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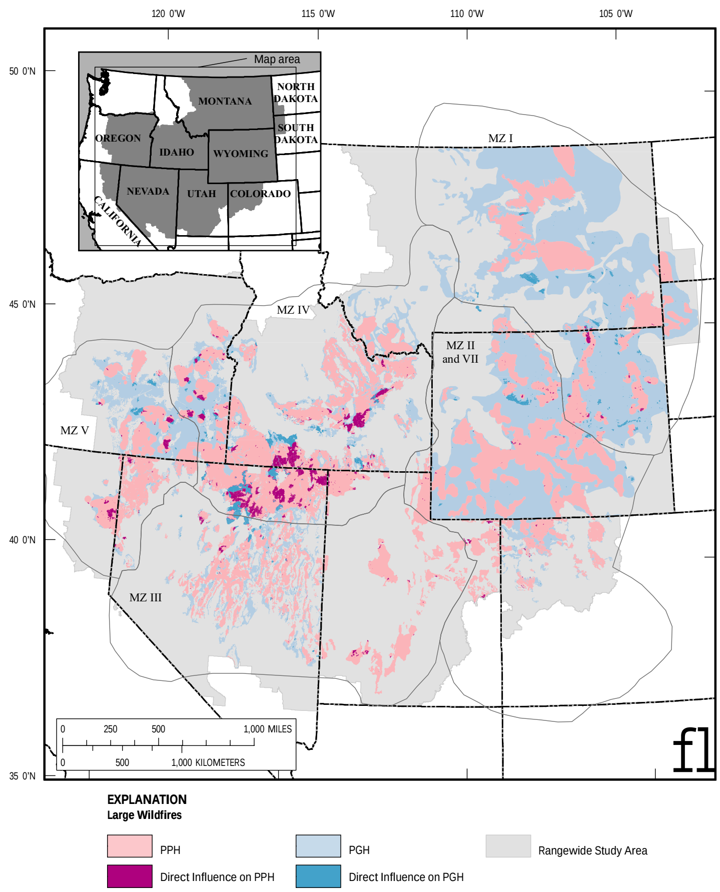

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Figure 25. Compilation of fires reported to NIFC within the past decade (2001-2011) within priority and general sage-grouse habitats (PPH and PGH).

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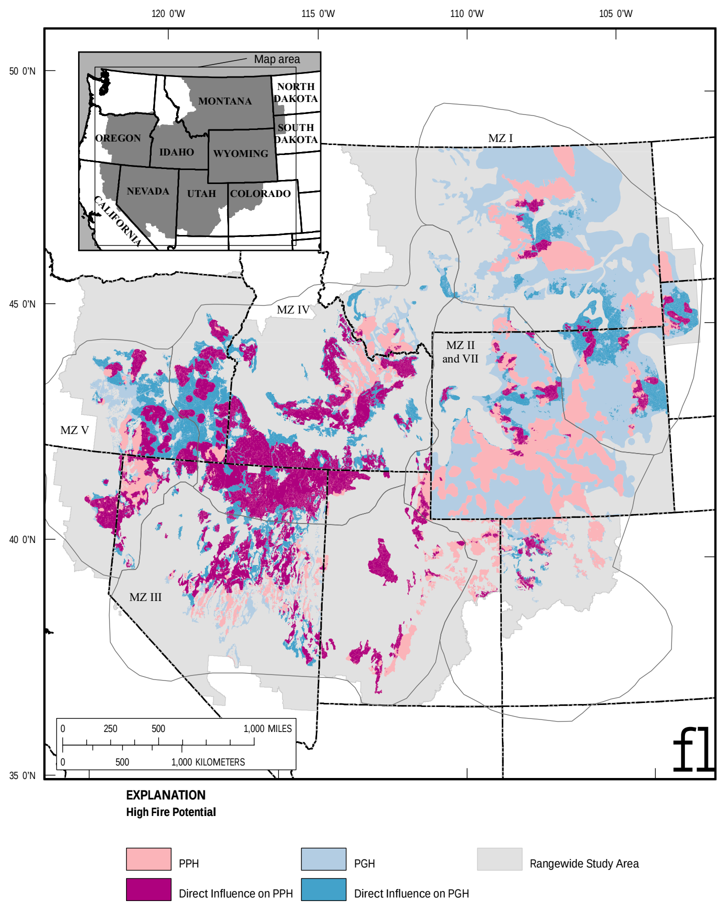

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Figure 26. Estimated risk of fire on federal lands affecting sage-grouse habitats (PPH and PGH) within each Management Zone

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In contrast, lack of fire at higher elevations, **where** moisture and productivity **are greater** than neighboring communities at lower elevations, have **contributed to an** increase in juniper cover (Miller and Rose 1995, Miller et al. 2000, Miller and Heyerdahl 2008, Sankey and Germino 2008, Shinneman et al. 2008, Bradley 2010). In these areas, active restoration using fire, or "fire-mimic" (mechanical) treatments, is needed to improve sagebrush habitats (Bradley 2010, Rowland et al. 2010). Importantly, all sites do not have equal restoration potential, with the greatest potential being in recent and "incomplete" invasions where vegetation and soils can readily recover (Shinneman et al. 2008); but recovery processes may be supported and enhanced through methods and timing of application (Bates et al. 2011, Rau et al. 2011).

Because of the important value of sagebrush canopies and tall grasses for nesting cover (Holloran et al. 2005, Beck et al. 2009), wildfires, prescribed fires (and treatments with similar effects on vegetation) that reduce these values are likely detrimental for sage-grouse. On the other hand, fire control and mitigation in all MZs represents an important modern component of habitat management due to the recent (circa 50 years) threat of wildfire in many areas. Particular caution and concern is warranted when noxious invasive species (notably, but not limited to, cheatgrass) are present in the pre-disturbance community, because these species may have lasting, detrimental effects on post-disturbance habitat conditions. The threat of large wildfires in priority habitats, resulting in removal of large stands of mature sagebrush, remains the significant threat due to fire for sage-grouse conservation.

A7. Invasive Plants

Presence of invasive species is a mechanism whereby any disturbance, especially larger ones, has the potential for a strong, negative effect on habitat quality (Crawford et al. 2004). In Wyoming big sagebrush types, especially in the Great Basin (all or part of MZs III, IV and V), the invasion by exotic annuals has resulted in dramatic increases in number and frequency of fires with widespread,

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detrimental effects on habitat conditions (Young and Evans 1978, West and Young 2000, West and Yorks 2002, Connelly et al. 2004). For example, big sagebrush communities invaded by cheatgrass have estimated mean fire return intervals of less than 10 years in many areas (Connelly et al. 2004) whereas the natural regime is estimated (conservatively) to be 10 to 20x longer. Increased fire frequency and intensity typically results in removal of the sagebrush canopy in affected areas with replacement by annual species that provide little, to no, habitat value (Knapp 1996, Epanchin-Niell et al. 2009, Rowland et al. 2010, Baker 2011, Condon et al. 2011). Presumably cheatgrass (*Bromus tectorum*) was able to thrive in this region, in part because there was no pre-existing (native) dominant annual plant species. **A**s this optimal colonist species established, chronic grazing **by** cattle, sheep,**,** and horses combined with drought and fire to increase the distribution and frequency of disturbance and further optimize this region for an annual grass (Knapp 1996). Importantly, research in sagebrush ecosystems has revealed an inverse relationship between cheatgrass dominance and native perennial herbs, especially grasses (West and Yorks 2002). Further, the post-disturbance response of sagebrush communities to fire and similar disturbances is strongly affected by the condition and composition before disturbance, **the** presence of propagules**,** and/or sprouting of native species (West and Yorks 2002, Beck et al. 2009, Epanchin-Niell et al. 2009, Condon et al. 2011). Cheatgrass competes with native grasses and forbs that are important components of sage-grouse habitat. **C**heatgrass abundance **is negatively** correlated with habitat selection **by sage-grouse** (Kirol et al. 2012)**,** indicating that changes in composition and structure associated with cheatgrass specifically degrade sage-grouse habitat. Invasion by Medusahead (*Taeniatherum caput-medusae*), which can replace cheatgrass in some circumstances, may be even worse as it also reduces perennial productivity, degrades wildlife habitat, supports high-frequency fire return intervals, and requires intensive treatment for restoration (Davies 2010). Infestation of these species, and others, cause direct degradation of sagebrush habitats resulting in (indirect) effects on local

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sage-grouse populations by affecting forage and cover quality with potential to cause complete avoidance (effective habitat loss).

In southern habitats (MZs III, IV, V, VII), cheatgrass is found primarily at elevations between 1600-2000m, compared to 600-1800m in the sagebrush steppe in Idaho, but has been expanding in habitats down to 1200m (Connelly et al. 2004). Large scale restoration is needed in many areas, making minimally-invaded areas highly valuable for habitat conservation. In the sagebrush steppe of northern habitats (all or parts of MZs I, II, IV, V, VI), cheatgrass is less ubiquitous but demonstrates increased dominance, productivity, and elevation range on south-facing slopes (Connelly et al. 2004) which indicates the need for careful local considerations and best-practices that minimize disturbance in areas with a threat (presence) of cheatgrass expansion. Potential for cheatgrass occurrence has been modeled (based on environmental correlations, which can help discern locations and habitats that have the greatest risk, either because cheatgrass is already on those landscapes (some of the risk has been realized) or the conditions are right to support cheatgrass (Figure 27). Summary data indicate that invasion potential is widespread and similar among assessed MZs (Table 20). Although the distribution of cheatgrass, and other annual invaders such as Japanese brome (*Bromus arvensis*), has been documented across shrub and grasslands of Colorado, Wyoming, and Montana, the currently available model was only parameterized for the Great Basin therefore only MZs III, IV and V are described here (Table 20; Figure 27). Similar information is being developed range-wide, as well as with sub-regional details. Due to the emerging nature of invasive plants, especially cheatgrass, information and rapid changes in species distributions, details of invasion, control and risks will be best provided by local information and sub-regional to regional scale models. Data presented here demonstrate the potential risk to priority habitats within the Great Basin and Snake River Plain based on a spatial model trained using field observations and GIS representation of dominant environmental patterns (that predict and/or

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restrict the distribution of the species). Model results suggest the most serious risk of cheatgrass invasion (in these analytical units) lies in the Snake River Plain where more than 20% of PPH and more than 30% of PGH are projected to be at risk of cheatgrass invasion (Table 20). The northern Great Basin follows closely behind with nearly 19% of priority habitats (PPH and PGH, respectively from an independent, non-overlapping estimate) whereas less (8% and 11% of PPH and PGH, respectively) of the southern Great Basin MZ (III) is projected to share this level of risk. Importantly, most (more than 50%) of the affected lands in each MZ are managed by BLM (negligible on USFS lands according to these data; Table 20).

Because of ecological and morphological characteristics, cheatgrass can often out-compete native perennial plants and promote rapid fire-return intervals (Klemmedson and Smith 1964, Connelly et al. 2004). The positive feedback cycle of fire, sagebrush loss, and cheatgrass dominance has resulted in entire landscapes being converted to annual grasslands (D'Antonio et al. 2009), and these areas typically require active restoration, including costs and effort, associated with eradication of weeds and re-seeding of native species, if local priorities indicate important habitat value of restored lands. Based on the scale of such efforts, locally planned and implemented sagebrush restoration efforts will likely benefit from planning and assessment at regional scales to strategically combat the spread and dominance of invasive annuals in priority habitats and connected areas.

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Table 20. Summary of overlap between sage-grouse habitats (PPH and PGH) and areas with a high potential for the potential distribution of cheatgrass across Management Zones III, IV and V (Great Basin).

Management Zone Entity	PPH SG Habitat (acres)	PPH Direct Footprint (acres)	PPH Direct Footprint (%)	PPH Relative Influence[1] (%)	PGH SG Habitat (acres)	PGH Direct Footprint (acres)	PGH Direct Footprint (%)	PGH Relative Influence[1] (%)
MZ I - GP	11,636,400	n/a	n/a	n/a	34,663,000	n/a	n/a	n/a
MZ II and VII - WB & CP	17,476,000	n/a	n/a	n/a	19,200,200	n/a	n/a	n/a
MZ III - SGB	10,028,500	823,700	8.21		3,970,100	445,800	11.23	
BLM	6,309,400	478,100	7.58	58	3,199,800	330,300	10.32	74
Forest Service	1,236,200	100	0.01	0	356,200	400	0.11	0
Tribal and Other Federal	260,800	42,200	16.18	5	29,100	2,700	9.28	1
Private	1,836,200	252,000	13.72	31	384,800	112,400	29.21	25
State	385,900	51,400	13.32	6	200	0	0.00	0
MZ IV - SRP	21,930,600	4,942,000	22.53		10,958,500	3,557,400	32.46	
BLM	13,710,700	3,034,900	22.14	61	4,928,200	2,006,900	40.72	56
Forest Service	1,613,800	68,700	4.26	1	1,113,500	77,700	6.98	2
Tribal and Other Federal	633,600	400,800	63.26	8	522,500	261,900	50.12	7
Private	4,890,200	1,239,500	25.35	25	3,516,742	985,500	28.02	28
State	1,019,373	167,200	16.40	3	846,200	201,900	23.86	6
Other	62,900	30,900	49.13	1	31,400	23,400	74.52	1
MZ V - NGB	7,097,200	1,330,300	18.74		5,808,000	1,101,500	18.97	
BLM	5,117,500	1,067,300	20.86	80	4,196,700	773,700	18.44	70
Forest Service	62,200	21,700	34.89	2	114,900	8,800	7.66	1
Tribal and Other Federal	717,100	150,700	21.02	11	101,800	19,400	19.06	2
Private	798,000	63,000	7.89	5	1,199,000	273,500	22.81	25
State	64,900	2,400	3.70	0	115,800	11,700	10.10	1
Other	337,500	25,100	7.44	2	79,800	14,300	17.92	1

[*] Data Source: Meinke, C.W., S.T. Knick, and D.A. Pyke. 2009. A spatial model to prioritize sagebrush landscapes in the intermountain west (USA) for restoration. Restoration Ecology 17:652-659.

[1] For management entities within a management zone, these were calculated as the percent of the total direct impact in the management zone represented by that management entity; i.e. the relative area of direct influence among management entities.

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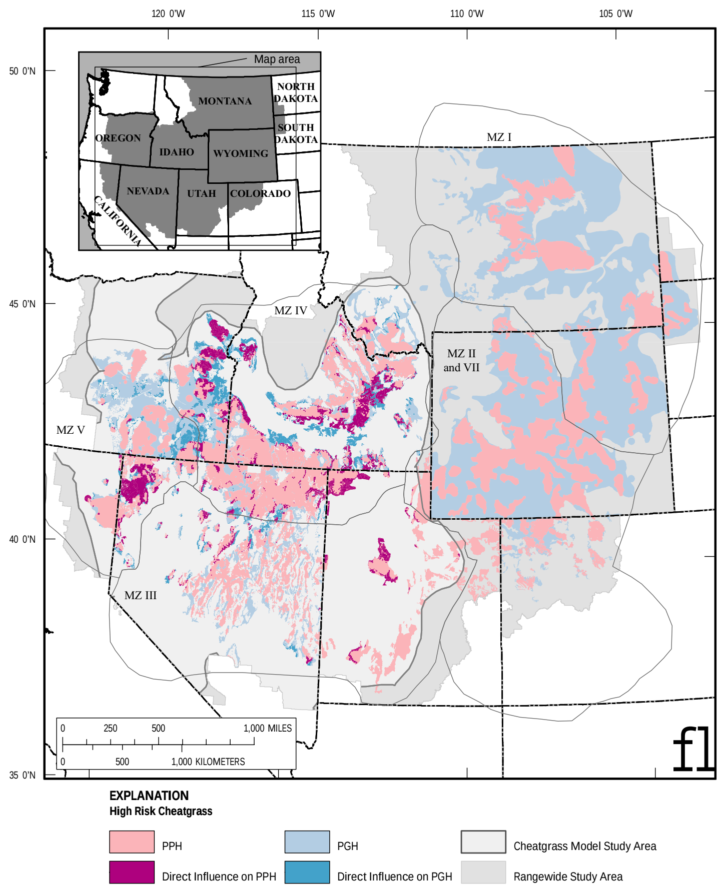

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Figure 27. High probability of cheatgrass occurrence in Management Zones III, IV and V (Great Basin) from logistic regression models of presence/absence using several environmental predictors.

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Invasive plants are thought to alter plant community structure and composition, productivity, nutrient cycling, and hydrology (Vitousek 1990) and may competitively exclude native plant populations (Mooney and Cleland 2001). In particular, invasive plants can reduce and eliminate vegetation that sage-grouse use for food and cover, resulting in habitat loss and fragmentation. An assortment of nonnative annuals and perennials and native conifers are currently invading sagebrush ecosystems. Many areas throughout the range of sage-grouse are at high risk from invasive plants, yet the most concentrated areas of risk include the Intermountain West and Great Basin (MZs III, IV, V, and VI). Much of the Great Basin is at risk for invasion by cheatgrass or pinyon-juniper encroachment within the next 30 years (Leu et al. 2008, Doherty et al. 2008), and where cheatgrass has invaded, there has typically been an increase in fire frequency resulting in further degradation of sage-grouse habitats by removing, and excluding sagebrush (Knapp 1996, Epanchin-Niell et al. 2009, Rowland et al. 2010, Baker 2011, Condon et al. 2011). Regions that are currently invaded or predicted by distribution models to be highly invasible may benefit from explicit guidance and practices that avoid, eliminate, or mitigate feedbacks in this cycle, including natural disturbances, over-grazing, treatments, new roads and industrial developments that disrupt native vegetation cover and destabilize soils. **Disrupting** the processes that generate chronic disturbance and thereby facilitate dominance of annual plants is a necessary first-step in the restoration and conservation process. At low levels, invasive plants can decrease forage quality and compete with native species that provide high-quality habitat values for sage-grouse, and similarly to agricultural systems, this decline can be expected to cause a decrease in secondary productivity (in this case, sage-grouse). But in cases of severe infestation, system phenology (timing of green-up), cover and forage quality, and fire regimes are often altered with widespread, severe, and detrimental effects on sage-grouse habitat conditions.

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A8. Conifer Woodland Expansion and Encroachment

Expansion of conifer woodlands, especially juniper (*Juniperus* spp.) present a threat to sage-grouse because they do not provide suitable habitat, and further, mature trees displace shrubs, grasses and forbs through direct competition for resources which are important components of sage-grouse habitat; juniper expansion is associated with increased bare ground and an increased potential for erosion (Petersen et al. 2009). Mature trees may offer perch sites for raptors, thereby, woodland expansion may also represent expansion of raptor predation threat, similarly to perches on powerlines, poles and other structures (also see Section III.C Predation). While the prolonged drought at the beginning of the 21st century (2002-2004) caused significant (55%) mortality of mature pinyon pine (Clifford et al. 2011), reducing the threat attributed to this species in some areas, increased pinyon-juniper forest density and distribution continue to be documented following the drought period and are recognized as a threat to the sagebrush ecosystem in other areas (Romme et al. 2009, Bradley 2010, Rowland et al. 2010). Intensive grazing in the late 1800s and early 1900s, coupled with climate and fire, have been associated with invasion of annual grasses at lower elevations and expansion of juniper and pinyon-pine at higher elevations (Burkhardt and Tisdale 1976, Miller et al. 1994, Provencher et al. 2007, Miller et al. 2011). Precipitation and fire are thought to drive long term trends in cover (Clifford et al. 2011, Miller et al. 2011), and disturbance-free periods coupled with grazing that reduced competition and sufficient moisture for tree seedlings increased success of tree establishment and woodland expansion during the 20th century (Miller and Rose 1995, Eva et al. 2007, Miller et al. 2011). In some areas (best documented in MZs III, IV, V, and VI) conifer encroachment is connected to reduced habitat quality in important seasonal ranges when woodland development is sufficient to restrict, reduce, shrub and herbaceous production (Connelly et al. 2004). While widespread, this problem affects specific sagebrush habitats and sage-grouse populations because of local juniper and/or pinyon-juniper

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expansions; notably, USFS research indicated more than 55% of Great Basin sagebrush ecosystems (MZs III and V) are at risk of cheatgrass invasion, whereas approximately 40% of this same landscape was at risk of displacement by juniper expansion. The encroachment problem is likely exacerbated by adjacent land-uses and cheatgrass invasions that have decreased the habitat values in nearby, lower-elevation big sagebrush communities, thereby increasing the importance of remaining habitats. Thus it may be important to consider surrounding land-use when prioritizing habitats for treatment to insure that the net result is more usable (for example, accessible to local populations) sage-grouse habitat across the local and regional landscape. Further, while juniper may have negative implications for sage-grouse habitat quality, these areas can provide important winter range for ungulates (Anderson et al. 2012) indicating potential interactions among multiple species and habitat functions at the sagebrush-forest ecotone. These locations can be mapped with reasonable accuracy, therefore encroachment within priority habitats may be specifically targeted. Regional modeling efforts suggested that locations within 1000m of current pinyon-juniper woodlands have the greatest (20%) juniper-expansion risk and locations, beyond this distance (1000-2000m) experience ½ of this potential (Bradley 2010). Based on a simple proximity modeling approach, whereby sagebrush habitats in close proximity (250m) of an existing conifer woodland (especially juniper and pinon pine, but also ponderosa pine and Douglass Fir) are recognized as having increased invasion risk due to proximity of the seed source, we estimate that 6 to 13% of sage-grouse habitat within all MZs may be at risk of conifer expansion. The most pronounced risks are, again, across the Great Basin where an estimated 13% (both PPH and PGH; southern Great Basin) and 10 to 12 % (PGH and PPH, respectively; northern Great Basin) are predicted to be at risk (Table 21). While substantial, the estimated risks in the Snake River Plain (7 to 8%, PPH and PGH) and Wyoming Basin (6 to 7%, PPH and PGH) are perceived to be smaller (i.e., less area projected to be affected). Importantly, the acreage of predicted woodland expansion is one-half of the area projected for

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cheatgrass risk, and not all of these areas will be invaded uniformly or completely. In addition, acreage projected to be a "high fire risk" is 2x to 10x greater (depending on MZ) than the area of projected conifer expansion. While the precise probability and realization of woodland expansion will likely vary (from these model results) within zones identified, based on local environmental conditions, for example, this risk assessment identified large portions of sage-grouse habitat in MZs III, IV and V as at risk of tree-invasion based on proximity to seed sources (Table 21) making this a potentially important consideration for managing habitats in those regions.

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Table 21. A model of potential woodland expansion based on proximity of sagebrush habitats to woodland seed sources within each Management Zone.*

Management Zone Entity	PPH SG Habitat (acres)	PPH Direct Footprint (acres)	PPH Direct Footprint (%)	PPH Relative Influence[1] (%)	PGH SG Habitat (acres)	PGH Direct Footprint (acres)	PGH Direct Footprint (%)	PGH Relative Influence[1] (%)
MZ I - GP	11,636,400	130,600	1.12		34,663,000	894,500	2.58	
BLM	2,994,300	33,100	1.11	25	4,524,900	180,800	4.00	20
Forest Service	292,400	1,100	0.38	1	515,300	20,300	3.94	2
Tribal and Other Federal	219,700	1,700	0.77	1	2,427,700	25,400	1.05	3
Private	7,132,500	82,800	1.16	63	24,682,800	604,800	2.45	68
State	995,600	12,000	1.21	9	2,498,400	63,100	2.53	7
Other	1,900	0	0.00	0	13,900	0	0.00	0
MZ II and VII - WB & CP	17,476,000	1,076,300	6.16		19,200,200	1,390,500	7.24	
BLM	9,021,200	499,700	5.54	46	9,012,500	595,500	6.61	43
Forest Service	162,000	18,200	11.23	2	452,500	62,300	13.77	4
Tribal and Other Federal	784,000	77,100	9.83	7	1,354,600	88,400	6.53	6
Private	6,233,900	373,000	5.98	35	7,394,800	545,800	7.38	39
State	1,244,800	106,600	8.56	10	979,800	97,800	9.98	7
Other	30,100	1,700	5.65	0	6,000	700	11.67	0
MZ III - SGB	10,028,500	1,292,400	12.89		3,970,100	517,400	13.03	
BLM	6,309,400	751,400	11.91	58	3,199,800	394,000	12.31	76
Forest Service	1,236,200	247,000	19.98	19	356,200	86,800	24.37	17
Tribal and Other Federal	260,800	29,400	11.27	2	29,100	4,600	15.81	1
Private	1,836,200	217,400	11.84	17	384,800	32,000	8.32	6
State	385,900	47,100	12.21	4	200	0	0.00	0
MZ IV - SRP	21,930,600	1,698,500	7.74		10,958,500	918,100	8.38	
BLM	13,710,700	938,700	6.85	55	4,928,200	311,300	6.32	34
Forest Service	1,613,800	248,200	15.38	15	1,113,500	228,100	20.48	25
Tribal and Other Federal	633,600	10,000	1.58	1	522,500	11,100	2.12	1

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Management Zone Entity	PPH				PGH			
	SG Habitat (acres)	Direct Footprint (acres)	Direct Footprint (%)	Relative Influence[1] (%)	SG Habitat (acres)	Direct Footprint (acres)	Direct Footprint (%)	Relative Influence[1] (%)
Private	4,890,200	427,500	8.74	25	3,516,742	295,200	8.39	32
State	1,019,373	67,700	6.64	4	846,200	69,600	8.23	8
Other	62,900	6,400	10.17	0	31,400	2,900	9.24	0
MZ V - NGB	7,097,200	823,500	11.60		5,808,000	533,700	9.19	
BLM	5,117,500	597,500	11.68	73	4,196,700	346,600	8.26	65
Forest Service	62,200	11,300	18.17	1	114,900	29,200	25.41	5
Tribal and Other Federal	717,100	44,000	6.14	5	101,800	8,100	7.96	2
Private	798,000	106,800	13.38	13	1,199,000	132,300	11.03	25
State	64,900	2,700	4.16	0	115,800	7,300	6.30	1
Other	337,500	61,200	18.13	7	79,800	10,100	12.66	2

* Data Source: Modeled from National GAP/ReGAP Landcover, National GAP Analysis Program, 2010. Identified sagebrush within 120m of conifer vegetation types.

[1] For management entities within a management zone calculated as the percent of the total direct impact in the management zone represented by that management entity; i.e. the relative area of direct influence among management entities .

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Prescribed fire is often used as an affordable and semi-natural means to control woody invasion and restore invaded communities (Pyke 2011). However, it is not clear that prescribed fire is the best management option in many cases (Rhodes et al. 2010). The best results reported were attained using manual treatments that retained cover of woody and herbaceous litter post-treatment (Baughman et al. 2010). A review of the impacts of treatments and grazing on grouse (Beck and Mitchell 2000) suggested that fire be applied cautiously since optimal patterns of burned-unburned habitat and the ideal size(s) for burned patches are unknown, suggesting that small treatment areas coupled with monitoring of subsequent habitat and use patterns may improve restoration success. Research focused on treatment effectiveness (Brockway et al. 2002) indicated that mechanical tree thinning increased native understory biomass by 200%; typically, this type of response represents improvement of sage-grouse habitat. Additionally, mechanical operations followed by seeding have been used successfully to restore shrub- and tree-dominant states, however these are typically the most expensive management actions (Provencher et al. 2007). Previous efforts indicate that the success of native plant recovery increases with less pinyon and juniper cover, and increases with improved condition of the pre-treatment community (Pyke 2011). Gradients of condition and potential, estimated locally and applied during the planning process, coupled with local habitat and restoration priorities, can be used to guide specific actions (see Section III.A11 Habitat Treatment and Vegetation Management).

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A9. Grazing

The effect of livestock grazing on range condition is one of the most contentious issues underlying the management and use of sagebrush habitats (Crawford et al. 2004). However, livestock grazing is the most widespread land use across the sagebrush biome (Connelly et al. 2004), making discussion of its role in sagebrush ecosystem and specifically sage-grouse population conservation a necessary consideration. Although isolated areas exist that have not been grazed by domestic livestock

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(for example, the kipukas in the Great Rift lava fields of southern Idaho), most sagebrush habitats have been grazed in the past century (Knick 2011b). Livestock grazing has been described as a diffuse form of biotic disturbance that exerts repeated pressure over many years on a system; unlike point-sources of disturbance (for example, fires that have acute perturbations from well-defined origins), livestock grazing is characterized as a "press" form of disturbance because it exerts *repeated* pressure across the landscape (Knick 2011b). Thus, effects of grazing are not likely to be detected as disruptions – except in extreme cases as around water sources or mineral-nutrient blocks – but rather as differences in the processes and functioning of the sagebrush system (Knick 2011b). Importantly, effects of grazing are not distributed evenly because historic practices, management plans and agreements, and animal behavior all dictate differential use, and therefore different effects.

Historically, the numbers of livestock and the area grazed increased from 1880 to 1905 combined with the drought that followed in the 1920s and 1930s **severly altered** the condition of western landscapes (Connelly et al. 2004). Numbers of livestock increased from 4.1 million cattle and 4.8 million sheep in 1870 to 19.6 million cattle and 25.1 million sheep in 1900 (Knick 2011b). Native perennial grasses and forbs that were not adapted to heavy grazing pressure were depleted from the vegetative community and replaced in much of the Great Basin, Snake River plain and surrounding inter-mountain regions by grazing tolerant grass species, exotic annual grasses, or both. Loss of protective vegetation cover in some communities resulted in extensive soil disturbance and erosion, and shrub density increased (although the total distribution of shrubs across the region likely remained similar). Research revealed that the decline of palatable forage species and increases in plant species of low palatability took only 10 to 15 years at any given site under heavy uncontrolled grazing (Knick 2011b). Forage production for livestock dropped to an estimated 10% of site potential following depletion of the vegetation community in some regions. The area required to support an animal unit

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month (AUM; the amount of forage required to feed one 1,000-pound cow and her calf, one horse, five sheep, or five goats for one month) was estimated at 1.2 ha prior to European settlement, 3.7 ha in the 1930s, and 3.2 ha in the 1970s (Knick 2011b). Implied in this estimate is the assumed relationship that, 3 times the area per AUM is required because current primary production is approximately 1/3 of what it was during the first interval, years after severe over-grazing and droughts of the early 1900s ended. Current use patterns vary based on local and regional plans and conditions, and grazing allotments and pastures on public lands (management units) represent the typical planning, leasing, and evaluation units used in grazing management across sage-grouse range. Grazing, assessed using Field Office records of grazing allotments not meeting wildlife land health standards due to livestock grazing, most influences sage-grouse habitats predominantly throughout MZ IV and western portions of Management Zone III, although BLM lands not meeting wildlife land health standards can be found throughout the range of sage-grouse (Table 22, Figure 28). **Importantly**, assessments for some lands were not available, and conditions have changed since the data were gathered, so regional scale comparisons may be misleading (contemporary, local data should supersede this information in most cases). Approximately 6.6 million acres (10.42%) of BLM controlled sage-grouse range did not meet land health standards, and 17.9% of priority habitats in MZs III and IV did not meet these standards.

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Table 22. Area of BLM Allotments Not Meeting Wildlife Standards with grazing as the causal factor overlapping sage-grouse habitats (PPH and PGH) within each Management Zone.* Only the BLM-managed portions of allotments were evaluated.

Management Zone Entity	PPH			PGH		
	SG Habitat (acres)	Direct Footprint (acres)	Direct Footprint (%)	SG Habitat (acres)	Direct Footprint (acres)	Direct Footprint (%)
MZ I - GP						
BLM	2,994,300	82,500	2.76	4,524,900	52,100	1.15
MZ II and VII - WB & CP						
BLM	9,021,200	286,900	3.18	9,012,500	366,000	4.06
MZ III - SGB						
BLM	6,309,400	965,400	15.30	3,199,800	654,600	20.46
MZ IV - SRP						
BLM	13,710,700	2,617,200	19.09	4,928,200	968,900	19.66
MZ V - NGB						
BLM	5,117,500	417,000	8.15	4,196,700	158,700	3.78

* Data Source: (Veblen et al. 2011, Assal et al. 2012).