example of conifer removal in the sagebrush (*Artemisia* spp.) ecosystem in Oregon. The South Warner Project Area before (2008, top) and after (2015, bottom) hand felling of Phase II stands of juniper (*Juniperus* spp.) trees in 2013 (photograph by Todd Forbes, U.S. Department of the Interior, Bureau of Land Management).

especially problematic on relatively warm and dry sites with low initial cover of perennial herbaceous species. In areas susceptible to invasion, treatments should be located where sufficient perennial species occur for site recovery. Alternately, to prevent invasion, land managers should plan follow up treatments such as invasive plant control and seeding (Chambers and others, 2014c; Miller and others, 2014a; Roundy and others, 2014a).

Treatment effects depend on site type; the initial abundance and composition of trees, shrubs, and herbaceous plants; and the type of treatment. Pretreatment tree cover is usually negatively related to pretreatment herbaceous cover, and this relationship influences the potential for posttreatment increases in perennial herbaceous species (Roundy and others, 2014a; Chambers and others, 2014c; Williams and others, 2017). On sites with low to moderate resistance to invasive annual grasses, increases in cheatgrass and other annual exotics are typically greater on sites that have more pretreatment tree cover (Chambers and others, 2014c). On these sites, perennial native herbaceous cover of at least 20 percent appears necessary to prevent a large increase in cheatgrass and other annual invasive plants after treatment (Chambers and others, 2014c). On cooler and moister sites with relatively high resistance, perennial native herbaceous cover may be less important for preventing dominance by annual invaders owing to lower climate suitability (Chambers and others, 2007). However, adequate cover of perennial herbaceous species and root-sprouting shrubs is still necessary for soil stabilization and overall site recovery (Miller and others, 2014a). Sites differ in topography, soil characteristics, and productivity as well as resistance to invaders, and all of these factors should be considered when selecting sites for treatment and evaluating indicators of potential site recovery.

Treatments performed on sites in early stages of woodland succession (Phases I and II) are typically more effective at maintaining the desired sagebrush ecosystem state,

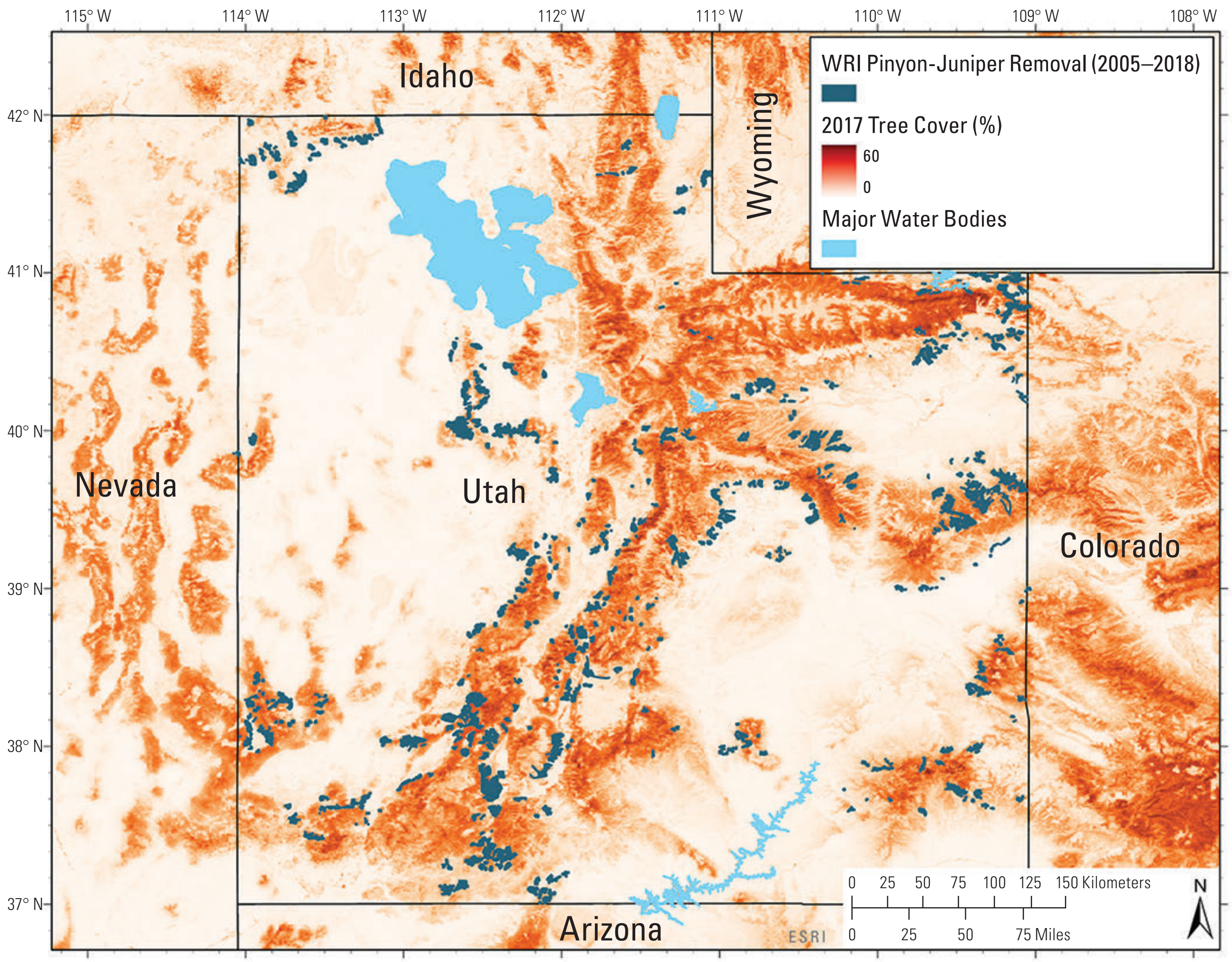

Figure M7. Location of pinyon pine (*Pinus* spp.) and juniper (*Juniperus* spp.) removal conducted through Utah's Watershed Restoration Initiative (WRI) from 2005 to 2018 overlaid on tree cover (from Jones, M.O., and others, 2018). Most pinyon-juniper removal is located in conifer expansion areas along the shrubland-woodland ecotone. %, percent.

but plant composition and structure as well as the treatment method may affect outcomes. Mechanical treatments (for example, cutting, mastication) typically retain most understory shrubs and perennial herbaceous vegetation, whereas prescribed fire often removes nonsprouting shrubs and may result in initial decreases in perennial herbaceous vegetation depending on fire severity (Roundy and others, 2014a; Williams and others, 2017). Higher pretreatment abundance of woody fuels can increase fire severity, which negatively impacts the abundance of perennial grasses and the ability to resist invasive annual grasses posttreatment (Williams and others, 2017). For example, prescribed fire in Phase III plant communities can create a high risk of postburn annual grass invasion (Bates and others, 2013) necessitating follow up weed control and seeding.

On warm and dry sites with lower resilience and resistance, mechanical treatment of Phase I and II conifer expansion is recommended to increase the probability of recovery to a desirable state (Chambers and others, 2014c; Williams and others, 2017). On cool and moist sites with higher resilience and resistance, prescribed fire can be an effective means to restore sagebrush shrublands and provide a longer treatment lifespan before the return of trees, depending on treatment objectives and site conditions (Bates and others, 2017; Boyd and others, 2017b). Regardless of the method used for conifer reduction, posttreatment abundance of deep-rooted perennial bunchgrasses and exotic annual grass response will be important determinants of recovery (Condon and others, 2011).

When strategically targeted and properly designed, conifer removal is one of the few restoration practices with documented efficacy for benefiting sagebrush-dependent birds. In a before-after, control-impact (BACI) study in southern Oregon, nesting habitat suitability for sage-grouse increased after tree removal

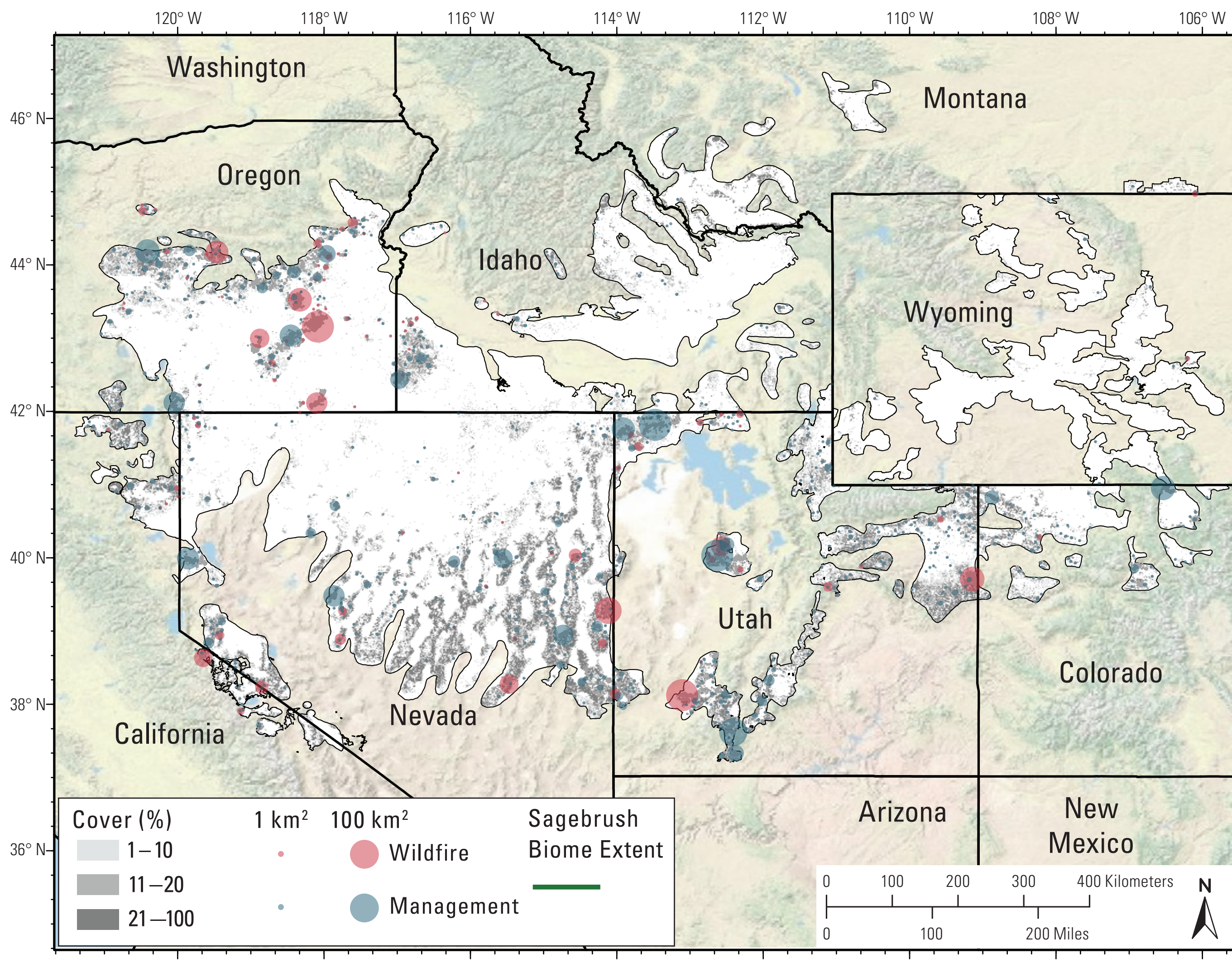

Figure M8. Locations of predicted conifer reductions owing to management and wildfire in occupied greater sage-grouse (*Centrocercus urophasianus*) range (adapted from Reinhardt and others, 2020). White/grey areas on the image represent mapped tree cover, and the green boundary represents the extent of the sagebrush (*Artemisia* spp.) biome. %, percent; km², square kilometer.

(Severson and others, 2017c), and nesting females were quick to use restored habitats (Severson and others, 2017b). Trends in annual female survival (an increase of 6.6 percent) and nest survival (an increase of 18.8 percent) were higher in the juniper-removal treatment area relative to the control landscape (Severson and others, 2017d). Population benefits accrue with time, and integrated population modeling shows that conifer-removal treatments result in an approximately 12 percent increase in population growth rates in treated landscapes relative to untreated control landscapes (fig. M9; Olsen, 2019). In northwest Utah, female sage-grouse using restored habitats were more likely to raise a successful brood (Sandford and others, 2017). Taken together, studies show that conifer removal can increase habitat availability for nesting and brood-rearing sage-grouse with population-level benefits. The potential efficacy of conifer removal for Gunnison sage-grouse (*C. minimus*) was modeled, with results showing large-scale, coordinated conservation efforts to remove conifers, while ensuring treatment areas regenerate into sagebrush-dominated cover, could increase breeding habitat by 46–69 percent (Doherty and others, 2018).

Shrub and grassland songbirds also benefit from treatments. Brewer's sparrow (*Spizella breweri*) increased following tree removal on the Colorado Plateau (Crow and van Riper, 2010). Studies in other locations suggest that removing juniper benefits the species (O'Meara and others, 1981; Noson and others, 2006; Reinkensmeyer and others, 2007; Knick and others, 2014a). In the Great Basin, abundances of Brewer's sparrow, green-tailed towhee (*Pipilo chlorurus*), and vesper sparrow (*Pooecetes gramineus*) more than doubled following mechanical conifer removal (Holmes and others, 2017), and 85 percent of conifer removal conducted through SGI coincided with high abundance centers for Brewer's sparrow (Donnelly and others, 2017). Sagebrush sparrow (*Artemisiospiza nevadensis*) density was either increased or not affected by conifer removal (Reinkensmeyer, 2000; Woolley and Heath, 2006; Reinkensmeyer and others, 2007; Knick and others, 2014a). Though sage thrashers (*Oreoscoptes montanus*) may use small trees, pinyon-juniper encroachment into sagebrush stands does not benefit the birds, and the removal of encroaching junipers improves habitat (Reinkensmeyer, 2000; Noson and others, 2006; Reinkensmeyer and others, 2007). Findings illustrate that conifer removal performed for sage-grouse that retained shrub cover can result in immediate benefits for other sagebrush birds of high conservation concern, but treatment technique and location, ecological site type, and pretreatment understory vegetation matter (Knick and others, 2014a; Miller and others, 2014a).

Benefits of conifer removal have also been recorded for sagebrush mammals of management concern. In an 11-year BACI study, tree removal in Phase II pinyon-juniper areas maintained small mammal densities more effectively than untreated control areas (Hamilton and others, 2019). Several studies have experimentally tested for effects of pinyon-juniper treatments (mechanical or chemical thinning) on mule deer habitat use or demography (Bender and others, 2013; Bergman and others, 2014a, b, 2015). In Colorado, overwinter survival of mule deer fawns was 15 percent higher in mechanically treated areas with follow up seeding and weed control than in untreated control areas (Bergman and others, 2014a). Furthermore, mule deer use of mechanically treated areas was positively related to body size and condition of adult females in New Mexico (Bender and others, 2013) and Colorado (Bergman and others, 2014b), though the significance of these effects was marginal.

Depending upon site conditions and conifer phase, conifer removal can extend the duration of soil water availability by up to 26 days (fig. M10; Roundy and others, 2014b), providing added moisture for plants during critical growth periods. Conifer cutting also extended the period of active growth for understory plants by up to 6 weeks because of greater soil water availability (Bates and others, 2000). At the watershed scale, water delivery can be delayed by an average of 9 days in sagebrush shrublands compared with juniper-dominated systems (Kormos and others, 2017). However, a synthesis across the western United States found that the initial ecohydrologic and erosion impacts of tree reduction on pinyon-juniper woodlands by fire, mechanical tree removal, or drought depend largely on the degree to which perturbations alter vegetation and ground cover structure, initial conditions, and inherent site attributes (Williams and others, 2018). Overall, the literature is inconclusive regarding tree reduction impacts on watershed-scale changes in groundwater and streamflow (Williams and others, 2018). Conifer removal may provide the added ecosystem service of improved water capture, storage, and delayed release in some semiarid ecosystems (which will become increasingly important with warming climate conditions), but results are variable and site specific.

Potential Impact of Conifer Removal on Sagebrush Species

Negative effects of conifer removal are typically associated with woodland-affiliated species and not with sagebrush species. A literature review by Bombaci and Pejchar (2016) found no consistent positive or negative trend in the overall effects of pinyon-juniper woodland reduction on wildlife. However, trends were apparent when analyzed by taxonomic and functional groups and treatment types (Bombaci and Pejchar, 2016). Pinyon-juniper treatments were generally benign or beneficial for sagebrush and shrub-grassland obligate species while generally benign or negative for woodland and woodland-shrubland species. Research was limited in some cases and especially so for nontarget species like invertebrates and reptiles (Bombaci and Pejchar, 2016). Few studies showing positive responses to conifer reduction by sagebrush and shrub-grassland species were found, but the authors noted that most studies were short-term, and species may not respond until several years' posttreatment (Bombaci and Pejchar, 2016).

Detrimental impacts of treatments on sagebrush wildlife species are plausible depending upon factors like technique, location, size, and time since treatment. For example, prescribed burning to control expanding conifers may have negative impacts on sagebrush-obligate species in the short-term if sagebrush cover is lost and trees are not sufficiently removed (Knick and others, 2014a). Conifer removal conducted in areas with poor herbaceous vegetation and invasive annual grasses can result in state shifts to annual grass dominance if follow up weed control and seeding are not implemented. Conifer thinning, as opposed to complete removal, may create ecological traps for sagebrush obligates, like sage-grouse, who might use areas of low conifer cover but suffer lower survival (Coates and others, 2017a). Mechanical conifer removal may also elevate wildfire hazard, and subsequent fire severity, in areas where slash is not properly addressed posttreatment. Some migratory raptors use conifer expansion areas in sagebrush shrublands for nesting habitat. Removal of nest trees can displace individual birds, although the population-level consequences are not well understood.

Habitat structure and composition for both sagebrush and woodland wildlife species have been affected by tree expansion and infill over the last 200 years, and wildlife habitat availability and use have likely been impacted across the woodland-to-shrubland spectrum. Adaptive habitat use may lead to perceived conflicts where restoration of sagebrush ecosystems directly affects wildlife using conifer expansion areas. For example, species that relied on historically open-stand persistent woodlands that have now transitioned to dense forests may have shifted habitat use to tree-encroached sagebrush shrublands.

Big game, such as mule deer, use pinyon-juniper woodlands, savannas, and conifer-invaded shrublands seasonally in some areas (for example, historical winter ranges), and pinyon-juniper has been suggested to provide important thermal and hiding cover. Concern over potential conifer removal impacts on deer habitat use has been raised (Coe and others, 2018), but mule deer use of pinyon-juniper can be a misleading indicator of habitat quality (Bergman and others, 2015; Maestas and others, 2019), and experimental evidence supporting the thermal and hiding cover hypothesis is lacking. In contrast, demographic benefits of pinyon-juniper removal for wintering mule deer have been rigorously tested and documented (Bergman and others, 2014a).

Gray flycatchers (*Empidonax wrightii*) occupy the shrubland/woodland ecotone and have increased significantly in recent years as pinyon-juniper has expanded into sagebrush and other shrublands (Sauer and others, 2017). Not surprisingly, chaining of pinyon-juniper woodlands and hand removal of juniper that had expanded into sagebrush reduced the breeding density of gray flycatchers (O'Meara and others, 1981; Holmes and others, 2017). The pinyon jay is closely associated with pinyon-juniper woodlands and has experienced long-term (1968–2017) population declines. Pinyon jay declines are perplexing given they have occurred during a period of preferred habitat expansion, suggesting changes in woodland extent may

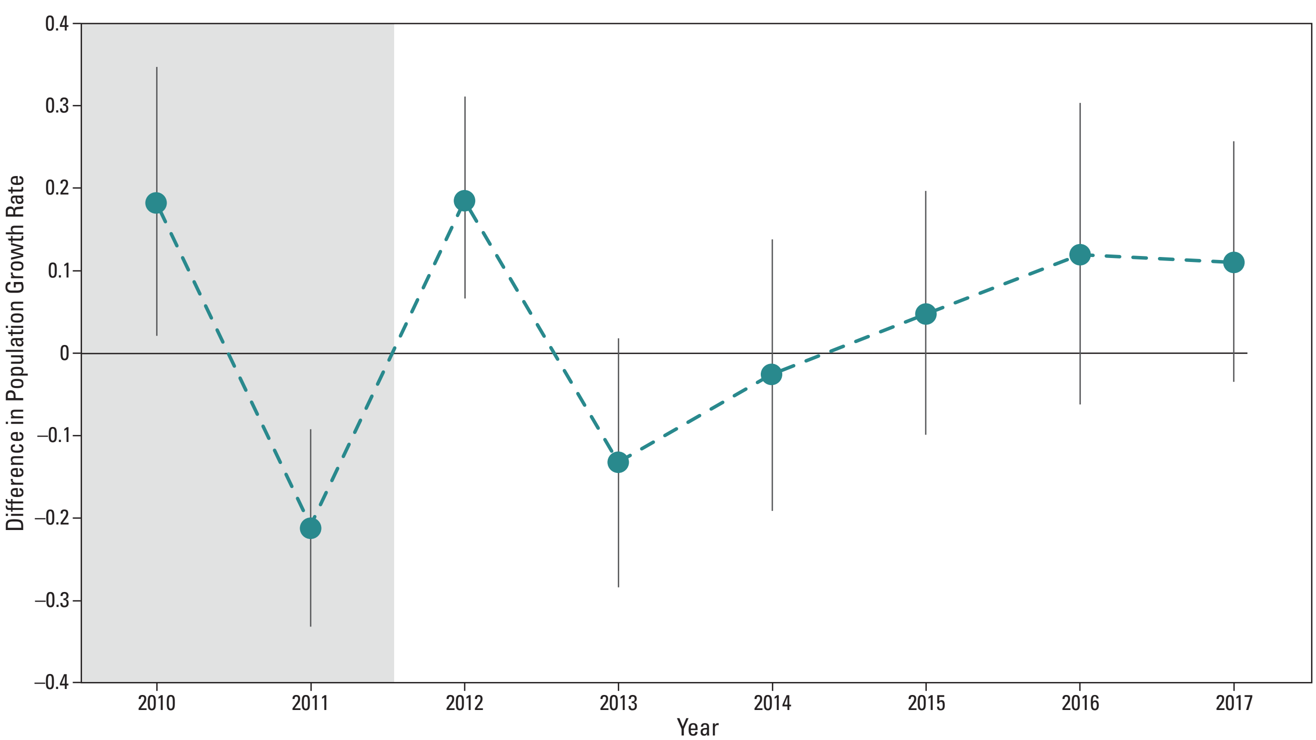

Figure M9. Population growth rates of greater sage-grouse (*Centrocercus urophasianus*) prior to and following landscape-scale conifer removal (adapted from Olsen, 2019). Graph represents the difference in population growth rates in the treatment and control study areas in the Warner Mountains, Oregon. Gray box indicates pretreatment period. Positive growth rates represent years the treatment area performed better than the control (vertical lines are 95-percent credible intervals).

not be the primary driver (Boone and others, 2018). Several alternative hypotheses for these population declines have been put forth including changes in woodland habitat structure and quality (for example, canopy closure), landscape-scale structural changes, pine productivity, and climate change (Boone and others, 2018).

Regardless, the lack of information on pinyon-jay declines has spurred speculation about unintended impacts of conifer management for sagebrush-associated species (Boone and others, 2018; Johnson and others, 2018; Magee and others, 2019). Local decreases in pinyon jay habitat occupancy have been documented in response to fuels treatments in persistent pinyon-juniper woodlands and savannas, but higher occupancy in treatments at landscape scales have been documented (Magee and others, 2019). Pinyon jays have large home ranges (typically 3,500 ha [8,645 acres] to 6,400 ha [15,800 acres]; Marzluff and Balda, 1992; Johnson and others, 2016), and in winter, flocks occasionally move hundreds of miles from their home ranges when food resources are limited (Johnson and Balda, 2020). Pinyon jay habitat use is highly dynamic across years in response to availability of pinyon pine nut production (Somershoe and others, 2020). Thus, the location of treatments along the woodland-to-shrubland spectrum relative to pinyon jay occurrence and breeding colony location likely influences the degree of potential effects. Ultimately, landscape scale studies that identify pinyon jay habitat requirements within and across seasons will allow for improved design of treatments so that sagebush obligates benefit while not impacting pinyon jays.

Conifer removal is often cast as creating winners and losers among wildlife species (Bombaci and Pejchar, 2016), but a more nuanced approach that considers ecological site potential and the full woodland-to-shrubland spectrum (see fig. M1) would allow for holistic multispecies management. Existing wildlife literature evaluating pinyon-juniper management rarely distinguishes between tree removal performed in persistent woodlands and savannas from that in historical shrublands and grasslands, yet ecological sites and associated state-and-transition models are widely used in rangeland ecology as a basis for management decisions (Caudle and others, 2013; Miller and others, 2015). Carefully crafted management prescriptions based on ecological site potential and historical stand conditions and dynamics (Floyd and Romme, 2012) are likely needed to address all species' habitat requirements at a whole watershed scale in the appropriate places on the landscape. Increasing availability of spatially explicit data now allows for optimization in landscape restoration planning that can further balance multispecies needs (Reinhardt and others, 2017; Ricca and others, 2018).

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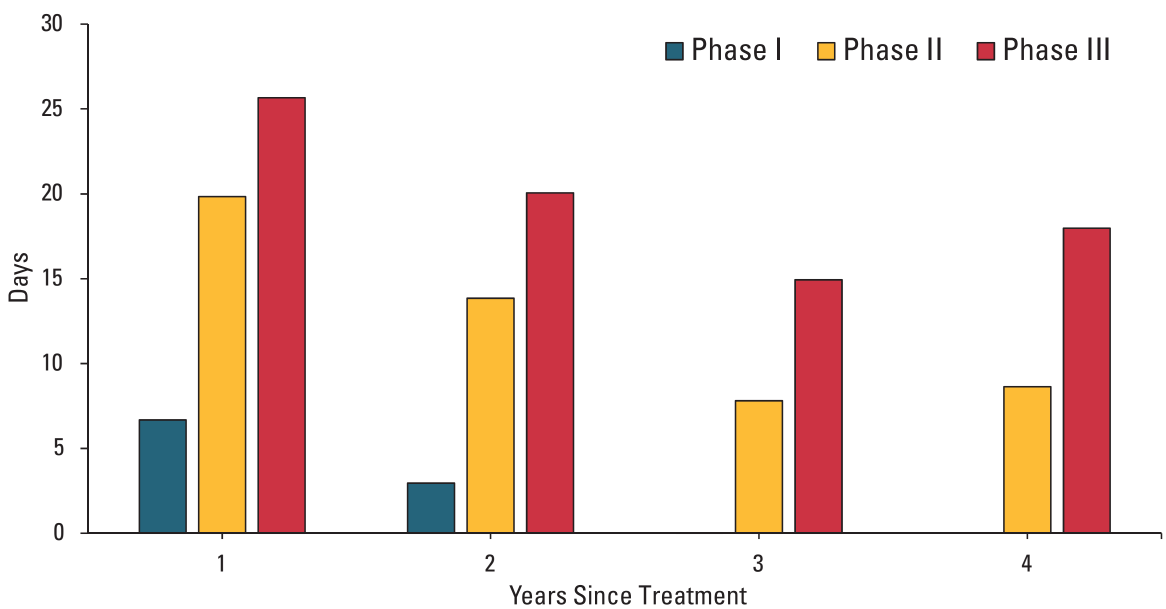

Figure M10. Additional days of soil-water availability following removal of encroaching conifer in Phase I, II, or III of conifer expansion. Tree removal decreases canopy interception of precipitation and tree water use, which results in additional days of soil water availability compared to untreated areas. Additional days of soil-water availability decline with increasing plant cover over time but remain significant for Phase II and III expansion (Roundy and others, 2014b). Figure adapted from Maestas and others (2016).