

Managing U.S. Grasslands for Birds, Carbon, and Co-benefits

Natural Climate Solutions
Habitat Management Brief — Appendix



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Adaptive Grazing

Adaptive grazing management is a goal-oriented approach to livestock management in which stocking rates, timing, and duration of grazing are continuously adjusted in response to monitored ecological conditions rather than fixed schedules or predetermined rotations. As implemented in adaptive multi-paddock (AMP) systems, it typically involves small paddocks grazed at high livestock densities for short periods with long recovery periods before regrazing (Shrestha et al., 2020). Decisions about where to graze, for how long, and at what density are driven by observations of residual forage availability, vegetation recovery, precipitation, and management objectives. There are 5 iterating components: identifying and prioritizing measurable outcomes, understanding rangeland dynamics, accounting for logistical and economic constraints, monitoring relevant metrics, and modifying management in response to monitoring data (Derner et al., 2022).

We use the term "adaptive grazing" rather than "rotational grazing" throughout this entry because "rotational grazing" encompasses too wide a range of practices (Russias et al., 2025). The adaptive management framework better describes both the approaches evaluated in the strongest relevant research and the management philosophy that guides our ACR program.

Birds

Grassland birds collectively require a mosaic of vegetation structures that vary across space and shift across time. By continuously varying stocking density, timing, duration, and rest periods in response to monitored vegetation conditions, adaptive grazing approaches can generate and sustain the range of disturbance intensities that no fixed grazing system reliably produces on its own (Golding and Dreitz, 2017; Morris, 2021; Vold et al., 2019).

The vegetation conditions birds require differ markedly by region and site productivity. More productive systems need more intensive disturbance to generate the structural heterogeneity that benefits short-grass species, while in semiarid mixed-grass systems even moderate stocking rates can reduce cover below thresholds needed by dense-grass obligates (Ahlering and Merkord, 2016; Davis et al., 2020; Vold et al., 2019). Under adaptive grazing, decisions about stocking rate, timing, and rest are made in direct response to vegetation recovery, forage availability, and

precipitation, allowing management to match local birds requirements rather than following a predetermined schedule. AMP-grazed systems have shown documented benefits for grassland bird habitat where this adaptive approach has been explicitly implemented (Davis et al., 2020; Derner et al., 2022; McGraw et al., 2024). Breeding season timing is a particular consideration: because grazing pressure during nesting can degrade habitat that attracted birds during territory establishment (Sliwinski et al., 2019), adaptive grazing managers can incorporate nesting phenology into decisions about when and how intensively to graze individual paddocks.

Grassland bird monitoring is therefore recommended as an integral component of adaptive grazing. Short-term indicators such as visual obstruction readings can inform within-season cattle movement decisions, while long-term population monitoring tracks whether bird outcomes are being achieved and whether management strategy needs to shift (Derner et al., 2022).

Carbon

Soil carbon and greenhouse gas outcomes in grazed systems are primarily driven by two variables: stocking rate and the adequacy of plant recovery periods relative to actual growing conditions (Briske et al., 2008; di Virgilio et al., 2019). Adaptive grazing management directly targets both. By continuously calibrating grazing pressure to observed forage availability and adjusting recovery periods to actual vegetation growth rather than calendar schedules, adaptive grazing creates the conditions under which carbon gains are most likely to occur: maintained plant cover and productivity, improved litter accumulation, and healthy soil microbial communities that support SOC formation (Bai and Cotrufo, 2022; Byrnes et al., 2018). The GHG evidence bears this out directly: AMP-grazed soils in northern temperate grasslands showed 1.5 times greater methane uptake than non-AMP soils, and at warmer temperatures AMP soils emitted 18% less CO₂ (Shrestha et al., 2020). Effects on nitrous oxide were independent of grazing system, and some conflicting results on GHG emission potential suggest outcome magnitudes vary with local conditions (Derner et al., 2022; Dowhower et al., 2020).

Co-benefits

There are documented co-benefits to adaptive grazing management, with the most consistent evidence pointing toward improvements in soil properties and water dynamics. AMP management improves litter accumulation, water infiltration, and soil microbial communities compared to continuously grazed soils

(Morris, 2021; Sanguinetti et al., 2025; Teague et al., 2011), and the associated shift toward fungal-dominated soils reinforces these gains, as fungi retain water and nutrients more effectively than bacteria (Morris, 2021; Teague et al., 2011). These changes help maintain soil moisture during dry periods and slow runoff, which likely underlies why U.S. Great Plains ranchers consistently rate drought recovery as the benefit they value most from adaptive multi-paddock management (Wang et al., 2022). Producers using genuinely intensive multi-paddock systems report grazing seasons roughly 39 days longer per year than continuous grazing users, a direct reduction in hay feeding costs (Wang et al., 2022). Evidence also points toward arthropod diversity benefits under AMP management (Schmid et al., 2024), and forage production increases have been documented in agronomically managed systems in the more productive mesic southeastern United States (Apfelbaum et al., 2022).

Plant diversity does not appear to increase with AMP management (Döbert et al., 2025; Grenke et al., 2022), and in semi-arid rangelands the high stock densities characteristic of AMP systems can reduce livestock weight gain by 11-16% (Augustine et al., 2020; Jorns et al., 2024). This represents a meaningful economic tradeoff for ranchers in semi-arid systems; reducing stock density is one approach to address this within an adaptive framework (Derner et al., 2021). Realizing the full suite of co-benefits requires sufficient paddock numbers and recovery periods calibrated to local growing season conditions, which are core elements of adaptive grazing management design.

Protect Wetlands from Grazing

Wetlands and riparian areas are an important component of grassland ecosystems. They promote moisture retention and biological productivity, playing a disproportionate role in sustaining ecosystem function as well as diversity and abundance of plants, birds, and other wildlife. Wetlands also play an important role in long term, belowground carbon storage, retaining higher belowground biomass than surrounding drier areas. Livestock grazing can impact wetland and riparian function, and careful management of the timing of grazing and distribution of livestock is important for minimizing negative impacts of overgrazing in riparian areas (Maestas et al., 2023).

Birds

Wetlands are key habitats for a variety of bird species and provide habitat for aquatic emergent insects that are an important prey source (Macdade et al. 2011, Berzins et al. 2026). Livestock grazing in wetlands and riparian areas has been shown to decrease bird species richness and abundance (Popotnik and Giuliano, 2000), whereas excluding livestock from riparian and wetland areas increases bird abundance and diversity, including for populations of key wet-meadow and riparian-dependent species (Dobkin et al., 1998; Poessel et al., 2020). One study noted that bird abundance and diversity increased steadily for 12 years following exclusion, and remained consistent 24 year post-exclusion (Poessel et al., 2020). While full livestock exclusion benefits bird abundance and richness most, a study by Nelson et al. (2011) found that seasonal livestock exclusion also results in increased bird abundance and diversity, although to a lesser extent (29% lower bird abundance on seasonally grazed versus complete cattle exclusion).

Carbon

Livestock exclusion from, and protection of, wetland and riparian areas is generally found to positively impact soil carbon storage (Ascenzi et al., 2025; Booth et al., 2022; Kauffman et al., 2004; Willard et al., 2025), although one study found no change in carbon fluxes between wetlands with livestock exclusion and those without (Finocchiaro et al., 2014). Booth et al. (2022) noted an increase in soil temperature and decrease in soil moisture with livestock grazing, enabling greater loss of soil organic matter, and indicated that it may be especially beneficial to “end growing seasons with full stands of riparian vegetation to reduce soil warming and build organic matter”.

There is mixed evidence on whether wetland restoration can be as beneficial to soil carbon storage as wetland protection, but both protection and restoration benefit carbon stores more than allowing full grazing access (Ascenzi et al., 2025; Willard et al., 2025). Exclosed wetland meadows in eastern Oregon were found to have 62% greater total belowground biomass than grazed sites (Kauffman et al., 2004).

Co-benefits

In addition to benefits to birds and carbon, livestock exclusion from wetland and riparian areas has numerous other ecosystem benefits. Water quality, water infiltration rate, stream bank stability, wetland and native plant community richness and diversity, and fish and other wildlife all benefit from livestock exclusion

(Belsky et al., 1999; Booth et al., 2022; Coles-Ritchie et al., 2007; Dobkin et al., 1998; Jackson, 2013; Kauffman et al., 2004; Kidd and Yeakley, 2015; Maestas et al., 2023), with one study noting that infiltration rate was 233% greater on sites with livestock exclusion (Kauffman et al., 2022). However, on sites where invasive plant species such as reed canarygrass are abundant, targeted grazing or short-term livestock exclusion may be more effective management alternatives that can control reed canarygrass dominance while reducing livestock impacts (Kidd and Yeakley, 2015).

Control of Invasive Plant Species

Grasslands are under constant pressure from conversion and degradation driven by the invasion of non-native grasses and forbs. Species such as

cheatgrass (*Bromus tectorum*) aggressively outcompete native flora, leading to a precipitous decline in plant diversity over time (Bernath-Plaisted et al., 2023). The primary impact of these non-woody invasive species on

avian communities is the degradation of habitat structure and quality. By forming dense, uniform monocultures, these plants eliminate the structural mosaic and bare-ground patches essential for the foraging and nesting of grassland-obligate specialists (Nelson et al., 2018; Scheiman et al., 2003).

Beyond biodiversity loss, these invasions fundamentally re-engineer ecosystem function, leading to adverse impacts to carbon storage and sequestration, wildfire, and water. Invasive species like cheatgrass can shift carbon from stable, deep-soil reservoirs into volatile, fire-prone surface litter (Nagy et al., 2021). While these invaders may appear productive, native grasslands are far more efficient at sequestering carbon into long-term underground pools (Liao et al., 2008). This shift toward surface-loading, combined with altered fire regimes, increases wildfire frequency and intensity, effectively transforming many western grasslands from carbon sinks into carbon sources. Furthermore, disrupted hydrological cycles—ranging from increased runoff and erosion to the depletion of deep-water reserves—exacerbate the economic burden on rural communities by significantly reducing grazing capacity and land value (DiTomaso, 2000).

Common Invasive Plant Species in U.S. Grassland Systems

Species	**Primary Region(s)**
yellow starthistle (<i>Centaurea solstitialis</i>)	West (CA, ID, OR, WA)
leafy spurge (<i>Euphorbia esula</i>)	Northern Plains, West, Midwest
sericea lespedeza (<i>Lespedeza cuneata</i>)	Southeast, Central Plains
spotted knapweed (<i>Centaurea stoebe</i>)	Intermountain West, Northeast, Midwest
crown vetch (<i>Securigera varia</i>)	Midwest, Northeast
bird's-foot trefoil (<i>Lotus corniculatus</i>)	Midwest, Northeast
spiny cocklebur (<i>Xanthium spinosum</i>)	Central & Southern Plains
field bindweed (<i>Convolvulus arvensis</i>)	Nationwide (Agricultural)
Russian thistle (<i>Salsola tragus</i>)	Arid West, Great Plains
bull thistle (<i>Cirsium vulgare</i>)	Nationwide
plumeless thistle (<i>Carduus acanthoides</i>)	Midwest, Northeast
purple star thistle (<i>Centaurea calcitrapa</i>)	West Coast, Northeast
cheatgrass (<i>Bromus tectorum</i>)	West, Intermountain West
kentucky bluegrass (<i>Poa pratensis</i>)	Northern Plains, Midwest
Japanese brome (<i>Bromus japonicus</i>)	Great Plains, West
medusahead (<i>Taeniatherum caput-medusae</i>)	West, Great Basin
buffelgrass (<i>Pennisetum ciliare</i>)	Southwest (Sonoran Desert)
johnsongrass (<i>Sorghum halepense</i>)	South, Southeast, Midwest
cogongrass (<i>Imperata cylindrica</i>)	Southeast, Gulf Coast
smooth brome (<i>Bromus inermis</i>)	Northern Plains, Midwest
king ranch bluestem (<i>Bothriochloa ischaemum</i>)	Southern Plains, Texas
bermudagrass (<i>Cynodon dactylon</i>)	South, Southwest

Common Invasive Plant Species in U.S. Grassland Systems

Species	**Primary Region(s)**
lehmann lovegrass (<i>Eragrostis lehmanniana</i>)	Southwest
guinea grass (<i>Megathyrsus maximus</i>)	Deep South (TX, FL, HI)
reed canary grass (<i>Phalaris arundinacea</i>)	North, Pacific NW (Wetlands), Midwest
tall fescue (<i>Schedonorus arundinaceus</i>)	Midwest, Southeast

Birds

Habitat alteration associated with invasive plants degrades habitat structure and quality, reducing the capacity of grassland ecosystems to sustain populations of grassland-obligate birds (Nelson et al., 2018; Scheiman et al., 2003). Invasive groundcover like cheatgrass, kentucky bluegrass and tall fescue create dense, matted, and uniform monocultures. These create biological deserts that homogenize the landscape, inhibit movement, and offer poor concealment from predators (Fuhlendorf et al., 2006). These dense mats lack the bare ground patches needed by species such as the Lesser Prairie-Chicken (*Tympanuchus pallidicinctus*) and Grasshopper Sparrow (*Ammodramus savannarum*) for foraging and movement. High infestations of leafy spurge (*Euphorbia esula*) significantly lower the densities of both Grasshopper and Savannah Sparrow (*Passerculus sandwichensis*) (Scheiman et al., 2003).

Controlling tall fescue improves habitat quality, leading to increased nesting, survival, abundance, and fledgling production and decreased cowbird parasitism for species like the Dickcissel (*Spiza americana*) (Coon et al., 2025; Nelson et al., 2018). Invasive grasses also crowd out the forbs underpinning grassland food webs. Because invasive plants also support fewer native insects (caterpillars and grasshoppers) than the native vegetation they replace, this loss of forb cover translates into a thinner prey base and lower fledgling weights in grassland songbirds (Flanders et al., 2006; Lloyd and Martin, 2005)(Flanders et al., 2006; Lloyd and Martin, 2005). The same prey limitation extends to upland game birds, whose chicks are largely insectivorous in their first weeks. Lesser Prairie-Chicken brood-habitat use and chick survival are tied to invertebrate (especially grasshopper) biomass, which increases with forb cover (Jamison et al., 2002; Pitman et al., 2006).

Although introduced "improved" forage grasses such as Old World bluestems (OWB, e.g., WW-B.Dahl (*Bothriochloa bladhii*) and King Ranch bluestem (*Bothriochloa ischaemum* var. *songarica*) and kleingrass (*Panicum coloratum*) are often perceived as equivalent to native grassland for birds, the evidence indicates

otherwise: their dense monocultures displace the forbs that sustain arthropod prey, yielding far lower food availability and bird species richness than native or CRP grasslands (George et al., 2013; Hickman et al., 2006).

Carbon

Invasive grasses alter carbon storage by modifying biomass levels and the distribution of carbon within the soil profile (Nagy et al., 2021). In general, invasive species like cheatgrass lead to a net reduction in carbon storage (Nagy et al., 2021), by reducing aboveground biomass (Maxwell and Germino, 2022) and shifting carbon into less stable surface pools. In the Great Basin, cheatgrass invasion led to a 55% reduction in above-ground and a 62% reduction in below-ground biomass carbon compared to native sagebrush (*Artemisia* spp.). (Nagy et al., 2021). Invasive species like reed canarygrass can decrease the retention time of carbon and accelerate turnover cycles, thus reducing sequestration capabilities over time (Wisconsin Reed Canary Grass Management Working Group., 2009). While fescues may initially show high sequestration, their lack of deep root structure limits their long-term carbon sink potential compared to diverse, native perennial systems (Skersiene et al., 2023). Unlike deep-rooted native grasses (6-10 ft), cheatgrass has a shallow, brittle root system that dies back annually, increasing dry fuel loads for fires and releasing carbon rather than sequestering it long term. (Bradley et al., 2006; Maxwell and Germino, 2022). This creates frequent, intense wildfires that release accumulated surface carbon, turning parts of the Western U.S. from a carbon sink into a source (Bradley et al., 2006).

Restoring invasive species landscapes back to native grassland communities recovers carbon storage (Maxwell et al., 2024). Soil health also recovers quickly after invasive removal; an increased fungal-to-bacterial ratio indicates a healthier, more stable soil with faster nitrification, lower nitrogen loss, and stable carbon and nitrogen levels (Dickens et al., 2015).

Co-benefits

Water- Invasive plant species disrupt hydrological processes through varied mechanisms. For example, the presence of spotted knapweed, with its shallow and low density-root system, can increase surface runoff by 56% and soil erosion by 192% leading to increased stream sedimentation and lower water quality downstream (Lacey et al., 1989). Thatch-forming species, like kentucky bluegrass, also increase runoff through reduced infiltration rates and increased overland flow (Gasch et al., 2020b). Conversely, deep-rooted invasives like leafy spurge and yellow starthistle deplete deep water reserves needed for native plant communities (DiTomaso, 2000).

Soil Health- Many invasive plant species re-engineer soil chemistry to gain a competitive advantage. Some non-native C4 grasses, like non-native bunchgrasses, are efficient in using limited nutrient resources but they alter the distribution of carbon and nitrogen and can decrease long-term soil fertility and have negative effect on plant diversity outcompeting native grasses (Reed et al., 2005). Sites with high levels of invasive plant species have accelerated nitrogen cycling, altering microbial communities, leading to carbon and nutrient loss through leaching and respiration (Liao et al., 2008). Some invasive plant species utilize allelopathic suppression, releasing toxic chemicals into the soil which directly inhibit native seed germination and root elongation (Bais et al., 2003). These changes disrupt mycorrhizal networks essential for native plant nutrient uptake—altering soil to favor fast-growing invasives and suppressing slow-growing natives adapted to low nutrient soils.

Biodiversity- Homogenized vegetation in invaded areas results in a loss of niches, triggering a decline in biodiversity, such as a reduction in native insect abundance and diversity (Litt and Steidl, 2010). For example, cheatgrass makes it harder for small mammals and reptiles to forage and can lead to reduced body condition (Ceradini and Chalfoun, 2017; Hall et al., 2009). Grassland restoration triggers a multi-taxonomic rebound—eliminating invasive groundcovers like tall fescue facilitates a rapid increase in arthropod biomass and diversity within the first year, providing a critical protein base for the entire food web (Coon et al., 2025).

Productivity- Invasive plant species lead to roughly \$2 billion in revenue loss annually in the U.S., due to lower forage yield and quality, increased management costs, and reduced land value (DiTomaso, 2000). For example, leafy spurge is unpalatable to cattle, significantly

reducing grazing capacity (Leitch et al., 1994). In addition, direct management costs for chemical and biological control can exceed the productive value of the land, and presence of invasive plants directly reduces land value, contributing to the overall decline of rural communities (DiTomaso, 2000).

Management Recommendations- The management of invasive plant species is an ongoing process, though early detection and rapid response (EDRR) is the most cost-effective strategy. This includes prioritizing the defense of intact core habitats and removing small patches before they become unmanageable (DiTomaso, 2000). Strategies should be multi-faceted, integrating biological, mechanical, and chemical controls—including rotational grazing, mowing, herbicide, prescribed burning, and active reseeding. As with WPE removal, successful management and restoration of invaded grasslands requires long-term planning that accounts for soil nutrient legacies to prevent re-invasion (Daryanto et al., 2019; DiTomaso, 2000; Limb et al., 2014).

Plant Native Species

Native grasslands serve as carbon sinks and support biodiversity and general ecosystem health (Ghosh and Mahanta, 2014; Pierson et al., 2007). The widespread conversion and loss of native grassland habitat is associated with declines in grassland bird population abundance and nesting success (Coon et al., 2022; Haffele et al., 2013). Research identifies various methods of native grass seeding, planting, and integration in an effort to counteract the decline of native species in grassland ecosystems, and improve carbon storage, benefiting the ecosystem as a whole.

Birds

Some native seeding and planting practices have resulted in positive avian responses, while others have mixed outcomes, or require additional maintenance practices to elicit a beneficial response. Native seeding can be implemented through various techniques, resulting in varying levels of native plant establishment, and potential benefit to birds. Seeding with dense nesting cover (3-5 multi-species mixture of cool season vegetation) and restored native plantings (16-22 multi-species mixture of native plants) resulted in similar nesting densities, though nest survival was more stable on an annual basis with dense nesting cover (Haffele et al., 2013). A combination of native seeding and herbicide treatments in invasive-dominated grassland resulted in increased abundance, nesting, nest survival, fledglings,

and decreased parasitism of Dickcissel (Coon et al., 2022). Integrating strips of native prairie grasses in cropland doubled native bird species richness, and prompted other environmental benefits, such as increases in insect biodiversity, pollinator abundance, and ecosystem services, and decreased runoff (Schulte et al., 2017).

Carbon

Previous research has identified higher carbon storage/concentration in native grassland habitats as opposed to other cover types (Conant et al., 2017; Hernandez-Ramirez et al., 2021). Of the studies reviewed regarding the impact of native planting and seeding practices on grassland carbon storage, 2/3 displayed positive results. One study found restoration of solar array fields through native grass plantings increased carbon storage potential by 65%, and was beneficial for pollinator sourcing, and water and sediment retention (Walston et al., 2021). While these benefits are well documented, some applications of native seeding did not result in significant changes in soil carbon, especially the timing of seed application overlapped with an uncharacteristically dry period (Stricker and O'Connell, 2025). For this reason, seasonal precipitation forecasting should be taken into account. The effectiveness of native seeding to improve carbon storage can be increased by application method. For example, seed drilling of native warm season grasses was found to increase bulk soil carbon (Luong et al., 2025). Benefits can also be amplified by pairing seeding with other management practices, such as composting, which have been shown to increase CO₂ assimilation when implemented simultaneously (Phillips, 2023).

Co-benefits

Beyond the influence of native seeding and planting on birds and carbon, the practice offers various co-benefits to the broader ecosystem. Native seeding can improve overall ecosystem health through enhanced native species biodiversity and abundance (McFarlane et al., 2023; Scott, 2024; Trowbridge, 2015). Increases in native seeding rates were shown to improve establishment of native plant species populations and decreases in the prevalence of invasive plants (Avila-Sanchez et al., 2021; Dhar et al., 2025; Hall, 2013; Holfus, 2021; Porensky et al., 2018; Trowbridge, 2015). The success of native seeding and planting treatments can be amplified with specific methodologies or combined practices. For example, when combining native seeding and herbicide treatments, carbon-based seed coatings increased seed success by 43% (Avila-Sanchez et al., 2021; Holfus, 2021). Studies have also shown broadcast seeding to be an effective application to achieve long-term establishment

of native forbs (Gornish and Shaw, 2017). However, the implementation of supplemental practices should be conducted intentionally, as some have been found to have negative implications. For example, while some herbicide treatments improve annual grass abundance and decrease exotic species, native abundance decreases as well (Avila-Sanchez et al., 2021). Fertilizer application can also favor the succession of invasive species as opposed to native grasses (Gornish and Shaw, 2017).

While native planting and seeding practices can benefit grassland birds and increase carbon storage, results heavily depend on methodology. While none of the reviewed papers identified negative impacts of native planting and seeding on birds, benefits varied based on planting strategy and supplemental practices. Carbon storage displayed more consistent positive results, although certain seeding techniques were more effective than others. Beyond bird and carbon benefits, ecosystems displayed increased biodiversity, native plant abundance, and general ecosystem health with the implementation of these practices.

Control of Woody Encroachment

Grasslands face continued conversion pressure and degradation from the invasion of woody plant species. Woody plant encroachment (WPE) is the expansion of trees and shrubs into historically grass-dominated ecosystems. In the U.S., this process is particularly prevalent in desert grasslands, tallgrass prairies, oak savannas, and rangelands, where species such as western juniper (*Juniperus occidentalis*), honey mesquite (*Prosopis glandulosa*), common buckthorn (*Rhamnus cathartica*), Russian olive (*Elaeagnus angustifolia*), snowberry (*Symphoricarpos albus*), and eastern redcedar (*Juniperus virginiana*) displace native understory grassland vegetation and fragment grassland habitats (Bernath-Plaisted et al., 2023). Globally, removal of WPE in grasslands and savannas is promoted as a management practice to improve forage, soil stability, nutrient cycling, carbon sequestration, and overall ecosystem productivity (Ding and Eldridge, 2024).

However, not all woody biomass in grasslands should be managed similarly. WPE control frameworks must distinguish undesirable brush encroachment from the vital need to retain legacy native trees, such as open-canopy, low-density oaks in savanna systems. Protecting these legacy trees provides multiple benefits,

including mitigating the decline of native oaks and oak savannas, supporting birds, maintaining significant long-lived aboveground carbon storage with deep stable prairie-soil carbon, and providing essential shade for wildlife and grazers (Yantes et al., 2023).

Birds

Habitat alterations associated with WPE compromise the capacity of grassland ecosystems to sustain populations of grassland-obligate birds (Andersen and Steidl, 2023; Lautenbach et al., 2017). WPE fragments and degrades these grassland habitats (Bernath-Plaisted et al., 2023), and alters habitat use through facilitating an increase in the presence of shrub-associated predators and competitors; trees provide visual perches for hawks and owls and attract Brown-headed Cowbirds (*Molothrus ater*), a nest parasitic species. Even minimal woody encroachment leads to a significant decrease in the density of species like the Grasshopper Sparrow and Lesser Prairie-Chicken. The latter shows extreme avoidance of woody plants, selecting habitats nearly 300 meters away from the nearest trees; they are 40 times more likely to use habitat without trees compared to sites with just five trees per hectare (Lautenbach et al., 2017). Bird densities can drop by 50% or more when woody cover exceeds 15% (Johnson, 2019; Pulliam et al., 2020). In the Northern Great Plains, Baird's Sparrow (*Centronyx bairdii*) and Sprague's Pipit (*Anthus spragueii*) are primary "losers" when even minimal woody cover appears (Johnson, 2019; Pulliam et al., 2020).

Conversely, significant increases in useable habitat are achievable through efficient management interventions (Silber et al., 2024). WPE removal significantly boosts the density of declining specialists; for example, herbicide and shredding treatments increase densities of the Eastern Meadowlark (*Sturnella magna*), Henslow's Sparrow (*Centronyx henslowii*), and Grasshopper Sparrow (Lautenbach et al., 2020). Strategic conifer removal also increases useable space for Greater Sage-grouse (*Centrocercus urophasianus*) (Olsen et al., 2021) and benefits other ground and shrub nesting birds (Holmes et al., 2017). While WPE removal may negatively impact non-specialist grassland birds, a mosaic of habitat types and successional stages best supports regional biodiversity (Coffman et al., 2014; Frost and Powell, 2011). [GOT Birds](#), a conservation support tool to 'estimate how grassland birds may respond to woody plant management' can be used to estimate the effect of WPE removal on bird species regionally.

Carbon

The relationship between carbon and WPE is complex, and varies geographically based on climate, habitat type, and soil characteristics (Ding and Eldridge, 2024). WPE can lead to higher carbon stocks, especially aboveground (Asner et al., 2003; Cable et al., 2009; Hughes et al., 2006; Li et al., 2016; Mushinski et al., 2024; Neff et al., 2009), but there is a trade-off between aboveground biomass quantity and storage stability (Mushinski et al., 2024). WPE-sequestered carbon is often unstable; aboveground carbon can be released quickly due to climate change, fire, or habitat clearing (Creamer et al., 2011).

Additionally, WPE can alter soil biology and chemistry, offsetting carbon gains from leaf litter with microbial activity that accelerates mineralization of organic matter (Creamer et al., 2012; McCulley and Jackson, 2012; Puttock et al., 2013). In the long run, native grasslands can hold up to twice as much total carbon as encroached soils (Gan and Hu, 2025; McCulley and Jackson, 2012) because grasses invest in deep, perennial root systems (McCulley & Jackson, 2012). While removal of woody plants significantly reduces aboveground carbon, the loss is often compensated by increased root carbon as grasses recover (Abdallah et al., 2020). These belowground carbon pools are more stable (Creamer et al., 2011), and mineral-bound carbon remains a long-term benefit after WPE removal (DeMarco et al., 2016). Consequently, total ecosystem carbon stocks often remain unchanged post-WPE-removal because the vast majority of carbon (84–97%) is stored belowground in roots and soil (Abdallah et al., 2020; Ding and Eldridge, 2024). Thus, removing woody biomass often results in negligible long-term carbon loss as native grasses stabilize mineral-bound carbon pools (Abdallah et al., 2020).

Co-benefits

Water- WPE fundamentally alters the landscape's hydrology, as trees and shrubs draw significantly more water from groundwater than native grasses, which decreases water availability through increased transpiration (Ding et al., 2024; Scott et al., 2006). In riparian areas, this results in reduced streamflow and compromised watershed health (Keen et al., 2023). Restoration through WPE removal can reverse these effects, restoring water flowing in streams and hydrological connectivity (Sankey et al., 2021). Furthermore, because shrublands are nearly as water-intensive as dense woodlands, even moderate encroachment can deplete water resources without providing a proportional carbon storage benefit (Scott et al., 2006). WPE can lead to a mosaic of shrubs and

bare soil patches, creating increased runoff into waterways, erosion, and soil loss (Puttock et al., 2013). Removing WPE restores water quality, as a healthy grassland understory captures sediments, and improves overall grassland ecosystem watershed health (Reisinger et al., 2013). Long-term studies indicated that maintaining open grasslands can help reduce sedimentation and help retain carbon storage for up to 60 years post treatment (Sankey et al., 2021).

Soil Health- While WPE can support high nitrogen and soil nutrient pools under shrub canopies — creating “islands of fertility” — these can also lead to a leak in the nitrogen cycle. For example, nitrogen-fixing species like honey mesquite can increase emissions of nitric oxide (NO), a potent atmospheric trace gas, potentially impacting long-term soil fertility (Martin et al., 2003). Restoration through WPE removal shifts nitrogen storage back into more stable belowground pools (Abdallah et al., 2021) and encourages “grass-friendly” microbial communities that balance the nutrient cycle and improve native grassland health (Creamer et al., 2012; Veach et al., 2015). Soil health typically bounces back quickly after removal, characterized by an increased fungal-to-bacterial ratio, faster nitrification (making nitrogen available to plants), and reduced nutrient loss (Dickens et al., 2015; Veach et al., 2015).

Biodiversity- Grassland restoration through the removal of WPE triggers a multi-taxonomic biodiversity rebound by restoring the structural complexity and open niches required by grassland specialists. Mechanical and chemical treatments used in removal of WPE also “release” the native seed bank, significantly increasing understory plant richness and cover as suppressed grasses and forbs regain access to sunlight and soil nutrients (Brudvig and Asbjornsen, 2009). While total removal is essential for “obligate” bird specialists, maintaining a mosaic of successional stages across the landscape may best support broader regional biodiversity for other taxa by providing additional habitat for facultative species that utilize both open and shrub-influenced habitats (Coffman et al., 2014; Frost and Powell, 2011). Ultimately, targeted removal offers high ecological returns by reconnecting fragmented corridors and restoring the “thermal mosaics” necessary for reptiles and mammals (Ding and Eldridge, 2024; Olsen et al., 2021).

Productivity- WPE consistently degrade the economic output of grasslands and rangelands through reduced forage production (Anadón et al., 2014; Ding et al., 2024). As WPE increases, the understory is shaded out, depriving high-quality native forage — specifically C4

grasses — of the direct sunlight they need. This competition for light reduces plant richness and biomass, eventually leading to an increase in bare ground (Puttock et al., 2013). WPE also depletes soil moisture, leaving grasses in a state of drought stress even during years of average rainfall (Scott et al., 2006). The accumulation of woody leaf litter also alters soil chemistry and creates physical barriers that inhibit the germination of native seeds (Creamer et al., 2011). These ecological shifts have direct financial consequences: for every 1% increase in WPE, there is a measurable reduction in livestock production of up to 1.6 reproductive cows per square kilometer in high-productivity areas (Anadón et al., 2014). In addition, juniper species harbor livestock pests like cedar flies and ticks, increasing livestock stress and disease transmission (Noden et al., 2021). Lastly, the depletion of surface water resources could negatively impact water availability for livestock.

Management Recommendations- To maximize both ecological resilience and cost-efficiency, land management strategies should prioritize “defending the core” by targeting early-stage encroachment (< 10% cover) before ecosystems cross critical biotic and abiotic thresholds (Ding & Eldridge, 2024). Early intervention maintains the native seed bank and prevents the alteration of soil chemistry and bird community structures that make late-stage restoration prohibitively expensive (Limb et al., 2010).

Effective management requires a multi-faceted strategy that integrates immediate removal with long-term stabilization. Once a landscape has transitioned into a closed-canopy shrubland or woodland, reversing the resulting state-shift is significantly more difficult (Ding and Eldridge, 2024; Limb et al., 2014). Prescribed fire remains the most efficient practice for broad-scale control, as it eliminates woody saplings before they develop the size and bark thickness necessary to survive a burn (Limb et al., 2010). However, removal of WPE should be paired with the management of invasive grasses, such as tall fescue, to ensure that bird nesting success is not undermined by continued habitat degradation. Mechanical removal should also consider soil compaction impacts and impacts to existing understory plants. In addition, restoration should include a long-term plan involving rotational grazing with appropriate stocking rates (Van Auken, 2009), active reseeding, and prescribed fire is essential to ensure that cleared sites do not remain bare or susceptible to re-invasion (Daryanto et al., 2019; Limb et al., 2014).

Reduce Insecticide Use

There is growing awareness around the widespread use of insect pesticides such as neonicotinoids, organophosphates, carbamates, pyrethroids, and ivermectin across landscapes and the negative consequences on ecosystems. Neonicotinoid insecticides, systemic compounds absorbed by plants and distributed throughout their tissues, are typically applied as seed treatments to common row crops such as corn, canola, and barley (Schepker et al., 2020). These compounds are water-soluble, allowing them to persist in soil and leach into waterways, and directly and indirectly impact non-target species. They account for ~25% of all insecticide use worldwide and are currently used in >120 countries (Jeschke et al., 2011). The insecticide is designed to be absorbed by the crop plant to deter herbivores, but only ~5% of the chemical is absorbed while the rest remains in the soil, enters the air as dust, or runs off into surface waters (Goulson, 2013; Main et al., 2016). As a result, neonics are ubiquitous in agricultural landscapes, with detections in up to 91% of sampled wetlands (Main et al., 2014; Morrissey et al., 2015). There they may remain active for years, with reported half-lives of 200 – 1000+ days (Goulson, 2013).

Similarly, ivermectin, widely used to control invertebrate pests in livestock, also poses direct and indirect risks to non-target species (Pecenka and Lundgren, 2019). Pesticide drift or run-off from treated areas can impact non-target species through direct and indirect pathways, altering survival, reproduction, and trophic system dynamics (Li et al., 2020; Graves et al., 2023; Molenaar et al., 2024). Although much of the existing literature on neonicotinoid pesticide use focuses on cropland systems rather than grasslands or rangelands, these agricultural lands typically occur within a grassland and/or rangeland matrix. Pesticides are widely applied in agricultural landscapes, often in proximity to natural grasslands (Main et al., 2014) and other habitats used by grassland birds throughout their nonbreeding ranges (Bernath-Plaisted et al., 2023). Because the vast majority of the chemicals applied remain in the soil, drift through the air, or runoff in surface water (Goulson, 2013; Main et al., 2016), their impacts extend far beyond agricultural systems.

Birds

Birds can be impacted by pesticides through both direct and indirect pathways. Direct exposure may occur when birds consume treated seeds or contaminated prey (Gibbons et al., 2015). The severity of these effects varies by pesticide type, concentration, and environmental

conditions. Although non-neonicotinoid pesticide use has decreased in the last two decades (DiBartolomeis et al., 2019), the use of neonicotinoids has increased and is expected to continue to increase (Zhang et al., 2023) despite being applied in lower amounts given their potency. Neonicotinoid pesticides are far more potent and persist in the environment for longer than many other pesticides (Mineau and Palmer, 2013). Per-unit effects on grassland bird abundance was estimated to be more than 40 times greater than those of non-neonicotinoid pesticides at the county level in the US (Li et al., 2020); one neonicotinoid coated corn kernel is enough to kill a bird equivalent in size to a Blue Jay (*Cyanocitta cristata*) (Mineau and Palmer, 2013). While organophosphate and carbamate pesticide use for grasshopper control has been found to have little impact on breeding bird communities in western rangelands (George et al., 1995), another study found Horned Lark (*Eremophila alpestris*), Western Meadowlark (*Sturnella neglecta*), and Thick-billed Longspur (*Rhynchophanes mccownii*) numbers decreased in areas treated with higher volumes of more toxic pesticides but remained stable in areas without treatment or lower toxicity (McEwen et al., 1975). Environmental impacts depend on pesticide toxicity and exposure, including the extent of the treated area and persistence of the chemical in the environment (Norelius and Lockwood, 1999). In some cases, pesticide use on grazing lands may be avoided by maintaining or improving range conditions, as overgrazed or drought-stressed systems are more susceptible to insect outbreaks that prompt chemical control (McEwen et al., 1972).

Reducing insecticide use can benefit birds by supporting healthier invertebrate communities and maintaining habitat structure (Gibbons et al., 2015). Greater insect abundance increases food availability for insectivorous species and reduces the likelihood that birds consume contaminated prey (Kraus et al., 2021). Reduction of neonicotinoid use is particularly important for birds such as aerial insectivores and other songbirds that rely on aquatic emergent insects as prey, as the extensive runoff and long persistence of these insecticides in surface water reduces the availability of key prey such as chironomid midges (Barmantlo et al., 2021; Cavallaro et al., 2018). These improvements to prey availability and food web integrity can help support bird populations, particularly in grassland ecosystems where many species rely heavily on insects. To better inform this review, we considered studies conducted in regions where neonicotinoid pesticide use is more regulated, such as the European Union (Donley, 2019). For

example, a recent study in France found that bird responses following the neonicotinoid ban varied by diet and time since the ban, suggesting pesticide bans alone may not produce immediate or full population recovery, likely due to persistence in the environment or bioaccumulation (Perrot et al., 2025). These findings provide useful context for U.S. grassland and rangeland systems where grassland birds are experiencing long-term declines and similar evaluations of pesticide reduction remain limited. Together, this evidence highlights the ecological consequences of continued pesticide use and the need for proactive research and policy consideration.

Carbon

The impact of reducing insect pesticide use on carbon dynamics remains poorly understood. While existing literature demonstrates insecticides can alter soil microbial communities and invertebrate populations, key drivers of carbon cycling, there is a lack of field-based evidence linking these effects to changes in soil organic carbon (Pecenka and Lundgren, 2019; Torabian et al., 2024).

Co-benefits

Pesticide use impacts biological processes linked to decomposition and nutrient cycling (Pecenka and Lundgren, 2019) and dung beetle (*Scarabaeidae*) abundance (Torabian et al., 2024). Ivermectin causes direct mortality, reduced reproduction, and developmental delays in dung-dwelling keystone species and beneficial insects. Indirectly, it disrupts ecosystems by impacting decomposition rates, harming predators and parasitoids that feed on intoxicated insects, and threatening aquatic life via runoff (Verdú et al., 2018). Torabian et al. (2024) found dung beetle abundance to be 50% greater in dung pats without ivermectin, which have important implications for nutrient cycling, improve soil microbial diversity, and regulate greenhouse gas emissions (Pecenka and Lundgren, 2019; Torabian et al., 2024). If a management system reduces ivermectin use, the ecosystem services they provide are enhanced, reducing the reliance on chemical products in ecosystems and reducing economic costs to the rancher (Pecenka and Lundgren, 2019). Other co-benefits of neonicotinoid and pyrethroid pesticide reduction include increased wild bee occurrences, butterfly abundance, and butterfly species richness (Guzman et al., 2024; Van Deynze et al., 2024).

Research on insect pesticide reduction in U.S. grasslands and rangelands remains limited, while the impacts of

pesticide use on birds in cropland systems are well studied. This creates a knowledge gap at the intersection of agrochemical management, biodiversity conservation, and climate mitigation in landscapes across the U.S., highlighting the need for field-tested research that directly measures ecological responses to pesticide reduction in grassland and rangeland landscapes.

Prescribed Fire

Fire has been an essential process in the maintenance of grassland ecosystems in the Great Plains for thousands of years, and Indigenous communities actively shaped these fire regimes through deliberate burning practices that sustained prairie structure and function long before European settlement (Huffman, 2013). Prescribed fires can effectively be used to mimic historical fire regimes, but the present-day application of fire in grasslands and rangelands comes with many complexities, including climate change impacts on fuels, risks to people and property, and the ability to use fire at appropriate scales to ensure habitat heterogeneity.

Birds

Evidence from multiple studies illustrates that fires can benefit some species and not others. Some species, such as Grasshopper Sparrow, Western Meadowlark (*Sturnella neglecta*), Horned Lark, and Vesper Sparrow (*Pooecetes gramineus*), frequently colonize burned areas and show increased densities or species richness in the years following fire (Grant et al., 2010; Madden et al., 1999; Petersen and Best, 1987). In contrast, species like Clay-colored Sparrow (*Spizella pallida*) and Common Yellowthroat (*Geothlypis trichas*) often reach highest abundance in unburned prairie, highlighting the importance of retaining unburned refuges (Madden et al., 1999; Reynolds and Krausman, 1998). Responses can also be temporal: initial declines in bird numbers or reproductive success may occur in the first-year post-burn, followed by stabilization or increases in subsequent seasons (Grant et al., 2010; Skagen et al., 2018). Predation and brood parasitism further shape these outcomes: burned areas may increase arthropod prey availability, but they can also elevate predator presence, sometimes reducing nest success for ground-nesting birds like the Grasshopper Sparrow or Dickcissel (Rahmig et al., 2009; Shochat et al., 2005).

Potential responses to a burn depend on factors such as predator presence, species-specific habitat preferences, and food availability for birds. Prescribed burns could be paired with monitoring to determine whether further

practices such as increased predator controls are necessary.

Carbon

Prescribed fire in western U.S. rangelands releases carbon into the atmosphere in the short term, primarily through the combustion of aboveground vegetation. However, multiple lines of evidence show that these emissions are offset by longer-term benefits to ecosystem stability and carbon retention. First, prescribed fire has smaller negative effects on carbon cycling than wildfire (Xu et al., 2022). In fact, the largest and most stable carbon pool in these ecosystems is soil carbon, which is only weakly affected by prescribed fire but much more negatively impacted by high-severity wildfire (Rau et al., 2012; Xu et al., 2022). Fire suppression results in poorer long-term carbon storage, as fuel accumulation results in negative feedback where excessive stored carbon eventually becomes a source when severe fires occur (Girod et al., 2007). After fire, much of the carbon fixed by regrowing vegetation is allocated belowground to roots, helping to rebuild soil carbon (Fellows et al., 2018). Prescribed fire also helps regulate nitrogen availability by accelerating plant nitrogen uptake and reducing nitrogen losses as potent greenhouse gases such as nitrous oxide, thereby improving the overall greenhouse gas balance of rangelands under increasing nitrogen deposition (Gomez-Casanovas et al., 2016). These findings appear to be specific to rangelands with low tree densities, as carbon losses in rangelands over 26% tree cover become much more severe (Rau et al., 2012). Note that this threshold reflects the dynamics of western rangeland carbon pools.

Co-benefits

Beyond carbon and bird outcomes, prescribed fire offers meaningful co-benefits for plant community composition, particularly through suppression of invasive species and restoration of native vegetation. Fire can reduce invasive annual grasses such as cheatgrass and medusahead, though outcomes depend heavily on burn season and pre-fire community condition, with native-dominated sites and fall burns generally showing better native plant retention (Berleman et al., 2016; Ellsworth and Kauffman, 2017). Throughout much of the Great Plains, fire is a highly effective tool to prevent woody plant invasion, which is a major threat to millions of acres of native grasslands. In oak savanna and woodland systems, prescribed fire is the primary restoration lever for reversing fire suppression-driven WPE and recovering characteristic structure and groundlayer diversity (Peterson and Reich, 2001). When combined with grazing (pyric herbivory),

prescribed fire has shown sustained long-term suppression of invasive forbs like *Sericea lespedeza* (*common buckthorn*) at operational management scales, outperforming herbicide applications alone (Sherrill et al., 2022). Together, these plant community outcomes support broader biodiversity goals and increase ecosystem resilience. However, single burns are rarely sufficient. Repeated fire, strategic timing, and integration with other practices such as managed grazing are often necessary for durable plant community outcomes (Havstad and James, 2010; Toledo et al., 2026).

Prescribed fire also provides important co-benefits for producer livelihoods and community safety. Spring prescribed burns can improve forage nutritive value and cattle weight gains, offering direct economic returns to ranchers managing livestock on native rangeland (Bernardo et al., 1988). By reducing accumulated fuel loads, strategic burns also lower the probability and potential severity of uncontrolled wildfires and protect producer income from the substantial economic losses associated with wildfire (Bergtold et al., 2024; Twidwell et al., 2013). This wildfire risk reduction is particularly consequential in rangeland landscapes adjacent to the Wildland Urban Interface, where prescribed fire is increasingly recognized as one of the strongest practical justifications for maintaining active burn programs (Twidwell et al., 2013).

Patch-burn Grazing

Heterogeneity-based grassland management approaches, defined by the intentional creation of spatial and temporal variability in disturbance, vegetation structure, and resource availability, are increasingly recognized as essential for sustaining biodiversity and ecosystem function in working grasslands, with patch-burn grazing (PBG) representing the most extensively studied and widely implemented example of this management paradigm (Duquette et al., 2022; Engle et al., 2003; Weir et al., 2013). PBG spatially and temporally couples fire and grazing and creates a shifting mosaic of recently burned, intensively grazed patches alongside lightly grazed and recovering areas without the use of fencing. Livestock preferentially select recently burned patches due to higher forage nutritive value, resulting in concentrated grazing pressure within burned areas and landscape-scale heterogeneity in vegetation structure (Cook, 2008; Gasch et al., 2020a; Gerhard et al., 2022; Spiess et al., 2024).

Birds

For birds, numerous studies show that PBG increases avian beta diversity by supporting distinct bird community assemblages across patches within a pasture, relative to homogeneous management; however, grassland specialist responses to PBG vary. While some studies show species such as Mountain Plover (*Charadrius montanus*), Upland Sandpiper (*Bartramia longicauda*), Greater Prairie-chicken (*Tympanuchus cupido*), and Lesser Prairie-chicken can benefit from PBG, others show negative impacts on species that require tall or dense vegetation structure for nest concealment, such as Lark Bunting, Western Meadowlark and Grasshopper Sparrow. In one study, Thick-billed Longspur and Horned Lark abundances were unchanged in response to PBG (Augustine and Derner, 2015; Griffis and Campa, 2023; Holcomb et al., 2014; Kraft et al., 2021; Winder et al., 2018, 2017). Although PBG does not universally increase nest survival or abundance, it improves habitat availability and breeding site selection by maintaining a diversity of vegetation structures across years (Erickson, 2017; Rischette et al., 2023; Sipe, 2023). Fragmentation, high stocking rates, predation, parasitism, and drought can constrain or offset these benefits, underscoring the importance of landscape and management context (Pillsbury et al., 2011; Verheijen et al., 2022). In highly fragmented landscapes, low stocking rates and periodic rest from grazing in unburned areas may be required to achieve a diverse bird community (Augustine and Derner, 2015; Pillsbury et al., 2011).

Carbon

Carbon outcomes under PBG are more nuanced but generally neutral to positive when evaluated within intact grassland systems. Evidence indicates that PBG primarily affects aboveground processes and vegetation composition, with limited or no negative impacts on soil

carbon stocks (Gerhard et al., 2022; Sipe, 2023; Spiess, 2021). By maintaining perennial grass cover and promoting plant functional diversity, heterogeneity-based grazing aligns with broader findings that improved grassland management can maintain or enhance soil organic carbon relative to degraded or converted systems (Conant et al., 2017; Ren et al., 2024; Sanford et al., 2022). Light to moderate grazing intensities within PBG systems appear particularly compatible with carbon retention, whereas heavy or poorly timed grazing can reduce sequestration (Zhou et al., 2017).

Co-benefits

Beyond birds and carbon, PBG delivers multiple co-benefits. It improves forage quality often increasing livestock performance while reducing the need for supplemental feed and herbicides (Scasta et al., 2016; Sipe, 2023). PBG is generally effective for controlling invasive grasses such as Kentucky bluegrass and tall fescue while maintaining native plant diversity (Gasch et al., 2020a; Gerhard et al., 2022; Kjaer, 2025; McGranahan, 2008), although a reduction in invasives has not been universally demonstrated (Duquette et al., 2022). Additional benefits include increased vegetation beta diversity

and structural heterogeneity, microbial abundance, ecosystem function, drought resilience, and support for ecological processes that historically structured North American grasslands (Duquette et al., 2022; Ricketts and Sandercock, 2016; Scasta et al., 2016; Spiess, 2021; Weir et al., 2013).

Geographic Coverage of Literature Review

Practice	**States (within main focus area of report)**	**Other Regions**
Protect Wetlands from Grazing	Wyoming, Idaho, Oregon, California, Pennsylvania, Nevada, Utah, North Dakota, South Dakota, Missouri	Two meta-analyses
Control of Invasive Plant Species	Iowa, Missouri, California, North Dakota, Great Basin (Nevada, Utah, Oregon, Idaho, California), Texas, Wisconsin, Montana, South Dakota, Kansas, Nebraska	New Zealand, one US meta-analysis, one global meta-analysis
Plant Native Species	New Mexico, Nevada, California, North Dakota, Indiana, Illinois, Missouri, Virginia, Idaho, Iowa, Wisconsin, Washington, Oregon, Michigan, Minnesota, Colorado, Utah	One global meta-analysis, one US review, Alberta, British Columbia, Oregon, Washington, California, Russia
Control of Woody Encroachment	Arizona, Idaho, New Mexico, Oregon, Kansas, Nebraska, Oklahoma, Texas, Ohio, Midwest, Great Basin (Nevada, Utah,	Four meta-analyses: one lower 48 US, three global

Geographic Coverage of Literature Review		
Practice	**States (within main focus area of report)**	**Other Regions**
	Oregon, Idaho, California), Southern High Plains (Texas, Oklahoma, Kansas).	
Reduce Insecticide Use	Wyoming, Montana, New Mexico, North Dakota, South Dakota, Utah, Colorado, Idaho, Michigan, Wisconsin, Ohio, Iowa, Illinois	France, one meta-analysis
Prescribed Fire	Nevada, Wyoming, Oklahoma, North Dakota, Idaho, Texas, Colorado, Kansas, California, New Mexico	Two meta-analyses, contiguous US
Patch-burn Grazing	Wisconsin, Minnesota, Colorado, North Dakota, South Dakota, Idaho, Nebraska, Iowa, Kansas, Oklahoma, Montana, Wyoming, Texas, California	Georgia, Florida, Tennessee, Louisiana, Maine, New York, British Columbia, Saskatchewan, North America and China meta-analysis
Rotational Grazing	North Dakota, Colorado, Montana, Nebraska, Texas, South Dakota	Four global reviews, Three global meta-analyses, Canada

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