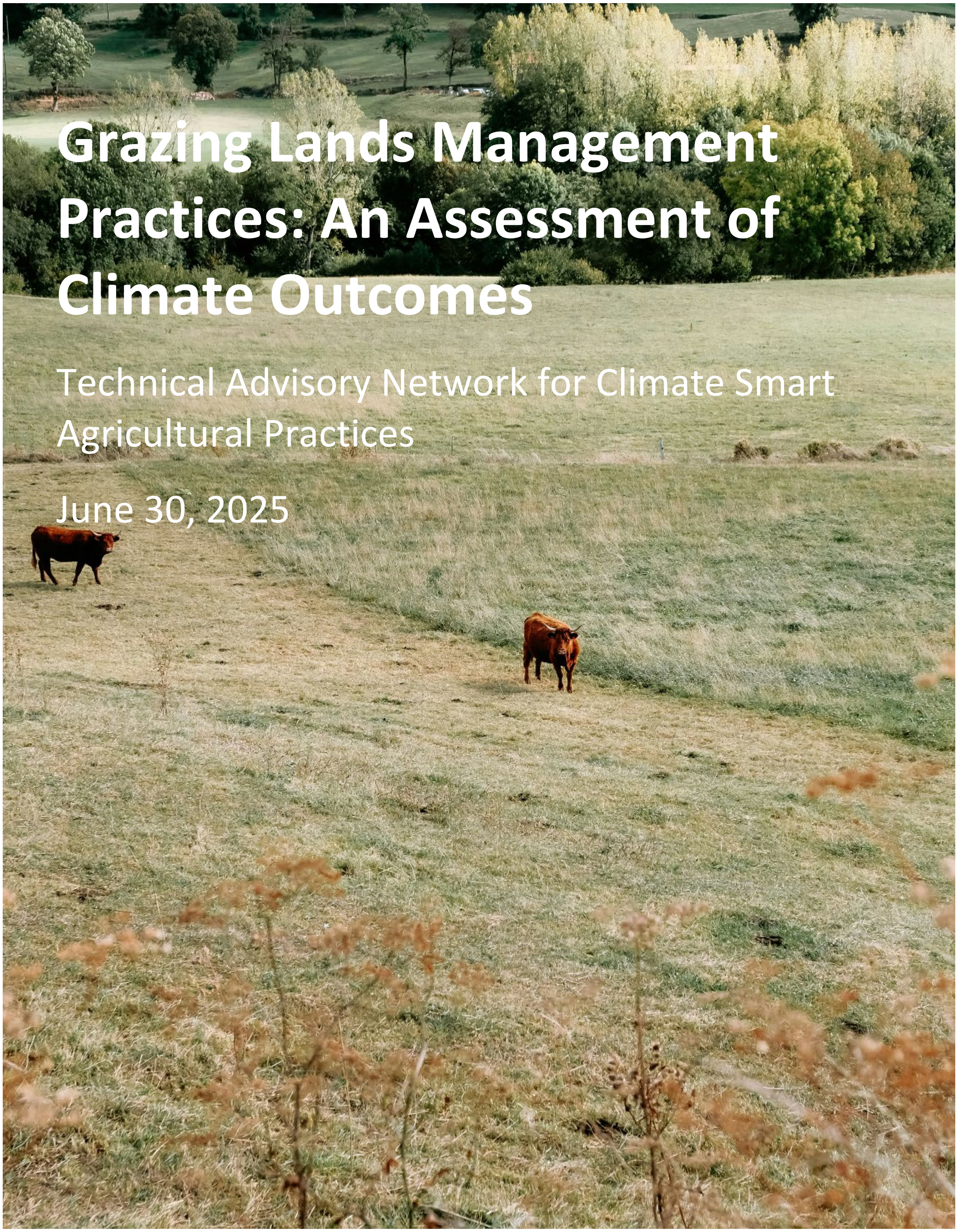

Grazing Lands Management Practices: An Assessment of Climate Outcomes

Technical Advisory Network for Climate Smart Agricultural Practices

June 30, 2025

About the Technical Advisory Network for Climate Smart Agricultural Practices

The Network emerged from a project of the Meridian Institute to better align Natural Resource Conservation Service (NRCS) Conservation Practice Standards with the best available science on GHG mitigation. In May 2024, Meridian Institute assembled six technical work groups (TWGs) to provide up-to-date scientific analysis of the impact of conservation practice adoption on GHG emissions and make technical recommendations regarding how best to achieve climate mitigation through those practices. The TWGs were comprised of academic scientists who are well regarded in their chosen field as well as scientists working within nongovernmental organizations focused on one or more of the chosen categories of interest: nutrient management, manure management, enteric methane, soil carbon, agroforestry, and grazing lands.

The technical work groups first identified agricultural practices in the six areas of interest for which evidence indicates considerable potential to reduce GHG emissions based on raw mitigation potential and adoption considerations. In August 2024 Meridian entered into a Contribution Agreement with NRCS to conduct literature reviews of the GHG emissions impacts of 39 practices, individually and in commonly practiced combinations. Meridian staff and TWG chairs and members worked with the NRCS National Discipline Leads to scope the reports and refine technical guidance regarding how best to implement practices to achieve GHG mitigation. The TWGs then conducted literature reviews, first and second-order meta-analyses, and modeling scenarios to identify the circumstances in which there is strong scientific evidence that the agricultural practices identified are likely to have net GHG benefits as well as implementation guidance for maximizing those benefits. The TWGs generated more than a dozen reports, which were delivered to NRCS in June 2025.

With the closing of Meridian Institute in mid-2025, the six technical working groups assembled by Meridian decided to establish the Network to provide a platform for sharing the work undertaken with a diversity of interested stakeholders and to support future collaboration within and/or across technical working groups. Slightly modified versions of the reports delivered to NRCS, such as this report, have been issued publicly by the Network and findings from several groups will be published in academic journals. The Network may also create briefs to summarize findings and technical recommendations for field conservationists and other interested stakeholders.

WORKING GROUP	CHAIRPERSON	PRIMARY AFFILITATION	EMAIL
Agroforestry	Dr. Nate Lawrence	Savanna Institute	nate@savannainstitute.org
Enteric Methane	Dr Ermias Kebreab	University of California, Davis	ekebreab@ucdavis.edu
Grazing Lands	Dr. Jennifer Watts	Woodwell Climate Research Center	jwatts@woodwellclimate.org
Nutrient Management	Dr. Justin Baker	North Carolina State University	jsbaker4@ncsu.edu
Manure Management	Dr. Rebecca Larson	University of Wisconsin-Madison	rebecca.larson@wisc.edu
Soil Carbon	Dr. Bruno Basso	Michigan State University	basso@msu.edu

AUTHORS

Dr. Jennifer Watts, Systems Ecologist, Rangeland Program Co-Director, Woodwell Climate Research Center; Affiliate Professor, Montana State University; TWG Chair

Dr. Sasha Gennet, Senior Science and Policy Advisor, The Nature Conservancy; Visiting Researcher, University of California, Berkeley

Dr. Corrine N. Knapp, Associate Professor of Human Ecology, Center for Rural Resilience & Innovation, Haub School of Environment & Natural Resources, University of Wyoming

Dr. Jocelyn Lavallee, Senior Agricultural Soil Scientist, Environmental Defense Fund

Dr. John Ritten, Agricultural Economist, Director of Grazing Lands Research and Engagement, AgNext, Colorado State University

RESEARCH ASSISTANTS

Dr. Kris Hulvey, Ecologist and Lead Scientist, Working Lands Conservation

Dr. Megan Nasto, Soil Science Program Director, Working Lands Conservation

Elana Feldman, Rangeland Ecologist, Working Lands Conservation

Dr. Samuel Willard, Soil Scientist, University of Pennsylvania & The Nature Conservancy

Crystal Toureene, Research Associate, Colorado State University

MERIDIAN TEAM

Tori Anderson, Mediator and Program Associate, TWG lead

Sarah Beck, Project Associate and Ruckelshaus Fellow

Clare Burke Ravizza, Project Associate and Ruckelshaus Fellow

Citation: Watts, Jennifer, Sasha Gennet, Corrine N. Knapp, Jocelyn Lavallee, John Ritten, Kris Hulvey, Megan Nasto, Elana Feldman, Samuel Willard, and Crystal Toureene. "Grazing Lands Management Practices: An Assessment of Climate Outcomes." Technical Advisory Network for Climate Smart Agricultural Practices, June 30, 2025. doi.org/10.5281/zenodo.15750303.

DOI: doi.org/10.5281/zenodo.15750303

Disclaimer: The views and opinions expressed in this report are those of the listed authors and do not necessarily reflect the views or positions of all participants in the Technical Advisory Network for Climate Smart Agriculture Practices.

Table of Contents

Executive Summary

This working group assessed the climate outcomes of several important management and conservation practices used widely on U.S. grazing lands. These practices included prescribed grazing, brush management, prescribed burning, pasture and range plantings, herbaceous weed treatment, and riparian/wetland restoration. The focus of the assessment was on ecosystem carbon storage and greenhouse gas (GHG) fluxes. The report also describes the potential co-benefits, ecosystem services of these practices, and considerations for implementation. Practices were reviewed individually and were not considered in combination, with the exception of herbaceous weed treatment.

We searched peer-reviewed publications using online search engines and databases to find relevant experiments and literature, assessed and synthesized the relevant findings of those papers, and added findings from some meta-analyses. Where needed to fill data gaps, we also include and cite ecological theory. Several expert interviews were also conducted; these researchers advised us on key readings and the current state of scientific knowledge for specific practices.

Climate mitigation from rangeland and pasture management is a potentially significant opportunity. We found that for several widely used land management practices, including prescribed grazing, plantings, and restoration, there are likely net long-term beneficial outcomes for GHG fluxes and ecosystem carbon. However, we found that there may also be tradeoffs between climate and conservation goals for some practices, and that the ecological context of practice implementation may have strong effects on its long-term outcomes. There were few practices with unequivocal and strong beneficial climate outcomes and no tradeoffs. To ensure maximum benefits with minimum costs and tradeoffs, the best data should be used and considered in the complex context of these socio-ecological systems.

The ecological heterogeneity of rangelands and pasture lands in the U.S., inconsistent research methodologies, and relatively recent interest in GHG and ecosystem carbon limit the amount and robustness of the current evidence for assessing these outcomes. Existing models, which also depend on input data, are valuable in some instances but insufficiently robust to assess many locales and practices. We urge the scientific community to coordinate on development and deployment of more long-term monitoring and measurement standards and projects for these ecosystems to understand and be able to optimize outcomes for multiple values including climate mitigation.

1. Introduction

1.1 Purpose

The primary purpose of this report is to synthesize the existing peer-reviewed published evidence for the climate-related outcomes of grazing lands management practices in the U.S. This information is intended to support decision-making by agencies, funders, non-profit organizations, and producers; identify gaps and needs for additional research; and advance conversations related to the climate implications of grazing land practices.

We refer to the Natural Resource Conservation Services (NRCS) Conservation Practice Standards (CPS) within this report as they are the primary mechanism for incentivizing conservation actions on private lands through Farm Bill funding and have been defined and standardized at a national level. This standardization allows for communication of practices in ways that are both transparent and actionable. The scope of this report is limited to rangeland and pastureland CPS. Rangelands are defined as lands whose native vegetation is primarily grasses, grass-like species, forbs and shrubs suitable for grazing and browsing and pasturelands are more agronomically managed lands, relative to rangelands, often with domesticated forage species and used for hay production and/or grazing. Focus was not given to grazing within croplands and heavily forested systems. We included CPS practices that are either currently defined as “climate smart,” proposed as “climate smart,” or where there was interest from the NRCS in exploring the evidence as to whether they are or are not “climate smart”. Our assessments do not include a full life-cycle assessment of GHG impacts (e.g., sourcing/producing, transporting). While we refer to associated CPS, for some practices scientific evidence does not clearly align with CPS as defined by the NRCS and in those cases we broaden the scope of the practice beyond what is stated in the CPS and note this within the text. This report synthesizes and reports the data that builds evidence for the climate implications of each included CPS (314, 315, 338, 512, 528, 550, 657). Assessments are made based on individual practices, with mention of practice combinations when appropriate. The authors refrain from making direct policy recommendations about the conservation practice standards although we believe that this assessment of evidence can inform future policy.

This report may be used for additional purposes: as a synthesis of literature for specific practices of interest, an assessment of the strength of evidence across practices, or a resource for identifying research needs to inform future policy. For readers interested in specific practices, the Table of Contents provides a list to navigate directly to practices of interest. For readers interested in broad take-homes related to climate implications of grazing lands practices, the conclusion summarizes these results. For readers interested in assessing gaps and research needs, these are addressed in each individual practice as well as summarized in the conclusion. We hope that this report will provide guidance on current scientific evidence for each practice as well as areas for future research.

1.2 Overview of U.S. Grazing Lands

In the United States (U.S.), grazing lands, defined as lands capable of providing forage for domestic livestock, are generally categorized as either rangelands or pasturelands (NRCS, 2003, Rinehart, 2008). Rangelands are typically referred to as more “natural” or “semi-natural" ecosystems because they contain native shrubs, forbs and/or grasses. Pasture typically refers to lands containing primarily non-native perennial or annual plants, that are often more agronomically managed using, e.g., irrigation, mowing, and fertilizer. In addition, many pastures, especially in the Eastern U.S., may have a history of cultivation before conversion to perennial vegetation.

Rangelands cover approximately 406 million acres in the contiguous U.S. (22% of land area), and pastures represent another 121.1 million acres (6% of land) (USDA, 2024; **Figure 1**). A majority (~65%) of rangelands are privately owned (Robinson et al., 2019). Approximately 46 million acres of rangelands are under tribal management, typically classified as either trust land or fee land (Rangelands Gateway, n.d.). Most of the remainder are federally owned, primarily managed through the U.S. Forest Service (USFS) and Bureau of Land Management (BLM) as grazing leases. Federally owned rangelands are more common in the Western U.S., whereas pasturelands are predominantly privately owned and may overlap with active cropping systems (e.g., hayed pastures; Allen et al., 2011). On U.S. public lands, cattle are currently the dominant form of livestock; however, sheep and goats are also grazed, and recently (in Montana) the BLM approved grazing of bison (Eggert, 2022).

Livestock production in the U.S. remains a large economic industry, with cattle contributing >$80 billion in 2023 annual revenue – 17% of total cash receipts for agricultural commodities (Knight, 2023). The economic value of these lands, considering forage production, wildlife habitat, conservation of biodiversity, etc., has been estimated at >$24 billion/year (Maher et al., 2021). Grazing lands (rangelands and pastures) provide a suite of invaluable ecosystem services, including wildlife habitat, biological biodiversity, water storage, forage for cattle, and climate mitigation including sequestration of CO_2 from the atmosphere through plant photosynthesis, and storage of carbon in belowground soil systems (Sala et al., 2017). These ecosystem services, including carbon storage and greenhouse gas (GHG) mitigation, remain largely unquantified, yet could easily exceed $100 billion (Havstad et al., 2007; Sala et al., 2017). Additional benefits provided by these lands include support for rural economies and food supplies, and cultural/spiritual experiences (Brunson et al., 2022). For instance, the unquantified benefits of rangelands for many Native American individuals and nations include deep cultural, spiritual, and historical significance and serve as the foundation of Indigenous identity, traditional practices, and ecological stewardship.

Grazing lands managers may adopt or reject conservation practices based on value alignment, perceptions of benefits, perceived risk, and costs. Managers have diverse values and priorities, leading to tradeoffs for ecosystem services. For instance, wildlife managers may prioritize habitat, ranchers may prioritize forage and beef production, and energy companies may prioritize energy development. Decisions on land use have implications for ecosystem services and climate-related outcomes. The primary goal for adopting a new conservation practice can result in different outcomes, so tradeoffs must be considered when assessing climate implications.

Adoption is influenced by landowners’ perceptions of ecological benefits and social factors like stewardship values. Ranchers who identify as stewards or have value-oriented leadership are more likely to adopt conservation practices than those with higher self-interest (Prokopy et al., 2019). Future willingness to restore is linked to prior restoration efforts and off-farm income, indicating that not all producers will adopt (Jellinek et al., 2013). Landowners can sometimes be more motivated by perceived threats such as invasive weeds and fire rather than ecological outcomes (Jellinek et al., 2013).

Adoption rates vary based on perceived risks, including unpredictable weather, invasive species, financial uncertainty, and long-term maintenance costs (Tankosic et al., 2023). Ranchers must evaluate the cost (upfront and ongoing), scale, environmental compliance, expertise access, and capital availability. Costs vary widely; for example, transitioning from degraded prairie to conservation prairie in the upper Midwest costs between $105-2,630 per acre, with a median of $750 per acre (Phillips-Mao et al., 2015).

Managing U.S. grazing lands is a complex challenge that requires balancing forage production and livestock utilization, economic sustainability, ecological resilience and climate mitigation. These lands span diverse Major Land Resource Areas (MLRAs; USDA, 2022a) and are subject to varied land tenure structures, including private, tribal, and public ownership. Differences in climate, soil, vegetation, and management histories make a one-size-fits-all approach ineffective. Instead, effective management strategies are often site-specific, informed by science-based conservation planning, and adaptable to changing environmental conditions.

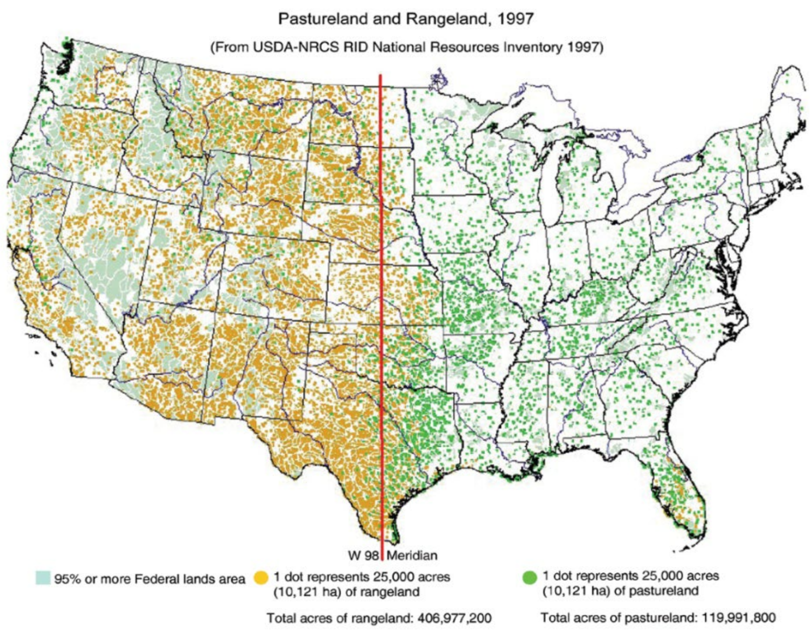

Figure 1. Distribution of private pastureland and rangeland in the 48 contiguous states of the U.S. based on USDA-NRCS data. [from Sanderson et al., 2009]

1.3 Climate Mitigation Processes and Potential from Grazing Lands

Although the total organic carbon (OC) storage and annual GHG budgets on U.S. rangelands and pastures are not well known, modeled estimates suggest they may be large, with OC at ~ 7.6 and 15 petagrams (1 Pg = 1 billion metric tons) respectively, stored primarily in soils (Pendall et al., 2018). Soil organic carbon (SOC) is an important ecological resource because it improves water holding capacity and retention of soil nutrients (including nitrogen, N), sustains belowground microbial and fungal systems, and supports productive and resilient plant communities (Johnston et al., 2009; Billings et al., 2021). Many countries, including the U.S., recognize that the protection of SOC is a matter of national security, needed for sustaining agricultural production while maintaining ecological services, while also guarding against land degradation and loss of biological diversity (Govers et al., 2013). Protection of SOC also important from a climate perspective, as transfer of carbon (and N) from soils to the atmosphere (in the form of GHGs, primarily carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) contributes to the acceleration of planetary warming (Hui et al., 2022).

Further benefits accrue from the sequestration of additional CO_2 into soils by plants through photosynthesis. Annual U.S. grassland CO_2 sequestration, based on model simulations, has been estimated at ~13 TgC yr^{-1}, although this estimate is also uncertain and incomplete because it does not account for CO_2 uptake from shrub/grass rangelands and pastures (Hayes et al., 2012). U.S. carbon sequestration budgets for shrub and shrub/grasslands are not well established, either, however global estimates indicate shrub sequestration rates may be even larger than grassland systems (Yu et al., 2018). Observations from CO_2 flux monitoring towers (**Figure 2**) scattered within the U.S. indicate that although median and average vegetation productivity (i.e., gross primary productivity, GPP) can be higher in grasslands relative to mixed grass and shrub systems, the net annual CO_2 sink (i.e., net ecosystem exchange, NEE) can be higher for grass/shrub systems relative to grasslands, possibly due to lower CO_2 losses from respiration (although more research is needed nationwide to confirm this, and the corresponding impacts of land management on NEE and the GPP and respiration components).

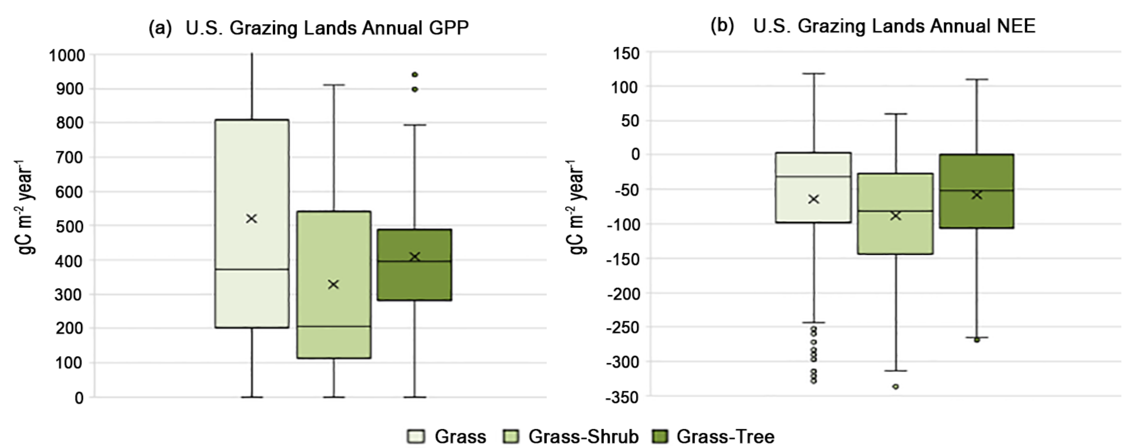

Figure 2. (a) Annual total vegetation productivity (GPP flux; gC m^{-2} year^{-1}) and (b) net ecosystem exchange (NEE; here negative values denote net land sink of CO_2; positive values indicate net source of CO_2 from land to the atmosphere) as observed for U.S. lands using tower-based eddy covariance flux monitoring systems (data from AmeriFlux (ameriflux.lbl.gov); Xia et al., 2025a),

according to grassland, grass/shrubland, and open canopy grassland/tree ecosystems. Note: boxplots indicate the median value and interquartile range; the "x" indicates the mean (average) value of the data.

Plant-sequestered carbon is translocated belowground into plant roots (**Figure 3**); carbon is then transferred into the soil system, or rhizosphere, through carbon-rich root exudates and root shedding (Kell et al., 2012). Grasslands allocate a large amount – approximately 60% – of their sequestered CO_2 into belowground root systems (Jackson et al., 2017). Shrub allocations of sequestered CO_2 to belowground systems are less studied, though the allocations may be similar to grasses (Carbone & Trumbore, 2007). Carbon in roots is then transferred to other belowground carbon pools, including microbial and fungal biomass, and soil organic matter (SOM, including particulate and mineral-associated) through root secretions and turnover. When considering plant growth and the distribution of plant carbon within aboveground and belowground biomass, it is important to keep in mind that the addition of nutrients (e.g., through fertilizer) can cause shifts in plant carbon allocation, resulting increased aboveground growth, but at the expense of reduced root growth.

The carbon cycle also includes respiration, where organic carbon sources are converted to CO_2. Respiration processes include plant growth and cellular maintenance respiration, and microbial and fungal respiration where SOC is consumed as an energy source. Plant respiration can vary among plant species and functional types, and across temperature and moisture gradients. For example, a study in the U.S. Southwest reported much higher respiration within annual plant communities relative to adjacent shrubs which tended to be more conservative with cellular growth and maintenance activity (Mauritz & Lipson, 2021). Microbial respiration generally increases with the increased availability of SOC to microbial communities (Bai & Cortrufo, 2022) and with increased soil moisture and temperature (Zhou et al., 2016; Hursch et al., 2017). Another component of the carbon cycle is the production and oxidation of CH_4. Methane is a GHG with a global warming potential (GWP) at least 84 times more potent than CO_2 over a 20-year period (Tian et al., 2017) and is produced when carbon is used as an energy source by specialized archaea (methanogens) under very wet, oxygen-limited conditions. Methane may be emitted at the soil surface (under water-saturated conditions) or consumed by methanotrophic bacteria in soils (converting CH_4 to CO_2) (Chan & Parkin et al., 2001; Xia et al., 2025). Methane from soil sources is considered insignificant and in general grasslands are a small net sink for CH_4, however livestock enteric and manure are large CH_4 sources (Saunois et al., 2020).

Lastly, N_2O can be produced under conditions where the soils are very wet (e.g., >80% volumetric water content) and there is excess N in the soil system (e.g., from overfertilization) (Wang et al., 2023). While N_2O and CH_4 are less abundant in the atmosphere, they are important for climate mitigation because of their high warming potentials, approximately 280 and 84 times that of CO_2 over a 20-year period, respectively (Tian et al., 2017)

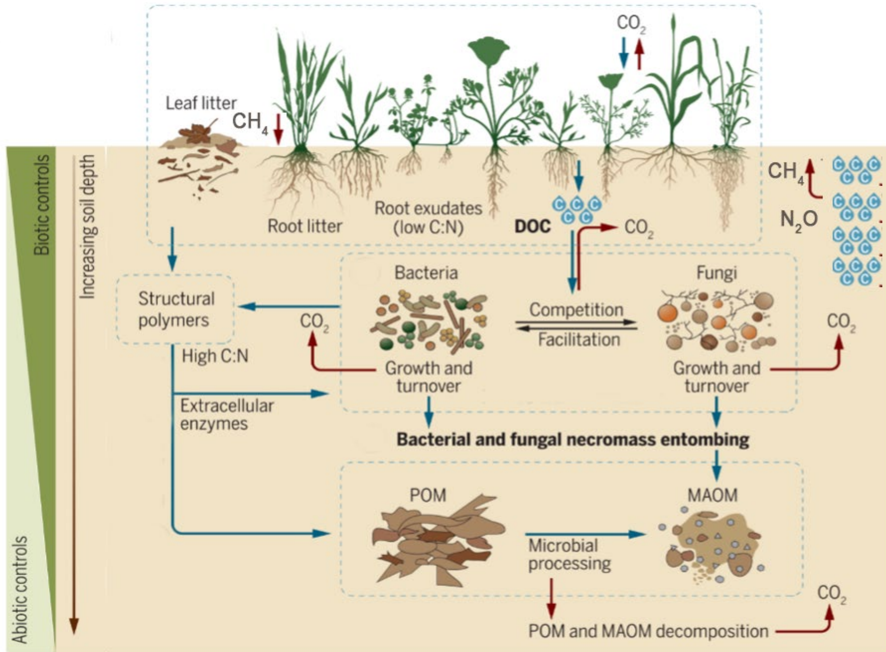

Figure 3. Carbon cycle processes in natural ecosystems. Plants sequester atmosphere CO_2 as bound (solid, non-gaseous) organic carbon (OC) through photosynthesis. Some OC is lost as CO_2 through plant (cell maintenance and growth) respiration. A portion of the plant-sequestered OC is transferred belowground to roots, and to litter. OC is transferred to other belowground pools, including microbes, fungi, soil particulate organic matter (POM) and mineral-associated organic matter (MAOM). OM can then be released as CO_2 through microbial decomposition, and as methane (CH_4) under wet, oxygen-limited conditions. (whereas CH_4 uptake can occur in dry ecosystems) Soil nitrogen can also be lost as nitrous oxide (N_2O) in very wet soils. [modified from Bai and Cotrufo, 2022]

HEALTHY RANGELANDS AND PASTURES, WITH CLIMATE CO-BENEFITS

Rangeland health has been defined as “the degree to which the integrity of the soil, vegetation, water and air, as well as the ecological processes of the rangeland ecosystem are balanced and sustained” (NRC, 1994; Pellant, 2005). Integrity is defined as “maintenance of the structure and functional attributes characteristics of a locale, including normal variability” (NRC, 1994; Pellant, 2005). In natural, “healthy”, ecosystems, system states (i.e., “patterns of ecosystem service bundles”, Bi et al., 2021) and biogeochemical cycles (i.e., energy, nutrient, water and carbon) have a high level of functioning (**Figure 4**), are typically more biodiverse (Folke et al., 2004; Hernandez-Blanco, 2022), and are thought to be more resilient to the impacts of short-term (pulse; e.g., drought and fire) and longer-term (press; e.g., decadal climate-driven changes in temperature and precipitation) disturbances. Degraded ecosystems, in contrast, are those where biodiversity (including presence of native plant communities) has been lost, plant cover has been lost (replaced by bare ground where it was not previously), and biogeochemical cycling has become sub-optimal or destabilized (e.g., limited use of sun energy by plants for photosynthesis; loss of carbon sequestration and storage, loss of soil nutrients, loss of water retention).

Because land degradation affects photosynthesis and GHG exchange, rehabilitation of degraded grazing lands will likely provide increased climate mitigation via increased CO_2 uptake and storage. However, in doing so it is important that management goals consider climate mitigation potential of a landscape (i.e., maximizing ecosystem photosynthesis (Smith et al., 2019) and reducing GHG loss), alongside other components of ecosystem health, such as biodiversity (Huston & Marland, 2003), to avoid tradeoffs.

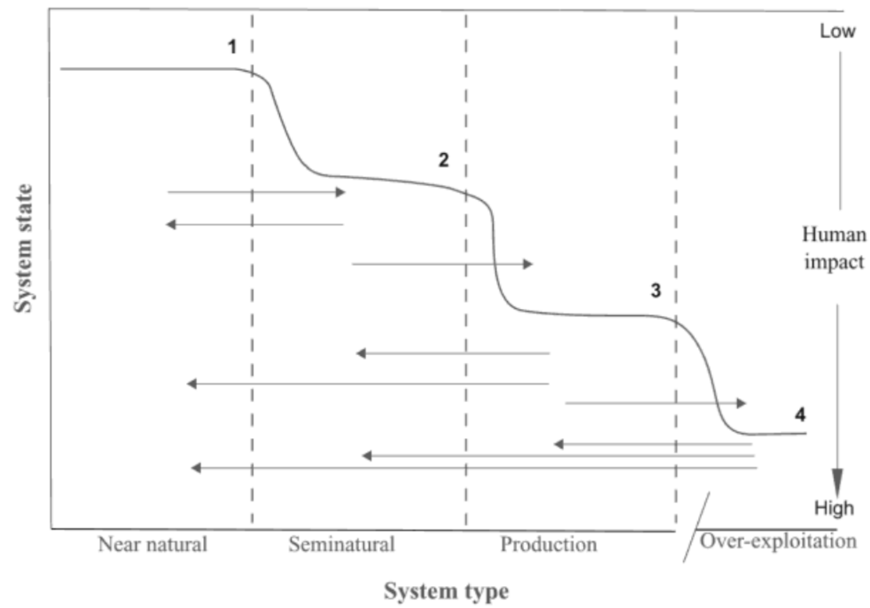

Figure 4. Conceptual diagram showing changes in System State and Cycles (y-axis) according to ecosystem conditions (x-axis), spanning from healthy, near-natural (left side of plot) to increasing degrees of degradation and declining health (towards right side of plot) due to direct (land use: e.g., grazing, cultivation) and indirect (e.g., climate change) human impacts. [modified from van Andel & Aronson, 2012]

Tracking the status and changes in an ecosystem's carbon cycle provides important information regarding the state of ecosystems (Janzen, 2005), while serving as a broader indicator of the health of agroecosystems (spanning more natural rangelands, to more actively managed pastures). Carbon is often considered as "The Great Integrator" of ecosystem function, as a robust carbon cycle corresponds with an active energy cycle, a healthy nutrient cycle (e.g., good retention and reuse of N and P; avoidance of excess nutrients and leakage) and water cycle (e.g., water is retained within the system to support healthy plant communities; water is used efficiently).

As mentioned earlier, the ecosystem carbon cycle includes two key components: (1) carbon transfer (i.e., fluxes); (2) carbon storage (i.e., stocks) (Heinemeyer et al., 2010). Primary inputs of carbon into grazing land systems are in the form of plant primary production (i.e., sequestration of CO_2 from the atmosphere through photosynthesis, resulting in the fixation in solid form as organic carbon) (**Figure 3**). This flux is often referred to (in scientific terms) as gross primary productivity (GPP) and is expressed in units of mass per area per time (e.g., g m^{-2} day^{-1}). Carbon is lost from the ecosystem through plant and microbial/fungal (heterotrophic) respiration (i.e., *Ra*, *Rh*; together termed *Reco*) as CO_2, as well as through harvest, grazing, pests, soil erosion, and leaching through the soil profile. Carbon storage (gC m^{-2}) is in the form of aboveground live biomass and litter, and belowground organic carbon in roots, microbial and fungal communities, and SOC.

From a carbon cycle and GHG management perspective, a healthy ecosystem is one in which productivity approaches the theoretical capacity of the ecosystem (Smith et al., 2019), where plant productivity and growth is controlled by available natural light, energy, and water conditions, and other factors including nutrients are at optimum (natural) levels to support plants. On the other hand, in degraded systems, productivity is below the natural capacity of a healthy, intact ecosystem (Cao et al., 2023). Global analyses indicate that the location-specific (theoretical) maximum photosynthetic capacity of an ecosystem is primarily controlled by climate (i.e., temperature, moisture availability), light availability, and green biomass, with soil fertility being an important, yet not primary, driver (Smith et al., 2019).

A healthy carbon cycle not only benefits nature, it directly benefits grazing lands producers by providing abundant forage for livestock. Further it stabilizes global climate by enabling CO_2 to become stored in ecosystems and reducing the amount of climate-warming GHG in the atmosphere. Importantly, while carbon storage can be an important indicator of ecosystem health, turnover of SOC is key to healthy ecosystem functioning as it provides basal resources for the soil food web, nutrient recycling for plant growth, and other key functions (Janzen, 2006). Similarly, having a balanced nutrient cycle, without excess N that may lead to N_2O emissions, is beneficial for ecosystems and climate.

ADDITIONAL CONSIDERATIONS FOR CARBON AND GHG ACCOUNTING

Understanding if grazing land conditions and management types yield co-benefits for ecosystems and climate ultimately requires documentation of long-term (multi-year) CO_2 exchange, plant biomass, and SOC stock status (see discussion in Xia et al., 2025 a,b). Additionally, other GHG fluxes, specifically CH_4 and N_2O, should be considered and accounted for. Enteric emissions of CH_4 are associated with the digestive processes of grazing animals and can vary from 53 to 446 g/day/animal depending on cattle breed, lifecycle, forage intake quantity and quality (see Xia et al., 2025b; Westberg et al., 2001). These emissions represent ~27% of total CH_4 emissions from anthropogenic activities in the U.S. and are higher than any other agricultural source of CH_4 (EPA, 2021). Whenever possible, annual enteric CH_4 emissions for a ranch operation should be calculated to understand the landscape and operational emissions.

Although CH_4 emissions from cattle manure may occur in rangelands or pastures, it is usually only under wet conditions where piles have formed. However, these emissions are minor relative to enteric and soil-originating sources (Xia et al., 2025b). In contrast, CH_4 uptake into soils may occur in drier rangeland and pasture systems (e.g., estimated at 2-4 kg CH_4 ha^{-1} yr^{-1}), with higher rates of uptake possibly occurring in well drained, more basic soils (Yu et al., 2017). A recent metanalysis of N_2O emissions for U.S. grasslands (see Xia et al., 2025b) found background emissions varied enormously, depending on the soil moisture conditions (i.e., wet), and the amount of N in the system. Although it is beneficial to consider soil N_2O emissions in assessments of grazing land contributions to climate mitigation, there is currently a lack of observation data and models allowing these calculations across U.S. rangelands and pastures. Additionally, the role of grazing intensity on N_2O emissions warrants investigation, as some studies have indicated higher N_2O emissions under higher intensity grazing and overgrazing relative to lower intensity grazing (see Table 2.8 in Xia et al., 2025b).

2. Grazing Lands Management Practices Assessments

This report assesses eight grazing lands practices that are widely used on private and public lands within the U.S. (**Appendix A Table 1**): Prescribed Grazing (528) (**Section 2.1**); Brush Management (314) (**Section 2.2**); Prescribed Burning (338) (**Section 2.3**); Pasture Plantings (512), Rangeland Plantings (550), Wetland Restoration (657), Riparian Restoration (390 and 391), and Herbaceous Weed Treatment (315) (collectively under Plantings, Enhancements, and Restorations, **Section 2.4**).

Within each section, we provide a definition and description of the practice, using NRCS Conservation Practice Standard definitions as a baseline. We then discuss the importance of the practice in the context of forage production, other ecosystem services, and potential climate impacts.

The potential climate impacts specific to each management type were assessed for each practice from peer-reviewed, published reports and research studies. The results of these literature reviews are described within the individual sections. Summary tables are also provided in the **Appendices**. Carbon and GHG-focused research specific to grazing lands is lacking for many of these practices. We therefore also considered ecological theory in the practice assessments. It is important to note that these assessments do not include any GHG impacts related to sourcing/producing, transporting, and/or applying seeds, chemicals or equipment/materials used in mowing/burning operations.

2.1 Prescribed Grazing (528)

2.1.1. WHAT IS PRESCRIBED GRAZING?

Across the U.S. and globally there is increasing interest in using prescribed grazing management (also referred to as “adaptive,” “holistic,” “improved,” “planned rotational,” “mob grazing,” or “regenerative” grazing) to improve the health and productivity of rangelands and pasture systems (Dillon & Machmuller, 2021; USDA 2022a; Xia et al., 2025a,b). In the context of climate mitigation, there is considerable speculation that this approach may significantly increase carbon sequestration and SOC storage, providing climate mitigation through “natural climate solutions” (e.g., Follett & Reed, 2010; Bai & Cotrufo, 2022).

According to the U.S. NRCS Conservation Practice Standard (CPS) #528, prescribed grazing involves “managing the harvest of vegetation with grazing and/or browsing animals with the intent to achieve ecological, economic, and management objectives.” Related, some producers and land managers move livestock across a landscape (aided by fencing and/or herding) in a manner that allow plants a long period of rest and recovery following short duration, intensive, grazing events (Morris et al., 2021) attempting to mimic the patterns of wildlife grazing (Kleppel & Frank, 2022). This approach is sometimes termed adaptive grazing management and reflects an active and engaged decision and response process. Note that although CPS #528 is currently entitled “Prescribed Grazing”, producers may adopt elements of adaptive management in implementing this practice.

The practice of prescribed or adaptive grazing might be generally summarized as “carefully managed grazing with intent”, targeting multiple objectives including livestock forage, ecosystem health and function, and climate mitigation (FAO, 2011). Achieving these goals involves the strategic control of grazing intensity, duration, and recovery periods to optimize plant regrowth, while improving soil structure, reducing erosion, enhancing water availability, and supporting bio-diverse plant and animal communities.

In contrast to adaptive management, “conventional” or “traditional” grazing allows livestock to continuously graze within a pasture or land unit for a full season or year-round, freely and without managed rotation through pastures or paddocks. Conventional grazing with high stocking rates is more likely to result in overgrazing of plants, leading to increased bare soil, declines in plant community health, declines plant CO_2 sequestration and SOC storage (Angerer et al., 2023). Continuous grazing (and, in certain cases, other grazing including rotational) with high stocking rates have substantially degraded many ecosystems across the U.S. and globally (e.g., described in Mohr, 2022), leading to severe SOC storage deficits (Sanderman et al., 2017). Addressing these deficits requires the replenishing of lost SOC through proactive, and adaptive, land management, and carefully designed and executed rangeland restoration efforts (Monaco et al., 2012). Below we synthesize the available published studies that explore whether prescribed grazing can help achieve those goals.

KEY GRAZING CONSIDERATIONS AND TERMINOLOGY

Important factors in grazing management that together play a critical role in shaping grazing outcomes include the kind and class of animal (e.g., browsers like goats, grazers such as cattle, age and reproductive status), timing, duration, frequency, selectivity and intensity (Bailey et al., 2019). *Timing* refers to when grazing occurs, while *duration* indicates how long animals graze on a particular unit of land. *Frequency* describes how often grazing takes place within a given period. *Selectivity* refers to the animals' tendency to focus grazing pressure on specific plant species or areas within the landscape. At present there is no consistent term or method for g*razing intensity* (Holechek & Galt, 2000; DeLonge & Basche, 2017). The lack of standard definition becomes especially problematic when comparing grazing outcomes across studies and geographies. Approaches that have been used to define grazing intensity include: (1) animal-based; (2) pasture-based; (3) forage allowance; (4) grazing pressure.

The animal-based approach is based on stocking rate (the number of animal units or pounds of liveweight per acre). An animal unit month (AUM) is generally considered as a bovine of 500kg requiring 350kg of dry plant material a month for feed. Pasture-based is measured based on the quantity of forage or plant height (Ogle et al., 2009). Forage allowance measures the amount of forage per unit of animal weight. Grazing pressure measures the relationship between animal liveweight and the amount of forage available.

A different measure of grazing intensity that has been proposed within the science community to support tracking and quantification of grazing intensity for specific land units and carbon purposes (Abdalla et al., 2018), based on estimated land unit NPP (mg dry vegetation matter ha^{-1} yr^{-1}, calculated using a combination of field measurements, climate data, remote sensing and models) and the estimated carrying capacity of the land (which is the number of AUMs that the land can support given local climate variability and other environmental factors). However, this approach has yet to be widely tested and adopted within the U.S. because it requires a large amount of technical capacity.

Grazing intensity is often used to describe how heavily an area is grazed. While the definition of grazing intensity is site specific, it is often tied to utilization rates. For example, according to U.S. Department of Agriculture guidelines (USDA, 2017), in New Mexico, light grazing intensity is where: up to 20% utilization of forage occurs at the end of the growing season (Nov. 15); the range visually appears undisturbed; there is no evidence of livestock trailing to forage; forage plants have abundant seed stalks and are topped or slightly used; young plants show little disturbance. Conservative grazing intensity is where: up to 40% of forage is used at the end of the growing season; there is no sign of livestock trailing to forage; forage plants have abundant seed stock (60-80% of stalks remain); up to ½ of forage plants have been grazed; most young plants are not damaged. Moderate grazing intensity is where: up to 50% of forage is used at the end of the growing season; most of the range unit shows evidence of grazing; there is little evidence of livestock trailing to forage; up to 2/3 of good forage plants show use; some young plants show damage; less than 10% of poor forage plants are utilized. Heavy grazing intensity is where: up to 60% of forage is used at the end of the growing season; all range shows use; there is evidence of trailing to forage; forage plants do not have seed stalks; all good forage plants are used; many young plants show damage; 10-50% of poor forage plants are utilized. Severe grazing intensity is where: greater than 60% utilization of forage is used by the end of the growing season; the entire rangeland has a clipped or mown appearance (stubble may be grazed to the soil surface); livestock trails to forage are common; there is no evidence of reproduction or current seed stalk; all herb species are completely utilized and shrubs are severely hedged; young plants are damaged or missing.

When grazing capacity and the appropriate amount of intensity required to achieve a healthy ecosystem are unknown or unclear, a general rule-of-thumb is to remove no more than 50% biomass during a grazing event (**Figure 5**) to avoid overstressing plant systems and hindering plant recovery. However, in reality, adaptive management requires thoughtful planning and monitoring to assess outcomes. This process should include grazing management plans specific to landscapes, management history, climate, and possible pulse and press stressors. The grazing management plan should guide the optimum grazing intensity used for specific land units, and the amount of rest and recovery needed to maintain forage and other ecosystem co-benefits. Ecosystem monitoring that includes metrics of plant community composition and biomass, bare ground, and soil health should be included within all grazing plans and is crucial for producers to quantitatively assess and track the success of (and deviations from) short-to-long term management goals.

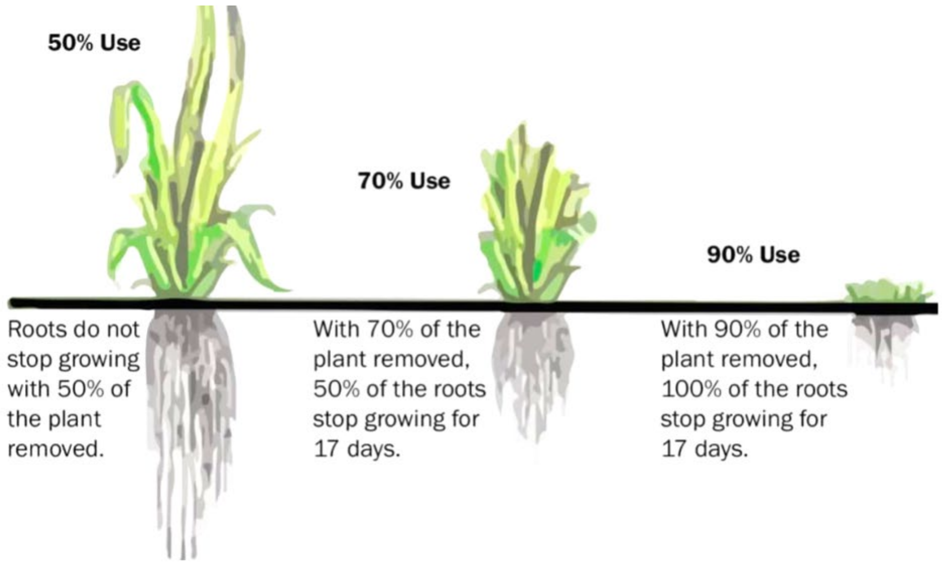

Figure 4. Conceptual illustration describing benefits of optimizing plant aboveground biomass removal through grazing, acknowledging that too much removal can hinder plant recovery and growth of larger, more robust root systems. [from Regeneration International].

2.1.2. METHODOLOGY

The search terms “grazing”, “adaptive grazing”, “rotational grazing”, “regenerative grazing”, “managed grazing”, “planned grazing”, “sustainable grazing management”, and “targeted grazing enhancements” in addition to “carbon sequestration", "gas flux”, “soil carbon” were applied to identify multiple publications through Agricola, the Web of Science and Google Scholar. The search focused on original research studies published in peer-reviewed literature occurring within North America, spanning grazing lands in the contiguous U.S. and southern Canada.

In total, nine relevant studies were identified for rangeland environments (**Appendix A Table 2**) and six were identified for managed pastures (**Appendix A Table 3**). These studies varied in geography, climate zones, vegetation and soil types, and the treatment examined. *Regarding management type:* only four of the rangeland studies included control (no grazing) plots; six of the rangeland studies compared continuous grazing (with various grazing intensities) practices with adaptive rotational grazing; four rangeland studies compared continuous grazing with various intensities (i.e., low, moderate, heavy). Of the pasture studies, five compared continuous grazing with adaptive rotational grazing; two compared grazing management with hayed; two compared intensity levels within continuous grazing approaches. *Regarding climate indicators measured*: only one rangeland study measured changes in CO_2 exchange, SOC and aboveground plant biomass; two studies measured CO_2 exchange and SOC; almost all (8/9) studies measured SOC. The pasture studies only reported measured SOC and did not measure GHG exchange.

In addition to information from the identified grazing studies, the findings from a recent metanalysis and systematic review of global grazing synthesis studies (Sanderman et al., 2025) are also presented. Sanderman et al. (2025) examined the original data from 69 studies globally that compared continuous grazing to rotational grazing management. They found that most studies lacked crucial data needed for robust statistical comparisons. Only nine of 69 studies reported information about treatment type and length, external factors (confounding variables) including climate, topography, soil type that would also influence SOC stock differences; these are necessary to comparatively assess outcomes.

Report authors conducted an expert interview with Dr. Jon Sanderman from the Woodwell Climate Research Center in winter 2025.

2.1.3. Results and Discussion

Important findings from the literature review are summarized below for rangelands and for pasture systems, followed by a discussion of considerations for how grazing management approaches might benefit ecosystems and climate.

Although the term “overgrazing” can be defined in various ways, here we loosely refer to overgrazing as where grazing events remove live plant biomass to the extent that plants are unable to rapidly recover, increasing the risk of plant mortality.

KEY FINDINGS FOR GRAZING MANAGEMENT IN U.S. RANGELANDS

Key Finding #1. *The literature clearly demonstrates the long-lasting detrimental impacts on plant productivity, carbon sequestration, SOC and CO_2 exchange that results from overgrazing in rangelands.*

The Morgan et al. (2016) study (**Appendix A Table 2**) at the USDA-Agriculture Research Service (ARS) research station in Colorado identified that continuous, heavy-intensity grazing resulted in a substantial net annual loss of CO_2 (NEE of 192 gCO_2 m^{-2} yr^{-1}; ~0.86 U.S. tons CO_2 acre^{-1} yr^{-1}) under drier precipitation conditions, whereas ungrazed and continuous moderate-grazing treatments remained net annual sinks of CO_2 (-142 and -186 gCO_2 m^{-2} yr^{-1}; -0.63 and 0.83 tons CO_2 acre^{-1} yr^{-1}). In years having more normal precipitation, the same site showed NEE as being CO_2 flux neutral under the heavy-grazing treatment, compared to a strong CO_2 flux sink under moderate grazing intensity (-267 gCO_2 m^{-2} yr^{-1}; -1.19 tons CO_2 acre^{-1} yr^{-1}). Similar response was documented in the Ingram et al. (2008) Wyoming study where SOC was lost in all treatments (ungrazed, continuous low-intensity, continuous high-intensity) during periods of severe drought, however the continuous high-intensity site lost much more SOC and had severely reduced plant productivity relative to the other sites. The wealth of data provided in Ren et al. (2024) also demonstrates how overgrazing can increase soil erosion, negatively impact plants, and result in losses of soil carbon across depths.

Key Finding #2 *There is evidence that some (low to moderate intensity) grazing may yield more plant productivity and SOC benefits relative to no grazing in grassland systems across arid to humid ecosystems.*

The Ingram et al. (2008), Hewins et al. (2018), Gomez-Casanovas (2018) and Ren et al. (2024) studies (**Appendix A Table 2**) indicate that light to moderate intensity livestock grazing in rangelands might yield positive ecosystem and climate benefits relative to ungrazed systems. For example, Ingram et al. (2008) reported that less vegetation productivity and SOC stocks were lost within a native mixed grass system in Wyoming having continuous, light intensity grazing during a period of severe drought, relative to a nearby ungrazed (control) system. This was possibly due to the light grazing pressure stimulating grass species investment in fuller and deeper root systems, whereas the ungrazed system did not benefit from stimulus of root development. Hewins et al. (2018) examined grazing in a mixed grass prairie within Alberta, Canada and reported higher SOC concentrations in areas having continuous light to moderate intensity grazing, relative to grazing exclosures (control sites). The Gomez-Casanovas (2018) study within a semi-native and invasive bahiagrass (*Paspalum notatum*) system in Florida also reported larger SOC stocks under continuous (light to moderate intensity) grazing relative to ungrazed land. In addition, the study also found that grazed sites had a higher annual NEE sink relative to the ungrazed.

Key Finding #3. *The carbon sequestration and SOC storage benefits of adaptive, rotational grazing vs continuous (low and moderate intensity) grazing in rangelands remains inconclusive, particularly for more moisture-limited environments.*

Multiple research studies within the U.S. have reported a lack of significantly higher SOC in adaptive (rotational) grazing management systems relative to continuous grazing having low-to-moderate intensities (e.g., see Hillenbrand et al., 2019; Schantz et al., 2024; Taylor et al., 2025). However, two studies did report possible positive impacts from adaptive grazing. This included a study in coastal California (Stanley et al., 2025; annual precipitation > 200 cm; 78 inches) that reported significantly higher SOC stocks under adaptive (rotational) grazing (with unknown and likely varying intensities) relative to neighboring sites that were continuously grazed (**Appendix A Table 2**), except for sites having very clayey soils. Similarly, a study across Alabama, Kentucky, Mississippi and Tennessee (Apfelbaum et al., 2022; annual precipitation > 100 cm; >~39 inches) reported higher SOC in some (not all) adaptive (low intensity) rotational grazing sites relative to paired continuous (low intensity) sites. Why some sites did not show a significant response was unclear in the study, possibly influenced by local differences in soil and plant characteristics, as were all the retrospective observational aspects of the study.

The overall lack of evidence at present supporting carbon sequestration and SOC benefits from adaptive management across many ecosystem types and climate zones is reflected elsewhere. The Sanderman et al. (2025) analysis found no statistical difference in SOC stocks between sites using continuous grazing relative to adaptive (rotational) management. In addition, Sanderman et al. (In prep) found the positive (and reportedly significant) findings in the often-cited Byrnes et al. (2018) global metanalysis to be erroneous, driven by questionable data and issues with the statistical analysis. Other global analyses (e.g., DeLonge & Basche, 2017; Rouquette et al., 2023) have also reported problems in assessing existing data due to issues with many grazing studies not reporting necessary information detailing site plant and soil conditions, important details about past and current grazing management (e.g., timing and duration of grazing, intensity, etc.). Collectively, they emphasize an overall need for well-designed experimental and observational research exploring the effects of changes to grazing management on SOC (and other indicators, including aboveground and belowground biomass, and NEE flux) while also tracking variability in environmental conditions that drive carbon cycle and GHG response (including air and soil temperature, soil moisture, soil nutrients). Because measurable SOC response may be relatively slow, and climate variability (e.g., warmer vs cooler; wetter vs drier years) is expected, research experiments focused on grazing impacts must operate consistently across multiple years (10+ years) to best identify ecosystem impacts. Whenever possible, incorporation of continuous GHG monitoring systems (e.g., eddy covariance towers) is also recommended to quantify GHG exchange (including NEE) at treatment vs control sites.

In addition, we encourage those implementing grazing management to apply a data driven approach (Derner et al., 2022), carefully and consistently monitoring annual changes in plant cover and properties and conducting repeat soil sampling to identify changes in SOC stocks (~0-30 cm depths) at 5-year intervals relative to control sites (Xia et al., 2025b). An annual reevaluation of grazing intensity based on stubble heights and the avoidance of heavy grazing (high intensity) continuous grazing of rangelands is also recommended (USDA, 2017).

KEY FINDINGS FOR GRAZING MANAGEMENT IN U.S. PASTURES

Key Finding #1. *There is some evidence suggesting that adaptive management in pastures can increase SOC and/or plant biomass relative to continuous grazing, however the results overall are inconclusive.*

Two studies (**Appendix A Table 3**) reported significantly higher SOC stocks in pastures under adaptive grazing management relative to those continuously grazed. This included the Mehre et al. (2024) study in southern Ontario, Canada, and the Mosier et al. (2024) study in Mississippi. Both studies included pastures that had been previously cultivated as grain crops and were replanted to pasture lands at some point in recent history. Because of this, caution should be used when interpreting the total SOC gains due to grazing management, as the variable land recovery trajectories influenced by cultivation and replanting could confound the observed outcomes (e.g., as discussed in Xia et al., 2025b). Although a study in Florida (Dubeux et al., 2006) did not observe significant differences in SOC between pastures under continuous low-intensity vs moderate-intensity vs high-intensity grazing, and adaptive rotational grazing (high-intensity stocking rates) they reported higher plant biomass accumulation in the adaptive treatment relative to continuous. However, a study in Arkansas (Amorium et al., 2020) found no difference in SOC stocks between continuously grazed and adaptive (rotationally) grazed pastures, although a manure amendment added to the pastures likely confounded the results.

Key Finding #2. *There is some (although very limited) evidence that high intensity grazing can be incorporated in moist, managed pastures without harm to plants and SOC, however, more studies are needed to confirm this observation.*

A study in North Carolina (Wang et al., 2014) comparing low intensity versus high intensity continuous grazing in a moist pasture having deep-rooted plant species (ryegrass, *Lolium perenne*; sudangrass, sorghum x drummondii; sorghum; bermudagrass, *Cynodon dactylon*) reported significantly higher SOC stock under the high intensity grazing treatment relative to the low intensity treatment four years after the study initiation. It is possible, in this case, the perennial plants in the pastures had substantial enough root systems and minimal nutrient and water limitations, allowing them to sustain the added pressure of high intensity continuous grazing.

Key Finding #3. There is some (although very limited) evidence that the grazing of pastures may yield higher SOC benefits relative to ungrazed (hayed) pastures. However, further investigations (and more data) are needed.

A study in Tennessee (Tilhou et al., 2021) compared planted pastures under low intensity grazing versus no grazing (with annual haying) and reported higher SOC concentrations within the grazed pasture (however, SOC stocks were not reported).

IMPORTANT GRAZING MANAGEMENT CONSIDERATIONS

Climate impacts of adaptive grazing remain unclear

The results of research studies within the U.S. remain inconclusive regarding the ability of conversion from conventional grazing to adaptive management in rangelands to yield significant ecosystem benefits in terms of improved CO_2 sequestration and SOC storage. At present it is unknown if, and under what conditions, the conversion from one grazing management type to another (including to adaptive grazing) might yield positive climate mitigation impacts. Many of the existing research on grazing impacts in the U.S. and elsewhere remain confounded by lack of carefully selected control sites, consistent grazing treatments and monitoring taking place over multiple years (typically up to 10 years is recommended to observe clear signals in SOC and GHG response, given inherent between-year climate variability), detail regarding herd site and grazing density, and the inclusion of multiple ecological indicators (e.g., SOC stock at 0-1 meter depth; soil temperature and moisture; microbial community characteristics; plant community characteristics and root depth; GHG flux) with the study design.

Carefully planned and monitored (conventional and rotational) grazing can yield multiple ecological benefits

As mentioned in Section 2.2.2.2, there is some evidence from research studies that livestock grazing in rangelands and pastures may yield positive benefits relative to no grazing (e.g., Ingram et al., 2008; Hewins et al., 2018; Gomez-Casanovas 2018). Especially important is that grazed systems might be more resilient to SOC losses and reductions in CO_2 sequestration during pulse disturbances (e.g., droughts) relative to ungrazed. These observations appear to align with ecological theory describing how grazing pressure (applied in a manner that does not overstress plants) can stimulate plant investments in more robust root systems (NRCS, 2017). In addition to promoting increased carbon storage belowground, co-benefits of stronger root systems include improved access to water and nutrients stored deeper in soils, likely leading to more production of plant forage, and an increased ability to recover following future grazing events and other disturbances such as drought and fire (e.g., Manning et al., 2017). However, care must be taken to avoid unwanted changes in species composition (through grazing pressure, which may result in selection of plant types inherently having shallower root systems (e.g., see Bonin et al., 2013).

Another important benefit of grazing management that is of interest across the U.S. are the co-benefits of reducing standing dead biomass, thereby greatly reducing the risk and severity of fire. For example, a study of grazing across rangelands in California found that the substantial removal of non-woody plant material through grazing led to more manageable wildfires and also lessened fire hazards in many areas (Ratcliff et al., 2022).

Although this report focuses on rangeland and pastures, some studies point to the benefits of livestock grazing in rotationally cropped lands, in efforts to biologically process residual biomass, and to graze weeds (possibly reducing or avoiding the need for herbicide applications) (Teague & Kreuter, 2020). A helpful commentary on integrated crop-livestock systems within the U.S. can be found in Sulc & Franzluebbers (2014).

Finally, regardless of the grazing management goals, *there is a pressing need for more producers and land managers to commit to consistent annual monitoring of multiple ecological indicators within rangelands and pastures* (**Figure 6**; DeLonge & Basche, 2017; Sayre, 2011; McCord & Pilliod, 2022) to determine if the ecological responses taking place are consistent with management intentions. Carefully prescribed grazing may also be helpful in managing invasive plants within rangeland and pasture systems (DiTomaso et al., 2010).

A need to protect and restore grazing lands

From a climate mitigation and ecological perspective, protection of large carbon stocks that exist in rangelands and pastures should be prioritized. This requires identifying and protecting healthy landscapes that hold larger amounts of carbon and provide net CO_2 uptake, avoiding land conversion to non-perennial systems or development, avoiding degradation through overgrazing, avoiding severe fire, etc. Protecting intact rangeland (and biologically diverse pastureland) is also important as it yields numerous ecological and societal co-benefits including habitat biodiversity, water retention/storage, forage and habitat for wildlife (Follett & Reed, 2010; DeLonge & Basche, 2017; Godde et al., 2022).

Overgrazing should be avoided. This can be achieved through thoughtful, detailed and adaptive prescribed grazing plans, based on local site-level information, assistance from technical service providers such as NRCS field offices, university extension agents, and local grazing networks, and seasonal-to-annual monitoring of plant and soil conditions (Sollenberger et al., 2012; Brown & Herrick, 2016) through pasture and rangeland scoring or other approaches (e.g. BLM, 2020; NRCS, 2020). In addition to improving climate mitigation potential of the landscape, avoiding overgrazing yields numerous co-benefits including avoiding increases in bare ground, undesired changes in plant species composition and quality, and detrimental reductions in green aboveground biomass.

One potential component of an adaptive management and planning process is the incorporation of Forage Balance Assessments (FBAs; NRCS, 2022). FBAs are foundational tools used by NRCS conservation planners and producers to evaluate whether a grazing operation has enough forage available to sustain livestock throughout the year, which is useful for avoiding overgrazing. It compares total forage availability with livestock forage demand, helping ranchers and land managers make informed decisions about stocking rates, pasture rotation, and supplemental feeding. The FBA is the first step in preventing overgrazing and ensuring healthy, productive rangelands. Without this assessment, grazing management decisions are based on assumptions rather than science, increasing the risk of overgrazing and land degradation. By integrating forage balance calculations into conservation planning, the NRCS helps ranchers sustainably manage their land, livestock, and natural resources for future generations.

If overgrazing does occur within rangeland or pastures (see Mount, 2009; USDA, 2022a for descriptions of how to identify overgrazing) the impacted lands (and any adjacent intact land) should be immediately removed from livestock grazing for a long enough time period to allow for full recovery of plant cover and reproduction (natural reseeding). If plants are unable to reestablish naturally in the absence of grazing, then rehabilitation efforts may be required, including the active reseeding of native grass mixtures or pasture mixtures (see **Section 2.4**).

Degraded landscapes that do not show adequate signs of recovery should be identified and prioritized for rehabilitation efforts (weed treatment and reseeding; removing grazing for an appropriate period of time -- potentially years within arid systems). Example case studies on rangeland remediation efforts within the U.S. include: 1) Herrick et al. (2006), which focuses on the Chihuahuan Desert Ecosystem; 2) Walsh & Rose (2022) which reviews rangeland restoration efforts at highly disturbed oil and gas sites (providing an extreme example of restoration in heavily degraded rangelands) across arid and semi-arid landscapes in North America; and 3) Smith (2010) which overviews approaches to prairie restoration in the Midwest.

Walsh & Rose (2022) emphasize the need to avoid grazing, or to only apply light intensity grazing with long rest, until perennial plant communities have been restored. In many cases, it may be appropriate to include a long-term grazing plan as part of the land restoration planning process (Papanastasis, 2009). Grazing should also be avoided in areas having intact biological soil crusts (also referred to as biocrust, which are symbiotic communities of moss, lichen and algae present in arid and semi-arid systems, which are important for CO_2 uptake) and where biocrust restoration is occurring (Zaady et al., 2016). Similarly, grazing should be limited or avoided in other ecologically sensitive zones including wetlands and riparian zones, particularly those that have been severely degraded and those under active restoration management.

GAPS IN GRAZING KNOWLEDGE, COMMUNITY DATA NEEDS

Across the U.S. there remains a severe lack of data detailing possible impacts to ecosystem services, including climate mitigation, that might result from various livestock grazing approaches taking place under a range of ecosystem conditions (i.e., climate gradients, plant communities, soil texture, moisture, nutrient availability). This information is needed not only for specific grazing types (e.g., conventional under light, moderate, heavy grazing intensities; rotational/adaptive management under various grazing intensities) but also documenting changes in ecosystem conditions and ecological services following a conversion from one land management type to another.

Addressing the lack-of-data requires the careful establishment of controlled and well-documented research studies, and substantial increases in research funding available to support multi-year (5- to 10-year duration) studies across the U.S. region. Research approaches that are more likely to yield high-quality data necessary for rigorous statistical analyses and the testing and advancement of process-based ecological state and response models representative of grazing systems must include well-replicated and detailed experiments featuring control vs (grazing management) treatment sites. High quality studies must also take care to record grazing (animal units; specific locations/timing of grazing events; forage removal), ecosystem (plant species, plant cover and bare ground, soil texture, soil organic matter), and environmental (air temperature, precipitation, soil moisture, soil temperature, pH, etc.) characteristics across time and space (DeLonge & Basche, 2017). For carbon and GHG focused studies, additional data about seasonal and annual aboveground carbon biomass (and when possible, root depth/biomass), and annual GPP and NEE flux should be obtained. Regarding collection of SOC information, more rigorous studies should collect soil cores up to at least 1 m (3.28 feet) depth, allowing multi-depth tracking of SOC concentrations and bulk density, which are needed to estimate total SOC stocks (e.g., see Xia et al., 2025 a,b).

Outside of rigorous research settings, there is a need to support ranchers and land managers in developing and implementing consistent annual ecological monitoring to identify and track changes in land health. Necessary support includes working with regional producers to assess and improve existing monitoring frameworks and providing financial incentives to producers for annual monitoring and reporting. Assistance in paying annual subscriptions to online commercial planning and analysis software such as *Pasture Map* (grassrootscarbon.com/pasture-map) should be considered, along with bonus incentives provided if ranchers demonstrate how changes in management, assisted by planning tools and (when appropriate) plant and soil sampling, has improved land health and SOC sequestration. Additionally, educational programs and knowledge-sharing workshops where ranchers and land managers can learn about how bringing together information from multiple data sources (e.g., field monitoring, GPS-mapping, remote sensing) within geospatial information systems (GIS) can provide invaluable ecological and geospatial knowledge needed to track land health wall-to-wall across grazing units.

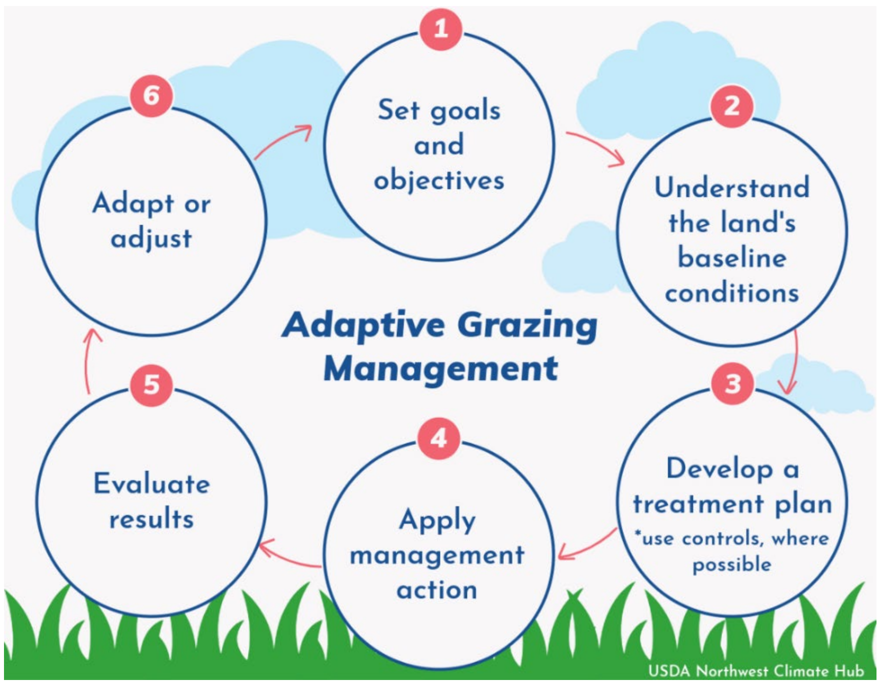

Figure 5. Conceptual illustration describing the adaptive grazing process. [from the USDA Northwest Climate Hub].

2.2 Brush Management (314)

2.2.1. SHRUB AND BRUSH MANAGEMENT IN U.S. RANGELANDS

Shrubs, also called brush, can occur in most landscapes; however, they may become dominant in habitats having nutrient-poor soils and/or disturbance (e.g., from overgrazing or fire) (McArthur & Kitchen, 2007). The spread of woody plants into grasslands, transforming rangelands into shrub-dominated landscapes, has been extensively documented over the past decade (Maestre et al., 2009; Van Auken, 2009) both globally and across the U.S. (Barger et al., 2011), driven by warming temperatures and changes in precipitation, in addition to land use and management (Wu et al., 2024).

The process of shrub expansion, known as encroachment, refers to the increase in the density, coverage, and biomass of native and/or non-native woody or shrubby vegetation. This trend is especially widespread across arid and semi-arid regions. For instance, in the Western U.S., the significant expansion of brush such as mesquite and creosote bush has contributed to the conversion of once-grass dominated rangelands into dense shrublands (Buffington and Herbel, 1965). This shift in vegetation has accelerated across the U.S. since the early 20th century (Archer, 2010), largely influenced by factors such as livestock grazing and reductions in human-managed fire occurrence and frequency (D'Odorico et al., 2011). At present, it is estimated that brush encroachment in non-forested lands within the Western U.S. effects is impacting > 330 million ha, or 25% of the total land area (Knapp et al., 2008; Morford et al., 2022**; Figure 7**). In consequence, a large amount of herbaceous biomass needed for livestock and wildlife forage has been lost since 1990, with estimated economic costs of $4.1 – 5.6 billion (Morford et al., 2022).

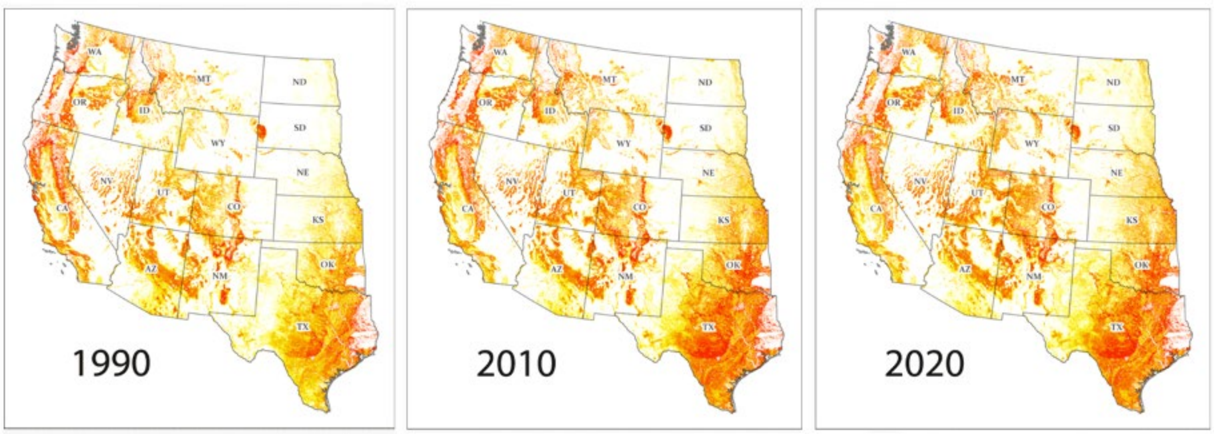

Figure 7. Maps of the western U.S. showing the decadal expansion of woody biomass (i.e., trees and shrubs) from 1990–2020, derived from Landsat satellite imagery. [www.landscapeexplorer.org]

From an ecosystem carbon and climate perspective, shrubs can provide net benefits from CO_2 sequestration and storage of carbon primarily aboveground in woody biomass and belowground in roots and soils, as has been documented in arid grazing regions (e.g., Hunt et al., 2004; Svejcar et al., 2008; Daryanto et al., 2013).

Described in USDA Conservation Practice Standard 314 (314-CPS-1), this management type focuses on the removal of woody (non-herbaceous or succulent) plants, including those that are invasive and noxious. Currently the NRCS considers brush management as a potentially climate-smart practice when implemented within U.S. regions having a mean annual precipitation (MAP) less than 340 mm (13.4 inches). Brush management may include chemical, mechanical, and/or biological (e.g., grazing) treatments. Per NRCS guidelines, woody materials must be left onsite to decompose naturally to mitigate carbon loss. Burning of residues or the off-site hauling of woody residues is not allowed with this practice standard.

The primary motivation for removing woody vegetation is to enhance deep rooted native perennial grass and forb communities that may have been encroached and outcompeted by woody plants through overcanopy shading and limitations in upper-layer soil water availability (e.g., Pierce et al., 2019). The re-establishment of native perennial plant communities is generally expected to increase forage for livestock, while possibly increasing overall plant uptake of CO_2 and SOC stocks (NRCS 314-CPS-1). It is also postulated that the removal of woody biomass and restoration of non-woody perennial ecosystems would reduce ecosystem susceptibility to severe wildfire that could lead to large decreases in vegetation and aboveground and belowground carbon stocks (Twidwell et al., 2013; Li et al., 2022).

2.2.2. METHODOLOGY

We used published articles indexed in Google Scholar from anytime in the past to 2024 to collect data on the ecosystem carbon and greenhouse gas impacts of brush management across U.S. grazing lands. The search for published articles targeted experimental studies of brush management methodologies (e.g., mechanical or chemical), or correlative studies of the effects of brush encroachment (e.g., naturally encroached and uninvaded landscapes), that took place on a variety of grazing land types (e.g., rangeland, pastureland, shrubland, grassland, and savanna), and included at least one measured pool or flux of ecosystem carbon (e.g., soil total carbon, SOC, belowground plant carbon, aboveground plant carbon, soil respiration, CH_4 flux, N_2O flux, aboveground GPP or NPP, or NEE). Livestock grazing, prescribed burning, and plantings were not included as a method of brush management given their statuses as standalone NRCS Practice Standards but were noted as an added benefit or detriment to ecosystem carbon where relevant. Although NRCS Climate-Smart applications of CPS-314 are specific for U.S. regions having MAP $\leq$ 314 mm (12.36 inches), this analysis was extended to include field studies having a MAP > 314 mm to compare impacts of woody shrubs on the ecosystem carbon cycle and climate mitigation across multiple climate zones. Data collection was restricted to the results of field studies (i.e., not modeling or simulation studies) that either experimentally applied a brush management treatment with a paired untreated control, or that examined encroached environments relative to nearby brush interspace areas. Data collection was also restricted to the results of field studies that took place in open canopy grazing lands. Grazed closed-canopy forests (i.e., those described as having a dense, continuous canopy, shading out understory growth) were, therefore, not included in this review as these represent systems with a different carbon cycle structure. Hence, closed forest ecosystems are outside of the scope of this report.

In total we identified 23 field studies (**Appendix A Table 4**) within the U.S. that examine impacts to carbon storage and/or GHG exchange associated with shrub encroachment in grasslands, and treatment of encroached shrubs. The studies represent gradients of precipitation. However, only two studies reported MAP of $\leq$ 340 mm (~ < 13.38 inches) and three sites had a MAP of 345 mm. The remaining sites spanned MAPs >345 and < 900 mm (13.58 to ~35 inches).

2.2.3. RESULTS AND DISCUSSION

KEY FINDINGS FOR SHRUB MANAGEMENT IN U.S. RANGELANDS

Key Finding #1. *There is evidence in the literature that encroachment by woody species into U.S. grasslands, spanning arid to humid climate zones, may provide carbon benefits that should be considered in the management of grazing lands.*

A majority of the studies examined that included analysis of SOC (i.e., Connin et al., 1997; Gill & Burke, 1999; Geesing et al., 2000; Hibbard et al., 2001; Jackson et al., 2002; Hibbard et al., 2003; McCulley et al., 2004; Liao et al., 2006; Wheeler et al., 2007; McClaran et al., 2008; Miwa & Reuter, 2010; Throop et al., 2012; DeMarco et al., 2016; Zhou et al., 2017; Abdallah et al., 2020; Connell et al., 2020) reported higher SOC concentrations or stocks associated with the locations of woody species, relative to perennial grasses (including native prairie).

Of the two studies having a MAP $\leq$ 340 mm, one (Miwa & Reuter, 2010) reported higher SOC concentrations associated with the past locations of shrubs (Western juniper; *Juniperus occidentalis*) in Eastern Oregon even 8 to 15 years following mechanical cutting, relative to nearby grass interspaces. In contrast, the other arid study (Derner et al., 2014) which examined the impacts of chemical treatment and mowing on Wyoming Big Sagebrush (*Artemsia tridentata wyomingensis*) reported no difference in SOC stocks between treatment and control sites under sandy loam soil conditions, and higher SOC stock following treatment in a loam soil (attributed to increases in grass cover and productivity under less water limitations). Interestingly, another study (Throop et al., 2012) in Arizona (MAP 345 mm; 13.58 inches) reported larger SOC stocks in areas encroached by velvet mesquite (*Prosopis velutina*) relative to areas dominated by grass under drier soil conditions, whereas SOC was similar in wetter soils. Further investigation is needed to better understand the impact on SOC stocks as influenced by plant species (woody vs grass) and soil moisture.

Only five of the studies examined reported aboveground biomass, with mixed results. Derner et al. (2014) reported a loss of annual biomass (and carbon stock) following treatment (more so from mowing relative to chemical application). Abdallah et al. (2020) reported much *smaller* aboveground carbon stocks 13 years post-treatment (chainsaw removal of western juniper) relative to the control site, even though grass cover had increased. Coultrap et al. (2008) reported *no change* in overall aboveground biomass in California following mechanical removal of Western juniper and sagebrush, as decreases in shrub cover were offset by increased herbaceous and cheatgrass biomass. In contrast, the Vivoni et al. (2022) study in Arizona reported increases in velvet mesquite cover and decreased grass cover following chemical treatment, with *no overall change*. Lastly, Heitschmidt et al. (1986) found *no difference* in herbaceous biomass between treated and untreated sites following the chemical application in areas encroached by honey mesquite (*Prosopis glandulosa*).

Only four studies examined CO_2 fluxes. The Derner et al. (2014) study (Wyoming) reported loss of annual ecosystem CO_2 (NPP) uptake following shrub (Big Sagebrush) treatment, with larger losses associated with the chemical treatment relative to the mowing treatment due to more extensive shrub death and slower regrowth. Hunt et al. (2004) (also Wyoming) reported much larger NEE (CO_2) sink in sagebrush encroached lands relative to grassland communities (which varied between small CO_2 sink to net CO_2 source in drought years). The Vivoni et al. (2022) study in Arizona reported higher CO_2 (GPP) uptake in velvet mesquite two years following chemical spray, relative to the control site, attributed to shrub recovery and increased water use efficiency by new shrub growth. This same study reported higher NEE sink in the shrub ecosystem, relative to grassland, because deep shrub roots were able to access water at depth during dry periods. Heitschmidt et al. (1986) reported no difference in NPP from herbaceous biomass between shrub treatment and control sites (however, total ecosystem NPP including shrubs is unknown).

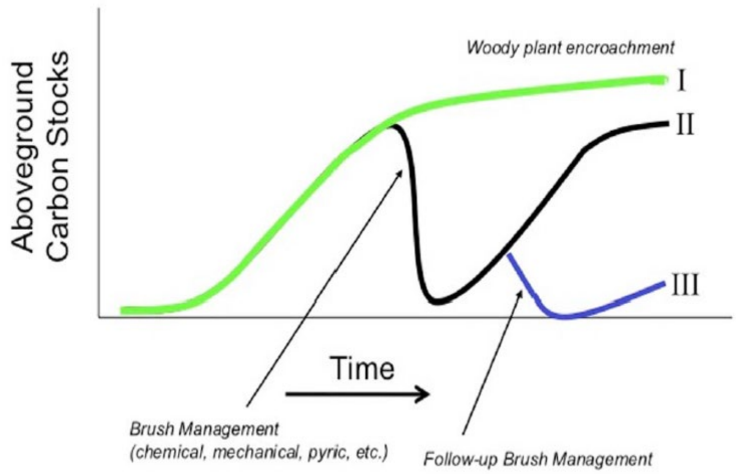

Figure 6. Diagram illustrating response in aboveground carbon stocks (i.e., plant biomass) over time following brush removal. [from Barger et al., 2011]

Key Finding #2. *From a carbon perspective, caution is needed whenever grazing management goals include the removal of woody biomass with the intention of increasing grass and herbaceous cover for livestock forage.*

Some studies report increases in bare ground and reduced grass and/or herbaceous biomass for multiple years following herbicide and mechanical treatments that result in shrub mortality (e.g., Coultrap et al., 2008; Derner et al., 2014; Abdallah et al., 2020; Vivoni et al., 2022; also see the Watson et al., 2019 California study). At least one field study (Coultrap et al., 2008) reported increases in cheatgrass (*Bromus tectorum*) cover following shrub treatment. This finding was echoed in a regional assessment of cheatgrass expansion coinciding with the conversion of shrublands into grasslands (Bradley et al., 2006).

If shrub management must occur within grazing lands, it is recommended that shrub thinning using mechanical and/or biological (targeted grazing; e.g., Fletcher, 2024) methods be applied (along with chemical spot treatment targeting individual shrubs, if needed) to avoid widespread shrub mortality. Whenever possible, reseeding of desired native grass and herbaceous species in the shrub interspaces is recommended to assist with biomass recovery (aiming for recovery to original or > aboveground stocks relative to pre-treatment; **Figure 8**).

IMPORTANT SHRUB MANAGEMENT CONSIDERATIONS

This report finds that multiple U.S. studies indicate positive impacts of shrub encroachment on SOC stock (in addition to aboveground biomass and CO_2 uptake) regardless of MAP. Many studies also report detrimental impacts on SOC stock and CO_2 uptake following the removal of shrubs in efforts to increase grass and herbaceous forage (see **Appendix A Table 4**; also, McClaran, Moore-Kucera, Martens, Haren, & Marsh, 2008). Similarly, CO_2 flux monitoring sites within U.S. grazing lands also indicate the potential for larger annual CO_2 sink in mixed grass and shrub systems, relative to grass-only systems (see **Figure 2**). Although carbon flux data from monitoring sites across the U.S. indicate that, across precipitation gradients, grassland sites are generally more productive than grass-shrub sites, because of higher associated respiration losses in grasslands, the grass-shrub systems might have larger net annual CO_2 sequestration (**Figure 9**). This factor should be considered in land management planning, when SOC storage and climate mitigation potential are part of the management goals.

Of additional consideration are the findings reported in the Barger et al. (2011) synthesis study which suggested productivity and carbon gain benefits associated with shrub removal within dry ecosystems having a MAP < ~340 mm (13.39 inches). This reference value has been adopted by the U.S. NRCS as part of Climate Smart management practice guidelines. We find that the Barger et al. (2011) synthesis applied a linear regression analysis based on aboveground NPP (ANPP) from multiple U.S. field sites and remote sensing studies (i.e., modeled products) and MAP. The resulting regression model indicated lesser increases in shrub ANPP at in arid regions (MAP < 336 mm), however it did not account for other factors including SOC storage and ecosystem CO_2 flux response. Based other findings in published literature, and the NEE observations for the U.S. across MAP gradients, there appears to be no viable evidence that shrub removal from arid and semi-arid systems might provide overall climate benefits.

A possibly confounding component is the impact of changing albedo associated with grassland versus grass-shrub communities. A recent study (Hasler et al., 2024) found that climate mitigation benefits associated with restoring woody (tree) cover might be overestimated if not also accounting for the lower albedo (i.e., less light reflection by plant surfaces) associated with tree communities. The albedo values of grazing land plant communities within the U.S. remain not well quantified, nor is the radiative forcing (climate impact) of plant communities under various environmental conditions (i.e., species, biomass and ground cover, precipitation and temperature) and management type (i.e., grazing, shrub removal or thinning, prescribed fire) when factoring in GHG flux and albedo. Accordingly, research is needed to identify what grazing land characteristics might collectively provide a lower radiative forcing. This would require much more understanding of seasonal, annual and longer-term GHG exchange and albedo (Kramer et al., 2021; McGregor et al., 2024) in grazing lands in response to various ecosystem conditions and management types.

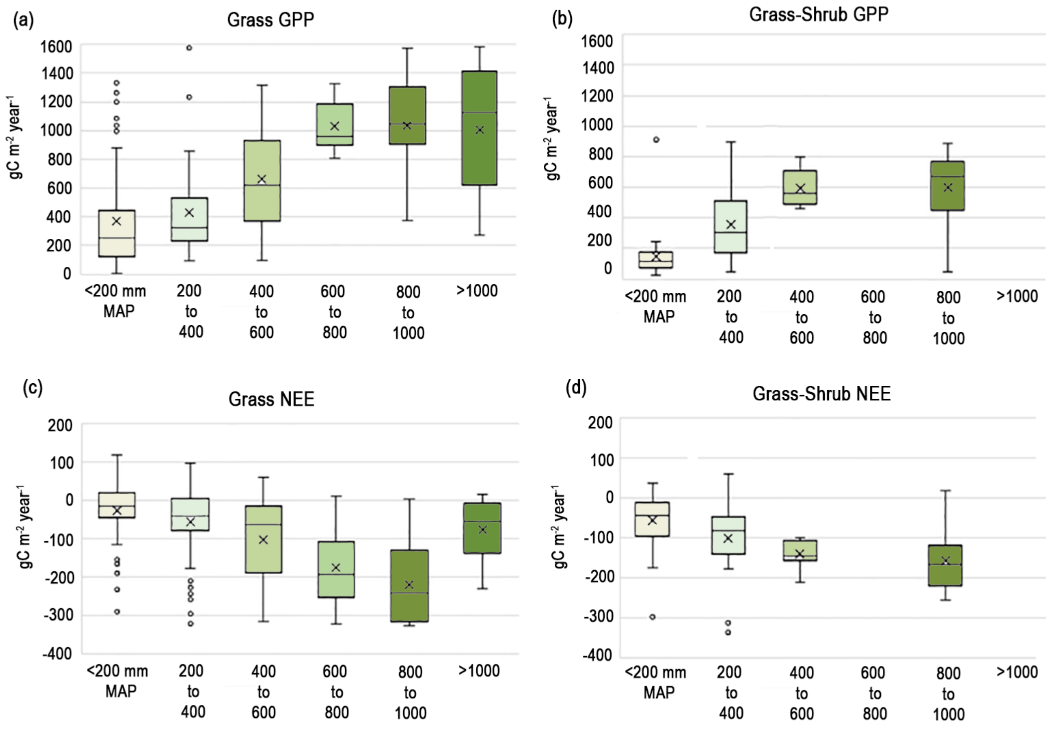

Figure 7. Plant annual gross primary productivity (GPP; gC m^{-2} $year^{-1}$) and ecosystem net annual CO_2 exchange (NEE; gC m^{-2} $year^{-1}$) flux observations from grasslands (a, c) and grass-shrub (b, d) systems within U.S. grazing lands, according to mean annual precipitation (MAP, mm) gradients. [data from AmeriFlux and Xia et al., 2025a].

In addition, more research is needed to gauge the impact of shrub and brush encroachment in U.S. grazing lands on fire risk. The occurrence of large wildfires in U.S. grazing lands (particularly in the western U.S.) has increased more than fivefold in recent decades, primarily in herbaceous/grass dominated landscapes (Li et al., 2021). In shrub-dominated landscapes, the proportion of rangeland area burned has actually decreased (as has burn severity). What is driving the decrease in wildfire within shrub-dominated rangelands is unclear, though this may be related to changing ecosystem moisture conditions. For example, lower pre-fire moisture tends to drive burns in woody biomass whereas higher pre-fire moisture conditions instead favor fine fuel loads (i.e., grass/forb biomass) (Li et al., 2022). Another factor is the spread of invasive plants, particularly cheatgrass, within grazing systems. Cheatgrass is known to be especially flammable and may increase the frequency and extent of fires (Bradley et al., 2006).

GAPS IN BRUSH MANAGEMENT KNOWLEDGE, COMMUNITY DATA NEEDS

In summary, results of the literature review described in this section indicate the importance of woody plant (including shrub) communities in providing CO_2 sequestration and carbon storage benefits across U.S. grazing lands. This factor should be considered in grazing lands management decision making, in addition to managing for increased grass forage, water storage, and biodiversity. Decisions made around shrub biomass removal to encourage increased grass and forb growth should be made carefully, considering that shrubs can function as nurse species, sheltering other plant communities and providing them access to deeper water stores through hydrologic uplift.

Based on the findings in the literature, light to moderate reductions in shrub branches (reducing canopy cover) may be favorable instead of removing whole shrubs, as “pruning” through mechanical methods and/or grazing can provide more sunlight to grass and forb communities, while also stimulating shrubs to increase CO_2 sequestration. If removal of shrubs is absolutely necessary for key management objectives, a rangeland restoration plan should be implemented that includes active land rehabilitation and reseeding to ensure plant community establishment that might offset the carbon impacts of shrub removal (see **Section 2.4**).

Although many species of shrubs exist on grazing lands within the U.S., the literature identified for this analysis primarily focused on Western Juniper (*Juniperus occidentalis*), Big Sagebrush (*Artemisia tridentata*), Velvet mesquite (*Neltuma velutina)*, Honey mesquite (*Neltuma gladulosa*), Creosote (*Larra tridentata*), Red Pine (*Pinus resinosa*), Eastern Red Cedar (*Juniperus virginiana*), and Roughleaf Dogwood (*Cornous drummondii*). To better understand the impact of shrub communities on climate mitigation, much more field research is needed to identify net annual GHG flux budgets, aboveground and belowground carbon storage, and other ecosystem characteristics (e.g., albedo, soil type, temperature, moisture, pH, nutrient availability) across more U.S. shrub species and community types (i.e., accounting for all plant species and fractional cover/biomass within the shrub system). The impacts of various management types (i.e., grazing management, biomass thinning, fire) on ecosystem services (including climate mitigation) in shrub communities is also in need of further study through structured research (including treatments and controls). Ecosystem monitoring efforts within grazing lands would also help to provide useful data needed to inform management decisions that benefit climate and local environment.

2.3 Prescribed Burning (338)

2.3.1. WHAT IS PRESCRIBED BURNING?

Natural and human-managed fire has long been a crucial factor shaping U.S. grasslands, woodlands, and other types of grazing lands (Axelrod, 1985). However, since Euroamerican settlement, significant changes in fire regimes have led to drastic ecological shifts, often resulting in entirely new or altered ecosystems (House et al., 2003; Hobbs et al., 1991). Approximately 75% of U.S. grazing lands dominated by native vegetation have deviated significantly from their historical conditions due to fire suppression and other land management changes (LANDFIRE, 2016). Since many grazing lands are inherently fire-dependent, reestablishing historical and more natural fire cycles is essential for maintaining or restoring native plant communities (Fuhlendorf et al., 2011). Without fire, many ecological sites transition toward increased woody plant dominance (Twidwell et al., 2016), leading to a decline in livestock productivity, negative impacts to native flora and fauna, and the loss of key ecosystem services (Chi and Molano-Flores, 2015). The documented rapid spread of woodlands into former grasslands in the U.S. highlights fire’s role in preserving the integrity of grazing land ecosystems (Archer, 1994; Limb et al., 2010). The NRCS Ecological Site Descriptions emphasize the need to restore historical fire regimes to sustain these landscapes (Fuhlendorf et al., 2011).

Despite the ecological importance of fire, prescribed burning has been underutilized in conservation management compared to brush management or prescribed grazing (Fuhlendorf et al., 2011). For instance, between 2004 and 2008 in Oklahoma, prescribed burning was implemented on only 84,700 ha, whereas prescribed grazing covered 919,800 ha – a difference of more than tenfold. This disparity becomes even more significant when considering that grazing management is an ongoing, multi-year practice, whereas prescribed burning is a one-time or infrequent application. The lower prioritization of prescribed burning as a resource management practice is largely driven by social and policy dynamics and constraints, which vary across regions. While certain areas, such as the Flint Hills of Kansas and Oklahoma, widely accept and integrate fire into land management, other regions actively resist its use. Legal liability and a patchwork of complex, frequently changing permit processes are also significant barriers. The convoluted interactions between government agencies, landowners, and public policy continue to shape the degree to which fire is utilized as a resource management practice, ultimately influencing the long-term sustainability of grazing land ecosystems and their services to people (Twidwell et al., 2015).

With few exceptions, fire regimes have been significantly altered due to intentional fire suppression and grazing practices that consistently reduce fuel loads (Limb et al., 2016). As a result, the absence of fire for extended periods following Euroamerican settlement has allowed both native and nonnative woody plants to invade grazing lands, leading to the transformation of many shrublands and grasslands into forests and woodlands in some regions including the Great Plains. Conversely, other regions, such as the Great Basin, have experienced widespread invasion by nonnative herbaceous (i.e., non-woody) species, which create a more uniform, flashy, and continuous fuel source (Miller et al., 2011; Balch et al., 2013). This shift has dramatically increased fire frequency. According to state-and-transition models, the dominance of woody vegetation or exotic annuals may eventually become irreversible, leading to stable alternative ecosystem states (Twidwell et al., 2013). Although most grazing land management professionals acknowledge the importance of fire in these ecosystems, numerous challenges – including historical fire exclusion, uncertainty about fire's effects, expanding wildland-urban interfaces, socioeconomic concerns, and resource policies – continue to hinder its reintroduction (Anderson and Inouye, 2001; Fischer & Lindenmayer, 2007). However, **if the management goal is to maintain grazing land ecosystem structure in alignment with historical conditions, restoring fire regimes across most grazing lands is essential, regardless of the potential carbon impacts.**

Key Finding #1. *Fire frequencies and intensities in grazing lands have changed dramatically, yet legal and permit complexities are currently a hurdle to implementing prescribed burning that would benefit ecosystems and enhance their services to people.*

HOW DOES THE NRCS DEFINE PRESCRIBED FIRE MANAGEMENT?

The NRCS Practice Standard for Prescribed Burning (Code 338) outlines several key objectives for controlled fire, including managing undesirable vegetation, managing pests and diseases, reducing wildfire hazards, improving vertebrate and invertebrate wildlife habitat, enhancing plant community structure and composition, enhancing soil health, and restoring ecological sites (USDA NRCS, 2020). To evaluate the effectiveness of prescribed burning in achieving these goals, the Conservation Effects Assessment Program (CEAP) was established to assess experimental research findings. However, because these objectives are broad and value driven, comprehensive experiment-based data are incomplete. To gain a comprehensive understanding of the ecological effects of prescribed burning, CEAP researchers analyzed literature covering a wide range of topics, including plant ecology, soil health, water resources, wildlife, arthropods, livestock management, fire behavior, smoke management, air quality, socioeconomic factors, and human health (Briske, 2011; Fuhlendorf et al., 2011; Limb et al., 2016). These topics were identified through input from CEAP grazing lands teams and an initial literature review that determined which areas had sufficient research to support meaningful conclusions. Additionally, considerations of spatial and temporal scales were incorporated into the analysis, ensuring that the findings could be directly related to the NRCS objectives for prescribed burning as a conservation practice.

2.3.2. METHODOLOGY

We identified published articles indexed in Google Scholar to collect data on the ecosystem carbon and GHG outcomes from prescribed burning on U.S. grazing lands. The search for published articles targeted experimental studies of prescribed burning that took place on a variety of grazing land types (e.g., rangeland, pastureland, shrubland, grassland, and savanna), and included at least one measured pool or flux of ecosystem carbon (e.g., soil total carbon, SOC, belowground plant carbon, aboveground plant carbon, soil respiration, CO_2 flux, CH_4 flux, N_2O flux, GPP, NPP, or NEE). Data collection was restricted to the results of field studies (i.e., not modelling or simulation studies) that experimentally applied a prescribed burn with a paired unburned control. Data collection was also restricted to the results of field studies that took place in open canopy grazing lands.

The literature search yielded over 500 published articles, of which 13 (**Appendix A Table 6**) were used for this report because they contained directly relevant quantitative data. In addition, a few recent and thorough literature reviews and syntheses of the effectiveness of prescribed burning as a conservation management tool were examined. Though these reviews and syntheses did not necessarily consider prescribed burning for climate mitigation or adaptation purposes, they contained some relevant information and data pertaining to its carbon and GHG impacts. Where these reviews and syntheses showed gaps in its carbon and GHG impacts, information and data were extracted from experimental data papers.

Report authors Drs. Jennifer Watts and Sasha Gennet also conducted an expert interview with Dr. Carissa Wonkka from University of Florida on January 16, 2025.

2.3.3. RESULTS AND DISCUSSION

PLANT PRODUCTION & COMMUNITY COMPOSITION

Plant production following prescribed burning was highly variable across studies, with increases, decreases, and no changes reported depending on ecosystem, fire intensity / season, plant functional group, and species identity.

Total aboveground biomass appeared to be negatively associated with fire frequency because repetitive prescribed burns combust aboveground biomass pools more completely (Limb et al., 2016). Such a pattern seems to suggest that prescribed burning may be a useful tool in reducing brush encroachment which has been claimed by some to have positive ecosystem carbon impacts. The evidence for that, however, remains inconclusive after a review of relevant studies.

Total herbaceous biomass tended to decrease for 1 – 2 years after prescribed burning but it was usually followed by a net increase relative to unburned controls in the long-term (> 5 years; Limb et al., 2016). This was particularly true in sagebrush-steppe grazing lands of the Great Basin. For example, most studies in this region showed that perennial grass productivity decreased in the short-term (<5 years) post-fire but increased in the long-term. Annual grass productivity, however, increased in the first post-fire growing season with *Bromus tectorum* (cheatgrass) being the notable example.

By contrast, perennial grass and forb productivity in the tallgrass prairie of the Great Plains tended to increase in the first post-fire growing season (Blair, 1997; Limb et al., 2016). Moreover, spring fires in the tallgrass prairie tended to favor late flowering and C4 plants that are more efficient at capturing atmospheric CO_2 and sequestering carbon both above- and belowground (Limb et al., 2016). This is in comparison to summer or fall fires that tend to favor early flowering and C3 plants that are less efficient at capturing and sequestering carbon (Limb et al., 2016).

In all U.S. grazing lands, shrub biomass markedly decreased following prescribed burning and remained low relative to unburned areas in the long-term (> 5 years; Limb et al., 2016). However, full recovery of shrubs occurred within 3 – 35 years, depending on the ecosystem, suggesting little or no overall long-term mortality.

Key Finding #2. *Overall, the literature only weakly supports the idea that prescribed burning increases the production of herbaceous plant communities, particularly perennial grasses. Possibly the strongest argument for the use of prescribed burning on plant communities is to maintain or restore a desired species composition. Although herbaceous vegetation rarely increased in the year of the fire, herbaceous dominance over woody plants seemed to be favored by increased fire frequency. Therefore, periodic prescribed burning may be required to maintain plant communities of deep-rooted herbaceous plants that are associated with greater carbon uptake than brush, stabilization, and storage* (*Blair, 1997; MacNeil et al., 2008).*

SOIL HYDROLOGY

When considering the climate mitigation potential of prescribed burning, the effects of prescribed burning on soil hydrology must be examined. This is because soil hydrologic processes, such as water repellency, infiltration, and erosion / runoff, indirectly affect the capacity for soil to process, stabilize, and sequester carbon.

In a literature review, Limb et al. (2016) found that many factors drove the effects of prescribed burning on soil hydrology, including aspect, slope, burn severity / frequency, and microsite. For example, burn severity is often spatially heterogenous across grazing lands due to large differences in fuel abundance and structure beneath shrubs and in the grass interspaces. This leads to similarly spatially heterogenous soil surface temperatures (Bates et al., 2011; Korfmacher et al., 2003) that create localized areas of water repellency and erosion. However, this hydrophobic effect typically diminishes within a few months as soil moisture increased, eventually returning to near pre-fire conditions.

The most significant reduction in infiltration rates and increases in soil erosion post-fire occurred beneath shrubs in shortgrass dune complexes of the southern Great Plains, which tend to have a relatively arid climate, well drained soils, and low-growing vegetation. By contrast, fire has minimal impact on infiltration and erosion in the interspaces of shrubs (Limb et al., 2016). In sagebrush-steppe grazing lands of the Great Basin, cooler and wetter sites – such as those on north-facing slopes or areas dominated by *Festuca idahoensis* (Idaho fescue) rather than *Pseudoroegneria spicata* (bluebunch wheatgrass) – experienced less fire-induced erosion than warmer and drier sites (Miller et al., 2011; Sankey et al., 2012).

Overall, the negative effects of prescribed burning on soil hydrology appear to be temporary, and the short-term alteration of soil properties are mitigated by the restabilization of the landscape via precipitation and vegetation regrowth. Therefore, any indirect effects on the capacity for soil to process, stabilize, and sequester carbon are negligible.

SOIL CARBON & CO_2 FLUX

The literature on the effects of prescribed burning on grazing lands carbon and CO_2 flux is abundant. Not surprisingly, the factors driving the effects of prescribed burning on soil carbon and CO_2 flux are complex, and include antecedent soil moisture, aspect, slope, burn severity / timing / frequency, ecosystem type, and plant community composition. Also, not surprisingly, the individual responses of soil carbon and CO_2 flux are variable. Generally, prescribed burning – regardless of burn timing and frequency – is associated with decreases in SOC (e.g., Rhoades et al., 2004; Sankey et al., 2012; Xu et al., 2022). This is particularly true of surface soil, as deep soil tends to be well-insulated against the heat of fire.

However, studies within specific ecosystems are mixed in their findings and are often related to plant community composition (Limb et al., 2016). For example, several studies within Great Basin sagebrush-steppe showed that SOC decreased post-fire where cheatgrass was the dominant herbaceous species within the community pre-fire (Bradley et al., 2006; Prater & DeLucia, 2006; Rau et al., 2011). By contrast, other studies showed that SOC did not change post-fire where perennial native grasses dominated the herbaceous plant community pre-fire (Davies et al., 2007; Jones et al., 2015).

While many studies showed decreases in SOC following prescribed burning in the short-term, many additional studies also showed decreases in CO_2 flux, or no change in aboveground NPP or NEE following prescribed burning in the short-term (e.g., Potts et al., 2012; Vargas et al., 2012). This suggests that the carbon lost from soils or combusted aboveground biomass will not be immediately re-sequestered and may require considerable time to reach pre-burned conditions.

A notable exception to these patterns are studies within the tallgrass prairie of the Great Plains. Studies in this ecosystem often show post-fire increases in SOC (Limb et al., 2016). However, the magnitude of changes were inconsequential because of the concurrently large CO_2 flux, and subsequent loss of carbon post-fire (Knapp et al., 1998). This is because prescribed burning removes accumulated litter that creates temperature and light limiting conditions for plant growth thus stimulating the productivity of herbaceous plant communities that simultaneously increases carbon belowground and increases carbon loss (Ojima et al., 1994; Blair, 1997). A negligible effect on ecosystem carbon is therefore the end result.

Key Finding #3. *Soil Carbon and CO_2 flux. Ecological theory would suggest prescribed burning can yield plant community dynamics that are often associated with increased ecosystem carbon uptake, the data on actual carbon pools and greenhouse gas fluxes are inconclusive. The effects of prescribed burning on carbon and CO_2 flux in grazing lands are well studied. Outcomes are highly varied among and within ecosystems, but in many cases carbon lost during burning is only slowly re-sequestered.*

OTHER GREENHOUSE GASES

Only one synthesis of grazing lands prescribed burning effects on N_2O flux emerged in the literature (Stephens and Homyak, 2023), and the number of studies therein were too few for formal metanalysis. Regardless, Stephens and Homyak (2023) found increases in N_2O emissions after prescribed burns, and this was likely due to increases in nitrification and denitrification.

Key Finding #4. *N_2O and Other Greenhouse Gases. Few studies have looked at N_20 as a result of prescribed burning in grazing lands, but those that did found large N_2O fluxes. Because N_2O has a very high warming potential, prescribed and wildfire N_2O emissions should be a priority for research*.

IMPORTANT FIRE MANAGEMENT CONSIDERATIONS

There are three primary mechanisms through which prescribed burning maintains grazing land health are by reducing woody vegetation, promoting native plant diversity, and preventing excessive fuel buildup that contributes to catastrophic wildfires. By reintroducing fire into fire-adapted grazing lands, prescribed burning may improve ecosystem carbon dynamics, regulate water cycles, and forage quality and availability for livestock and wildlife. Additionally, the controlled use of fire helps prevent uncontrolled wildfires, which release larger amounts of GHG and degrade air quality. As climate change drives shifts in vegetation patterns and increases fire risks, understanding how prescribed burning contributes to long-term carbon sequestration, GHG reduction, and overall ecosystem stability is increasingly relevant.

IMPORTANT KNOWLEDGE GAPS IN FIRE MANAGEMENT AND HIGH PRIORITY NEEDS

The intent of this literature review was to assess the outcomes of NRCS Conservation Practice Standard – Prescribed Burning (Code 338) on ecosystem carbon and GHG fluxes. This is particularly challenging due to factors including the limited scope of many studies in terms of time and space; insufficient documentation of conditions before, during, and after fires; and a failure to account for interactions with other disturbance processes such as brush management, prescribed grazing, and drought. Many studies also oversimplify complex fire regimes by treating them as isolated events rather than ongoing ecological processes. Finally, no study identified measured all carbon and GHG inputs and outputs necessary for the construction of a complete carbon budget, let alone more than a few pools and fluxes. Despite these limitations, a review of the literature suggests that while in theory prescribed burning can yield plant community dynamics that are often associated with increased ecosystem carbon uptake, the data on actual carbon pools and GHG fluxes (pulse emissions during and following a burn, over short-term and long-term periods) are inconclusive.

For example, prescribed burning has been shown to effectively limit woody plant invasion and stimulate herbaceous plant productivity in the long-term. Ecological theory suggests that ecosystem carbon uptake should increase under such conditions as deep-rooted perennial grasses and forbs reestablish. Their extensive root systems allow them to access water and nutrients from deeper soil layers, enabling them to sustain enhanced growth and carbon fixation via photosynthesis even during periods of drought or low surface moisture. Unlike shallow-rooted plants, which primary store carbon in aboveground biomass that decomposes more quickly, deep-rooted perennials allocate a significant portion of their carbon belowground, where it can remain in the soil for longer periods, contributing to relatively stable carbon sequestration.

Very few studies, however, provide data to support these conclusions about ecological processes. When and where increased herbaceous plant productivity occurs in response to prescribed burning, SOC often decreases. Even if SOC increases it tends to be offset by increases in CO_2 flux. The notable exception to these patterns is that of the tallgrass prairie in the Great Plains. Here, SOC does tend to increase with herbaceous plant productivity but so does CO_2 flux, rendering the potential positive carbon impact of prescribed burning negligible. Moreover, very few studies measure net ecosystem exchange with prescribed burning, and the few that did reported no changes. Finally, very few studies also measured N_2O flux with prescribed burning, and the few that did reported large fluxes. With a warming potential of 265 times that of CO_2, the GHG impact of prescribed burning as it related to N_2O could be huge.

In all, although prescribed burning can yield many ecological and ecosystem service benefits, the current state of knowledge regarding the ecosystem carbon and GHG impacts of prescribed burning in grasslands, rangelands and grazing lands does not indicate consistent GHG mitigation. While the evidence indicates that SOC may reliably increase through prescribed burning in the tallgrass prairies of the Great Plains, the changes are negligible and could be overwhelmed by N_2O fluxes with prescribed burning in this system that have not yet been studied.

2.4 Plantings, Enhancements, and Restoration

2.4.1. INTRODUCTION TO DEGRADED AND SENSITIVE LANDS

The current ecological status and change trajectories of grazing lands in the U.S., and subsequent impacts by existing states on the socioeconomic benefits offered by grazing lands, remains largely unknown and highly uncertain (McCollum et al., 2017; Xia et al., 2025b). However, it is widely understood that decades to over a century of overgrazing and cultivation of rangelands, globally and across the U.S., has depleted SOC, resulting in a deficit of ecosystem carbon relative to ecosystem storage potentials (Sanderman et al., 2017). Global estimates of achievable SOC sequestration, through added plant carbon uptake and by minimizing additional losses of SOC, are 2.3-7.3 billion tons of CO_2 equivalents per year ($CO_2e\ yr^{-1}$) through plant restoration, 148-699 megatons $CO_2e\ yr^{-1}$ for improved grazing management (based on definitions by Conant et al., 2017), and 147 megatons of $CO_2e\ yr^{-1}$ for adding legumes in pastures (Bai & Cotrufo, 2022). While there are uncertainties with these estimates, some studies suggest total SOC sequestration potential for grazing lands in the U.S. (based on management improvements) of 13-70 teragram (Tg) C yr^{-1} and a potential of 25-60 TgC yr^{-1} from soil restoration efforts (Chambers et al., 2016). However, much more detailed assessments are needed for U.S. grazing lands, especially those considering the co-regulating impacts of landscape characteristics (e.g., soil and vegetation properties, landscape position, etc.), land use (past, present, future), and changing climate.

Investing in improved land management and the restoration or reclamation of degraded lands is crucial to addressing the nation's SOC deficit. These activities provide numerous net carbon and ecosystem benefits by increasing plant carbon uptake (i.e., through improved plant cover, health, and diversity) and reducing ecosystem GHG (i.e., CO_2, CH_4, and N_2O) loss. Land *reclamation* is "the act and procedure of returning explored, distressed, and exploited land to a functioning state" (Ukhurebor et al., 2022). This may include the restoration of degraded rangelands to an improved state that increases the function of carbon, water, energy and nutrient cycles. Although the term "reclamation" is often in practice used interchangeably with "restoration", the term *restoration* should be more formally reserved for the practice of fully restoring an ecosystem to its prior, undisturbed, state (Van Andel and Aronson, 2012**).** In rangeland systems, full restoration of an ecosystem is uncommon given the high costs and resources required. Types of rangeland reclamation (or in special cases, restoration) typically found within U.S. grazing lands include: 1) pasture plantings; 2) rangeland plantings; 3) wetland and riparian restoration; and 4) herbaceous weed treatment.

2.4.2. PASTURE AND HAY PLANTING (512)

INTRODUCTION

Pasture and hay plantings (code 512) are defined as systems that have been established with adapted and compatible species, varieties, or cultivars of perennial herbaceous plants suitable for pasture or hay production (USDA, NRCS, 2020a). Pastures and hay fields are actively managed for forage production and often include non-native perennial and/or annual plant cover. Management activities on pasture and hay fields may include irrigation, fertilization, soil amendment, and tillage, in addition to direct grazing and/or cutting for hay. Because pastures are more likely to receive inputs such as fertilizer and typically involve different plant species, pasture and hay plantings are considered separately from range plantings and reclamation activities.

Management goals for pasture and hay plantings are typically aimed at increasing forage production and/or quality, but other goals such as improved habitat for wildlife may also motivate adoption. Environmental benefits can include erosion prevention, increased soil organic materials, and lower nutrient loss (e.g., in water and air). However, the extent of ecological benefits from plantings depends on the initial state of the system and the species being planted. Production and environmental outcomes also depend on the successful establishment of the target plant community.

Given the extent of pasture and hay lands in the U.S., even small changes in SOC storage or GHG emissions on a per-acre basis have potential to be scaled up to significant impacts. Ideally, plantings will be implemented in ways that increase productivity while also improving SOC storage and reducing GHG emissions, especially N_2O (i.e., co-benefits), but these outcomes will not always align and may instead present trade-offs. Identifying the conditions under which these outcomes are most likely to present as co-benefits is an urgent research need.

METHODOLOGY

A series of searches were conducted on Google Scholar to find published literature regarding GHG emissions in pasture and hay planted systems. To be included in this review, studies needed to be done in a pasture system used either for grazing or hay production or use a crop type that is typically used for livestock feed (e.g., clover) in a perennial system. The searches in Google Scholar were optimized to find papers that measured atmospheric GHG emissions and avoided enteric livestock emissions. As such, searches included terms for hay cropping type (e.g., alfalfa, clover, timothy, native grass).

The following search term was entered in Google Scholar for pasture and hay systems:

Pasture, hay, legume, alfalfa, Trefoil, clover, Timothy, Bromegrass, Orchardgrass, Canarygrass, Fescue, Ryegrass, bluegrass, greenhouse gas emission, CO_2 emission, CH_4 emission, N_2O emission, net ecosystem exchange.

The first 200 search results were scanned to assess their viability for the literature review. Only peer-reviewed, empirical publications were used as a data source. However, review papers were also scanned to find additional usable datasets. Studies needed to report at least one measured GHG (CO_2, CH_4, or N_2O) but not necessarily all three. When reported, SOC storage was also recorded. In total, 11 studies were found for pasture and hay planting systems (**Appendix A Table 7,8**).

KEY FINDINGS FOR PASTURE AND HAY PLANTING IN THE U.S.

Key Finding #1. *Conversion of cropland to herbaceous perennial vegetation including pasture and hay generally has clear positive effects on soil carbon stocks and may decrease N_2O emissions in some cases.*

In general, annual cropland systems are depleted in SOM compared to perennial plant communities on similar soil types (Sanderman et al., 2017). Therefore, establishment of perennial plantings on previous annual cropland has high potential to store additional SOM, especially where SOM is very depleted. Year-round inputs to SOM from perennial plant roots and litter combined with low soil disturbance can rebuild SOM stocks, but the growth of SOM stocks is non-linear, slowing with time as a new equilibrium state is reached. For example, studies have shown that while SOM may increase rapidly in the first 5-10 years post planting, it can take 50 years or more for newly planted grassland to reach a new equilibrium level of SOM, which may or may not match the levels observed in adjacent native grasslands (Guo and Gifford, 2002; Li et al., 2018; Poeplau et al., 2011). Gains in SOM from conversion of annual cropland to pasture or hay during the first 5-10 years of planting are typically on the order of 0.87 to 1.3 Mg C ha^{-1} yr^{-1} (~ 3 to 4.6 U.S. ton CO_2 $acre^{-1}$; Chebet et al., 2023; Conant et al., 2017), but the actual amounts are highly location specific and depend on many factors including past land use history, soil texture, climate, plant establishment and biomass production, plant functional types and species.

Perennial plantings typically have lower N losses in forms such as dissolved nutrients in runoff and gaseous N_2O than annual croplands due to lower rates of N fertilization (Bouwman et al., 2002). The inclusion of N-fixing legume species such as alfalfa and clover can further decrease N fertilization needs and may lead to even lower N losses (Anthony and Silver, 2024; Bouwman et al., 2002). However, if N fixation rates are high, such as in some legume monocrop plantings, N_2O emissions may be comparable to those from fertilized row crop systems (Robertson et al., 2000; Bouwman et al., 2002). Overall, the impacts of cropland conversion to perennial vegetation on N losses, including N_2O emissions, depend on the details of management (e.g. species planted, N additions) and the counterfactual scenario.

The potential benefits to SOC and N_2O emissions of converting annual cropland to perennial pasture or hay are clearer than they are for more nuanced changes in existing perennial plant communities (see Key Takeaways #2 and #3 for this Section). However, while the potential benefits for a given area of land are clear, the net benefits at larger scales (e.g., continental, global) are not because of the potential for "leakage". When land is converted from one use to another, there is potential for the lost land use to be compensated for by land use change elsewhere (i.e., leakage). For example, if land currently used to grow corn is converted to grass, the lost corn production may cause pressure for grassland or forest elsewhere to be converted for growing corn. Leakage poses a major risk to the overall effectiveness of land use change for climate mitigation and must be a major consideration of any policy or program designed to encourage such shifts.

Key Finding #2. *Modifying the existing plant community in perennial systems can increase SOC storage, but less so than conversion of annual cropland to perennial vegetation because the changes to the system are more nuanced (i.e., no major change in land cover).*

Many options for implementing pasture and hay plantings involve improvement of existing perennial vegetation stands (e.g., Enhancement Codes E512M, E512K, or E512L) for purposes such as improving productivity and enhancing habitat for wildlife. If planting leads to higher forage or hay production, higher carbon inputs to SOC from plant roots and litter should theoretically increase SOC stocks so long as losses (e.g., due to soil disturbance, erosion, removal through grazing and haying, or microbial decomposition) are not increased to the same degree.

Legumes such as alfalfa and clover are commonly used for hay production due to their high protein content and lower N needs due to their associations with N-fixing bacteria. Planting legumes can provide additional N to the surrounding plant community, thereby increasing productivity and carbon inputs to soils which supports SOC storage. A meta-analysis of studies on planting legumes found an average SOC sequestration of 0.66 Mg C ha^{-1} yr^{-1} (2.28 U.S. ton CO_2 $acre^{-1}$ yr^{-1}) based on 13 studies of varying lengths, measurement depths, and baseline/control conditions (Conant et al., 2017). Additional examples of studies on the impacts of legumes on SOC stocks are shown in **Appendix A Table 6**, which generally show positive impacts on SOC storage or ecosystem CO_2 uptake.

Other types of pasture plantings, for example using deeper-rooted species or higher diversity plant mixtures, are promising for increasing SOC storage based on ecological theory but have limited supporting evidence to date (Jordon et al. 2024; Whitehead 2020). A meta-analysis including four studies of sowing "improved grass species" found an average mitigation potential of about 3 Mg C ha^{-1} yr^{-1} (10.37 U.S. ton CO_2 $acre^{-1}$ yr^{-1}) but the precise improvement methods and quality of the individual studies are not clear (Conant et al., 2001). More diverse plant mixtures may increase SOC storage relative to the counterfactual. In some cases significant impacts (e.g., 11.6 Mg C ha^{-1} yr^{-1}, equating to ~40 U.S. ton CO_2 $acre^{-1}$ yr^{-1}; De Deyn et al., 2011) have been documented when functional diversity is increased using mixtures of grasses, forbs, and/or legumes (Rutledge et al., 2017; Skinner and Dell, 2016; Yang et al., 2019). Hence, some of the GHG benefits associated with "increasing plant community diversity" in the scientific literature are conflated with planting legumes.

Although the potential for SOC sequestration and N_2O emissions reductions from improvements to existing pasture and hay lands may be relatively small on a per-acre basis, their large aerial extent in the U.S. means that the total GHG mitigation potential of plant community modification is still relatively high (Henderson et al., 2015). However, an abundance of caution must be applied to climate mitigation potential estimates that assume widespread practice adoption, because significant economic, social, and political barriers hinder wide-spread implementation and continuation of new agricultural management practices.

Key Finding #3. *Reductions in N_2O emissions from pasture and hay systems are an important area for research, especially identifying specific areas or scenarios under which N_2O reductions can be reliably achieved without sacrificing production.*

Soil and plant fertility, in particular N availability, is critical for maintaining optimal forage and hay production. However, N availability also drives N losses including soil N_2O emissions. In more agronomically managed pasture and hay systems, N fertility may be managed through additions of synthetic fertilizers, which can lead to high N_2O emissions, especially soon after application and/or rain events (Flechard et al., 2007). These systems represent an opportunity to reduce soil N_2O emission through careful N management. However, many pasture and hay systems are managed extensively with fewer inputs for logistical or economic reasons, in which case N fertility may not be managed directly and soil N_2O emission may already be relatively low. In such cases, a reasonable goal is to manage for higher forage productivity without increasing N_2O emissions.

Adding legumes to existing perennial stands can provide biologically-fixed N to help maintain or increase plant biomass quantity and/or quality. Where biologically-fixed N is used to replace mineral fertilizer, adding legumes can decrease fertilizer costs and emissions associated with fertilizer production and application. It also has the potential to reduce soil N_2O emissions relative to when synthetic N fertilizers are applied (Abugandura et al., 2020), with one modelling study suggesting potential soil N_2O emission reductions of 37-40% through targeted legume-planting on managed pastures (Fuchs et al., 2020). However, empirical evidence for this is mixed (**Appendix A Table 6**) and there can be a trade-off with plant productivity when legumes are used to replace N from synthetic fertilizers (see Management Considerations).

In extensively managed grazing and hayed lands where external N sources such as mineral fertilizers are infeasible or economically prohibitive to apply, inter-seeding legumes can be a viable option for providing N to increase forage quantity and/or quality (Mortenson et al., 2005). Adding legumes may increase N_2O emissions relative to unfertilized systems with no legumes due to higher N availability (Bouwman et al., 2002), but not necessarily. Two studies have shown similar or even lower emissions of N_2O from grass-legume mixtures relative to unfertilized grass (Ingram et al., 2015; Abagandura et al., 2020). A large-scale modeling study including SOC, soil N_2O, and enteric CH_4 emissions estimated that planting legumes in amenable areas relative to maintaining pastures with no legumes and no fertilization could mitigate 1.3 Mg CO_2-eq ha^{-1} (3.45 U.S. ton CO_2-eq acre^{-1}) over 20 years in North America (Henderson et al., 2015). While there were cases where planting legumes did not significantly increase modeled N_2O emissions, there were also many places where soil N_2O emissions increased enough to outweigh benefits from SOC storage.

Given the mixed results for impacts of legume planting on N_2O emissions from pasture and hay systems, it is important to determine the geographic areas or management contexts for which this practice is likely to be most effective, to maximize benefits and minimize trade-offs. First, the counterfactual scenario, or what would have happened in the absence of the practice implementation, is extremely important. Specifically, it matters whether synthetic N fertilizer use is being decreased, and whether there are impacts on forage and hay productivity, relative to the counterfactual scenario. Second, environmental context is important in determining N_2O emissions. Current evidence suggests that soil texture, soil moisture, and SOC and N availability – which may be driven by the ratio of leguminous to non-leguminous species and time since legume planting – are important factors in determining N_2O emissions in pasture and hay systems. These factors can be incorporated into process-based models, which can then be used to predict effects of pasture and hay planting on SOC and GHG emissions across different systems and scenarios (e.g., Henderson et al., 2015; Fuchs et al., 2020). However, due to the scarcity of data available to calibrate and evaluate models to simulate N_2O emissions in grazing systems, the model results are highly uncertain. More empirical studies are needed to improve model accuracy so that they can be used to reliably inform management strategies.

MANAGEMENT CONSIDERATIONS

Pastures and hay fields are more limited in geographic scope than rangelands, but the likelihood of success of plantings is generally higher due to typically receiving more hands-on management including soil preparation, higher feasibility of inputs such as fertilizer or soil amendments, and (where applicable) systems for watering which can improve plant establishment. However, planting is still a relatively expensive and logistically challenging process which may hinder widespread adoption, especially for extensively managed pastures. There are very few papers about best approaches to the successful adoption of pasture and hay plantings given landscape and climate conditions, but one study has shown that concerns around alfalfa seeding include weather and management challenges, cost of stand establishment and longevity of seeding (Silva et al., 2021).

As with other practices, economic costs are highly variable and context specific. Kimball et al. (2015) report site preparation of $5 to 389/ha ($12 to 984/acre), seeding and planting costs ranging from $754-4,492/ha ($1,863 to $11,099/acre) and $1,857-11,440/ha/year ($4,588 to 28,268/acre/year) during the maintenance phase (after seeding) in Southern California. Further, current custom rates for drilling grass seed also vary geographically, e.g. $10-28/acre in Iowa (Plastina et al., 2024) and $14-40/acre in Nebraska (McClure et al., 2024), not including seed costs or site preparation or maintenance post-planting.

Nitrogen availability is critical to plant productivity in pasture and hay systems. Replacing synthetic fertilizer with biologically-fixed N provided by legumes can lead to lower N_2O emissions without sacrificing yields (Abugandura et al., 2020). However, if biologically-fixed N does not meet plant demand, a yield trade-off may emerge where lower N_2O emissions are accompanied by lower yields (Pannu et al., 2019). Successfully managing for lower soil N_2O emissions without sacrificing yields requires a firm grasp on a system's N status and adaptive management over time to balance forage requirements without providing excessive N that can cause N_2O emissions (McClellan et al., 2018).

GAPS IN UNDERSTANDING

There are currently few empirical studies of GHG impacts of pasture and hay plantings, especially net GHG impacts accounting for SOC storage and soil GHG emissions at field scales (**Appendix A Table 6**). Data for soil CH_4 fluxes is especially rare (Ingram et al., 2015). A critical area for improving understanding is the process of "priming", when higher plant carbon inputs increase turnover of existing SOC and lead to lower SOC stocks overall (Xu et al., 2024). This phenomenon can also stimulate N_2O emissions (Daly et al., 2024). It is currently difficult to predict where and when priming will occur, hampering our ability to predict where management activities aimed at higher plant productivity, including plantings, will positively impact SOC storage and decrease N_2O emissions. Several of the studies we investigated, that were aimed at understanding SOC and GHG impacts of plantings, were conducted at small scales (e.g., experimental plots) or in laboratory settings (e.g. Barneze et al., 2020) which may not be representative of effects on working pastures at larger scales and more complex landscapes. Modelling studies are useful for improving general understanding and guiding empirical research but suffer from high uncertainty due to scarce data used for model calibration and evaluation.

Given that the impacts of pasture and hay planting on SOC and GHG emissions is highly context-dependent, and the very little empirical data available, it remains difficult to tease apart the conditions under which SOC and GHG benefits of different types of plantings can be reliably expected. While it is generally accepted that environmental factors such as soil texture, soil moisture, and SOC and N availability are key drivers of GHG responses, the complex interactions between these factors that ultimately determine the response for a given field remain unresolved. The solution lies in efficient and rigorous measurement and monitoring of SOC and GHG impacts of plantings (prior state, immediately following conversion, and years after conversion, relative to adjacent control systems), which will lay the path for establishing mechanistic understanding and accurate prediction capability of the benefits of these practices.

2.4.3. RANGE PLANTING (550)

INTRODUCTION

Range planting (code 550) involves the seeding and establishment of herbaceous and woody species for the improvement of vegetation composition and productivity of the plant community to meet management goals (USDA, NRCS, 2020b). Planting or restoring rangeland systems is often necessary to improve degraded areas impacted by overuse, such as overgrazing, which can lead to bare soil, nutrient depletion, and the dominance of invasive weeds. Planting vegetation in rangeland systems should depend on local climate, native plant species composition, and land use history (e.g., livestock grazing or prescribed fire). In systems constrained by precipitation and water availability, shrubs and tree should be included, where appropriate, as they may provide an ecological benefit through soil water retention and hydrologic uplift (Horton et al., 1998; Seyfried et al., 2006). Although woody species are often less palatable to cattle than to browsing species such as goats, they can provide important seasonal forage including protein and are important for retaining soil structure through root biomass structures that do not turnover yearly, thereby increasing organic matter and maintaining soil moisture. While cattle, the most widespread domestic ruminant in the U.S., are nutritionally best adapted to grazing grasses, legumes, and forbs, all ruminant diets (sheep and goats) are flexible. Having a mix of herbaceous and woody species can help protect soils against erosion from extreme weather conditions such are drought, flooding, and wind (Zuazo et al., 2008). Planting these species simultaneously may help to increase the overall likelihood of plant community establishment (Gomez-Aparicio, 2009). Heavily degraded rangeland systems with low ecosystem productivity can be more nutrient and organic matter depleted than productive cropping systems (often aided by fertilizer and irrigation inputs). Therefore, degraded rangelands can be more of a 'blank slate', with the greatest potential for gains in carbon storage and GHG drawdown depending on the extent to which plant inputs are limited by climatic or other conditions.

METHODOLOGY

A series of searches were conducted on Google Scholar to find published literature regarding GHG emissions in pasture and hay planted systems. To be included in this review, studies needed to be done in a rangeland system for grazing. The searches in Google Scholar were optimized to find papers that measured atmospheric GHG emissions and avoided enteric livestock emissions. A search was conducted for range systems using the following search terms in Google Scholar: rangeland, planted, sowed, seeded, cultivated, woody, herbaceous, sagebrush, *Andropogon gerardi*, *Schizachyrium scoparium, Panicum virgatum, Sorghastrum nutans, Tripsacum dactyloides*, GHG gas, CO_2, CH_4, N_2O, NEE.

The first 200 search results were scanned to assess their viability for the literature review. Only peer-reviewed, empirical publications were used as a data source. However, review papers were also scanned to find additional usable datasets. Studies needed to report at least one measured GHG (CO_2, CH_4, or N_2O) but not necessarily all three. When reported, SOC storage was also recorded. No studies were found of plantings in rangeland systems. Five studies were found that examined GHG fluxes in established rangeland systems.

For this assessment we also interviewed rangeland reclamation experts Dr. Mark Paschke (Colorado State University) and Clay Wood (H2E, Inc).

RESULTS AND DISCUSSION

Key Finding #1. *There is no empirical evidence assessing the impact of rangeland woody and herbaceous plantings on GHG fluxes and SOC stocks. (i.e., studies have not specifically focused on carbon and climate benefits of plantings).*

Although no studies could be found that examined effects of rangeland woody or herbaceous plantings on GHG emissions or SOC stocks in the U.S., studies in established systems can be informative for understanding ecological function, plant community composition/succession, and carbon storage of rangeland systems. The literature search identified five studies in established rangelands.

A five-year study in Wyoming found that a sagebrush system released CO_2, while a nearby mixed-grass prairie had net-neutral CO_2 emissions (Hunt et al., 2004). In a two-year CO_2 assessment, there was a drawdown in the first year and a release in the second year at a site in Washington growing big sagebrush (*Artemisia tridentata*), green rabbitbrush (*Chrysothamnus viscidiflorus*), and grasses including cheatgrass (*Bromus tectorum*), bluebunch wheatgrass (*Pseudoroegneria spicata*), and Sandberg bluegrass (*Poa secunda*) (Yao et al., 2022). A sagebrush site in Oregon with 10% shrub cover and a mix of grasses was found to have much lower CO_2 emissions than a sagebrush site in Idaho with 40% shrub cover with a similar grass species composition (Gilmanov et al., 2006). A short-term study in a sagebrush system reported CH_4 sequestration but a release of CO_2 and N_2O emissions (Beltz et al., 2019). This study found no differences in emissions between control plots and those receiving hay or fertilizer (Beltz et al., 2019). CO_2 and N_2O emissions were measured near sagebrush plants surrounded by either cheatgrass or wheatgrass at a site in Wyoming. The plots growing sagebrush and western wheatgrass released more CO_2 but less N_2O than the plots growing sagebrush and cheatgrass (Norton et al., 2008).

While the rangeland systems highlighted above were largely found to release GHG, none were compared to degraded systems. Additionally, because the woody and herbaceous plants were already established, there was not a drawdown in CO_2 that is likely to occur during early succession of woody vegetation. Planting rangelands with woody and other perennial plants is also likely to increase carbon storage through increased above and belowground plant biomass. A study in Wyoming found that plots planted with sagebrush had higher aboveground biomass concentration than annual forbs or grasses after only two growing seasons, except in heavily seeded grass plots (Williams et al., 2002). Perennial plant abundance allows carbon stored in biomass to be retained, while ensuring that carbon remains in the soil. However, plant productivity in rangeland systems is usually limited by precipitation and increases in belowground carbon are likely to take considerable time to accumulate.

MANAGEMENT CONSIDERATIONS

Prior to replanting, it is recommended that rangeland sites are classified according to Ecological Site Descriptions as site-specific characteristics have a large influence on replanting establishment likelihood (NRCS, n.d.). Localized seed mixes should be chosen to maximize establishment and ensure lands are reseeded with a biodiverse native plant community. Recent efforts to streamline seeding mixes for native vegetation restoration have resulted in the International Standards for Native Seed in Ecological Restoration (Gann et al., 2019; Pedrini et al., 2020). These resources define native seeds and access seed quality and viability to develop a standardized protocol for native restoration.

Additional grower considerations such as seeding rate, fertilization, and post planting management can affect establishment and the overall GHG impact. In heavily degraded or dry soils, drill seeding and straw crimping may be needed to ensure seeds are integrated into the soil after germination. Higher seeding rate increases chance of germination and survival rate (Williams, 2002) and fertilizer application helps to promote seedling survival rate but can be expensive and is a direct and indirect emissions source. Shrub species store more carbon in aboveground biomass than grasses or forbs but take years to get established. If land is dominated by plant species (e.g., cheatgrass [*Bromus tectorum*] and juniper species that have expanded beyond their native ranges), then establishment of shrubs, forbs, or native grasses may require herbicide, which is an additional cost and GHG emissions source (production and application). It is often recommended that perennial cover reaches 70% before planted land is actively used for livestock grazing, which may require grazing exclusion or minimal, targeted grazing for at least the first 5 years or until the plant populations have fully established.

Planting deep-rooted perennial rangeland species can be effective at promoting soil organic matter accumulation (E550A), which aligns with outcomes of sequestering organic carbon and minimizing GHG emissions. Improving wildlife cover in rangelands (E550B) can allow for a more biodiverse ecosystem that helps to maintain soil structure and further aid long-term resource conservation.

GAPS IN UNDERSTANDING

There were no empirical studies that experimentally measured the effect that planting woody or herbaceous vegetation in rangeland systems had on SOC or GHG fluxes. This exposes a clear limitation in research, as any climate benefits must be estimated from modeling, ecological theory, and data collected in established rangeland systems. Without a clear control or comparison, it is challenging to provide a comprehensive examination of the quantifiable changes that result from rangeland restoration. To understand potential for SOC sequestration and GHG drawdown in planted rangeland systems, empirical measurement of these factors is needed along a timeframe that accounts for woody and herbaceous establishment (~20 years). While we recognize that such studies would require significant investment, collecting this data would help address a major limitation in rangeland science and provide support for ranchers seeking to effectively manage their lands for ecological and climate benefits.

2.4.4. WETLAND RESTORATION (657) AND RIPARIAN RESTORATION (390 AND 391)

INTRODUCTION TO WETLAND RESTORATION

Wetlands are defined as *"areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions"* (US Army Corps of Engineers for Clean Water Act Section 404 1987; EPA, 2025a). Wetlands have three primary characteristics: water tolerant vegetation, hydric soils, and soil saturation. While wetlands have potential to store carbon (see narrative below), many have been drained or reclaimed for human purposes, such as land conversion and for agricultural purposes, with recent assessments estimating a 21% decrease globally between 1770 and 2020 (Flouet-Chiounard et al., 2023).

The most recent accounting of wetlands in the conterminous U.S. sets their extent at less than 6% of land area, or 116.5 million acres (Lang et al., 2024). In this same assessment, authors found a net wetland loss of over 50% since the prior study period (2004 to 2009), with a gross loss of 221,000 acres between 2009-2019 (Lang et al., 2024). Most of this loss was in vegetated wetlands (-670,000 acres) with a net gain in non-vegetated wetlands (+488,000 acres) (Lang et al., 2024). In the U.S. Northern Great Plains and Prairie Pothole regions it is estimated that agriculture conversion (often through drainage and plantings of non-natives) has resulted in the loss of 10.1 Mg ha^{-1} (~ 35 U.S. ton CO_2 acre^{-1}) of SOC over 16 million ha (~ 39.5 Millon acres) – thus wetland restoration in these lands alone has the potential to sequester 378 Tg SOC over a 10-yr period (based on data and models) (Euliss et al., 2006). There is an opportunity to sequester an additional, potentially large, amount of carbon when also considering wetland loss that has occurred within other U.S. rangelands. In addition to their role in sequestration, wetlands have outsized value for a host of ecosystem services including water storage, water purification, flood mitigation, wildlife habitat, and native biodiversity (Jisha et al., 2021). These benefits of wetlands are based on their spatial distribution and positioning on the landscape (Mitsch & Gosselin, 2000).

Wetlands are known to be significant carbon sinks, storing about one-third of overall SOC (Lal, 2008). Wetlands are important for sequestering carbon for the long-term because they have the highest carbon density of any terrestrial ecosystem because carbon mineralization is slow in wet, oxygen-poor soils (Yang et al., 2022). In addition, vegetation in wetlands provide annual uptake of CO_2 from the atmosphere (Zou et al., 2022). Ultimately, the carbon sequestration potential of restored wetlands depends on their net balance of carbon fluxes, which includes both release of GHG gases, as well as sequestration and storage of carbon in plants and soils. Wetlands (especially newly reestablished wetlands) have the potential to release GHG that can offset sequestration of SOC. These gases include N_2O, CO_2, and CH_4. N_2O and CH_4 both have global warming potentials that are greater than that of CO_2 (273x and 27-30x respectively over a 100-year period) (EPA 2025a). CO_2 has the longest atmospheric lifetime (thousands of years), followed by N_2O (>100 years) and CH_4 (~10 years) (EPA 2025b). While N_2O, CH_4 and CO_2 all contribute to the climate impacts of wetlands, CH_4 has been a primary focus within wetland restoration research. Global simulations and measurements approximate that 25% of CH_4 produced globally is produced from wetlands (Melton et al., 2012, Bloom et al., 2010). One factor that can make distinguishing the balance of release to sequestration difficult is that these processes may occur on different timescales. Release of GHGs can happen immediately upon rewetting wetlands during restoration, while gaining the sequestration benefits of carbon in SOC may take decades.

Prior research in the U.S. and globally has focused on how histosol wetlands (whose soils are dominated by organic material) and their restoration may influence GHG emissions and carbon storage (Bridgham et al., 2006; Wilson et al., 2016). Research has focused primarily on histosols (sometimes referred to as peatlands) because of their ability to store large amounts of carbon (Limpens et al., 2008). However, two comprehensive studies of U.S.-wide wetland loss indicate that while histosol wetland restoration can have large benefits for the climate due to their high SOC holding capacity, over long periods of time (Nahlik et al., 2016), *non-histosol soils might also contribute significantly because of their more widespread distribution across the U.S. and globe* (Bridgham et al., 2006; Kochy et al., 2015; Nahlik 2016).

The NRCS created the wetland restoration conservation practice (657) as a guideline for re-establishing abiotic conditions on filled or drained wetlands in order to recreate pre-disturbance conditions. Currently, NRCS assessments indicate that wetland restoration is considered to have carbon mitigation benefits in histosols (organic material-dominated), but the practice has not been assessed for its carbon mitigation benefits in wetlands with non-histosol substrate. In this report, we assess the NRCS conservation management practice 657 (Wetland Restoration) as applied within U.S. rangeland settings and to non-histosols (i.e. mineral soils). Common supporting practices include dikes or levees (Code 356), diversions (Code 362) or water control structures (Code 587). There are a range of associated practices including practices related to planting (e.g., Codes 420, 612, 342), wildlife management (Codes 644, 646, 649) or other associated management practices, such as brush management (Code 314), herbaceous weed treatment (Code 315), forest stand improvement (Code 666), prescribed burning, (Code 338) or prescribed grazing (Code 528). This practice standard is differentiated from constructed wetland creation, in which the objective is to address pollution (Code 656), wetland enhancement, where restoration augments historic wetland features (Code 659) and creation of non-historic wetlands (Code 658). The practice has an expected lifespan of 15 years. For the purpose of this report, we focus on two common approaches used in this practice, including restoring hydrology and restoring micro-typography.

This practice applies to any location where there was a natural wetland that was disturbed or altered where the objective is to restore the area to pre-disturbance conditions. While this practice is aimed at restoration of historic wetlands, disturbed wetlands often contain non-historic wetland features that may be impractical to separate from historic wetlands, thus projects may include these features, with the overarching goal of replicating historic wetlands. This practice excludes inland or non-tidal wetlands which are common along floodplains. Instead, the report focuses on wetlands surrounding lakes and ponds, and low-lying areas (ex. vernal pools, bogs) which may be seasonally dry and are embedded within rangelands.

INTRODUCTION TO RIPARIAN RESTORATION

Riparian restoration is the process of reestablishing riparian habitat and riparian zone functions, ideally repairing the diversity and dynamics of indigenous ecosystems degraded by humans (Eubanks, 2004). Often this includes plantings of woody and herbaceous species, with a goal being the biomass-based protection of riparian soil banks from erosion. As areas both sensitive to grazing and more productive than other grazing lands, riparian areas are important to consider for their ability to sequester carbon. Currently the NRCS has two related conservation practices, riparian herbaceous cover (390) and riparian forest buffer (391).

METHODOLOGY

We adopted different methodologies for assessing the role of wetland restoration and riparian restoration in this report. We conducted a deeper and more systematic review for wetland restoration. Our exploration of riparian restoration overlapped with the plantings working group and was not a priority practice that was explored in-depth. Instead, we identified scholarship addressing this practice and conducted a non-systematic review of evidence (see description below).

Methodology for Wetland Restoration

This literature review draws on four sources. First, we examined papers listed on the NRCS publicly facing dashboard on Conservation Practices and GHG Mitigation Information. Second, we compiled published research studies using systematic and iterative searches on Web of Science (WOS) to examine how wetland restoration on non-histosol soil influences GHG emissions and carbon storage. Finally, we conducted two informational interviews with experts including: 1) Dr. Emily Ury at Environmental Defense Fund (interview with Dr. Corrie Knapp, December 12, 2024), and 2) Dr. Kristin Byrd at the Western Geographic Science Center, U.S. Geological Survey (interview with Dr. Corrie Knapp, December 5, 2024).

For the WOS search, we focused on two primary groups of restoration techniques including: **restoring hydrological function and restoring micro-typography**. We first used search terms to encompass studies focused broadly on wetland restoration, but excluding histosol soils (peats), constructed wetlands, and coastal (i.e. saline) wetlands. Our initial search was limited to 'grazing lands', but we found that this narrow focus resulted in very few applicable studies. We thus expanded our search by removing the 'grazing lands' term. Next, we searched the literature for techniques noted by the NRCS as commonly applied when conducting Wetland Restoration, including reestablishing site hydrology using dikes (356) and structures for water control (587), and for studies specifically examining restoring wetland microtopography. We targeted primary research papers, but did include results from some comprehensive syntheses (e.g. Zou et al., 2022).

Titles and abstracts of the first 200 papers were reviewed. When these indicated there was relevant information in the manuscript, or when it was unclear if there was relevant information in the manuscript, the entire manuscript was reviewed to determine if it contained relevant information.

Within searches targeting the general category of 'wetland restoration' we encountered overlapping results, which we interpret as an indicator that the pool of papers we encountered is likely representative of the published scientific research. When experiments generated data contributing to multiple manuscripts, such as when fluxes or pools were reported over the maturation of a restoration project, we included the article that reported the most complete data set and did not duplicate previous reports.

This review focused on studies occurring in the U.S. One well studied region of wetlands spanned the U.S. Canadian border – i.e. that of the Prairie Pothole Wetlands. Although we did not analyze data from this region in this report, because of the likely high relevance of studies in the Canadian Prairie Pothole region to the ecologically similar areas in the U.S. portion of the Prairie Pothole region, we collected the Canadian studies for potential future reference (**Appendix B**).

Activities with Potential Climate Benefits

Two techniques utilized for wetland restoration include **restoring hydrological function** and **restoring micro-typography**. Along with a review of literature in these relatively general techniques, we also more narrowly focused on specific activities highlighted within the existing Wetland Restoration Conservation Practice Standard (CPS 657).

One topic that was not part of our priority review, but came up within papers included in the review, was the GHG effects of adding amendments during wetland restoration (one paper). Prior knowledge of the author team also suggests that the use of methods to alter hydrology in degraded stream systems including beaver dam analogues (BDAs) and other low-tech restoration structures is an important synergistic topic, but did not show up in our review.

Methodology for Riparian Restoration

In WOS, 395 studies were identified that included the terms "riparian zone", "restoration" and "carbon". Similarly, the same search terms in Agricola yielded 936 studies in total. However, as with wetland restoration, it is unclear how many of these studies offer quantitative analysis of carbon sequestration and GHG mitigation gains following restoration (or some form of reclamation).

RESULTS AND DISCUSSION

Wetland Restoration

Restoring Hydrological Function

Restoring hydrological function includes activities which re-introduce water to previously drained or partially drained areas of **hydric soils** (Martinez-Martinez et al., 2014; Goyette et al., 2022). These activities include reinstating water flows or water pooling via the addition or removal of dikes, plugs, or other water control devices that can affect the timing, levels, and duration of water inundation. The choice of specific water control method used to restore function may vary depending on historical management, restoration budget, and local preferences (Craft, 2022). As such, we primarily focused on the outcomes of water management on GHG emissions and carbon sequestration and storage, rather than on the role of any specific method used to alter hydrological function.

Hydrological Function: Literature Review Results

Studies of GHG emissions resulting from re-wetting of previously drained or partially drained areas of hydric soils focused on the flux (i.e., release or emissions) of three primary GHGs: CH_4, N_2O, and CO_2. Carbon storage studies have focused mainly on gains of SOC but also included measurements of **total organic carbon (TOC)** and **dissolved organic carbon (DOC)**. Few hydrology studies included metrics of plant biomass including cover, above ground biomass, or root biomass.

One study examined GHG fluxes and carbon storage using a before-after wetland restoration experimental design (Sigua et al., 2009). Four studies examined SOC storage over a timeline since restoration using a space for time design (Euliss et al., 2006; Gleason et al., 2008; Reed et al., 2022). Others employed either paired comparisons of adjacent treatments or took advantage of natural or created gradients of inundation to examine effects on gas fluxes and carbon storage. Specifically, studies included:

- *Before-After* comparison of wetland GHGs and carbon storage (**Appendix A Table 10**)
- *Paired* comparisons of fluxes and storage among intact (undisturbed) wetlands vs. restored wetlands vs. unrestored wetlands (usually crop or rangelands) (**Appendix A Table 10**)
- Examination of SOC accrual in relation to *time since restoration* (**Appendix A Table 10**)
- Comparison of fluxes and pools across naturally occurring *inundation gradients*, which were sampling areas usually along a transect that transversed a pool of standing water, or areas of undergoing natural or man-made flooding and drying which allowed for remeasurement during wet-up and dry-down (**Appendix A Table 10**).

Nine of 14 studies included information on at least one of the three GHG fluxes (CH_4 = 7, N_2O = 8, CO_2 = 5 studies). SOC was examined in (10 studies) and DOC in (1 study). Five studies measured all fluxes, and three of these additionally included SOC. Studies measuring all fluxes and SOC allowed for a fuller picture of the potential climate mitigation outcomes of wetland restoration because they account for dynamic systems that are likely simultaneously emitting and sequestering climate forcing substances. There is no doubt that additional work examining all fluxes and pools simultaneously at single locations and over multiple timescales would improve our understanding of these benefits, and our results should be interpreted with this caution in mind. We found additional studies likely examining the relationships between wetland restoration and gas fluxes, but did not have time to examine all of these studies. For a list see **Appendix C**.

Hydrological Function: Literature Synthesis

> **Key Finding #1.** *Most reviewed literature indicates that restoring non-histosol wetlands via restored hydrology has high potential to have beneficial net climate mitigation outcomes.*

Paired field site studies comparing Disturbed vs. Restored vs. Unrestored Wetland areas: Most reviewed literature indicates that restoring non-histosol wetlands via restored hydrology has high potential to have beneficial net climate mitigation outcomes. Two studies found that restored wetlands did not emit higher levels of GHGs (CH_4, N_2O, CO_2) than unrestored sites (**Appendix A Table 10**, Gleason et al., 2008; Daniel et al., 2019). Additionally, one study found restored areas emitted less of each of these gases (**Appendix A Table 10**, Daniel et al., 2019). In contrast, one study found that wetlands that were drained for agricultural use and then restored led to emission of CH_4 that was not compensated for by carbon stored in other pools (SOC) (Tangen et al., 2015). This paper did not express the time since restoration, which is a factor in carbon fluxes.

Studies examining the potential of restored wetlands to store carbon mainly focused on soils. Soil storage ranged from no difference among restored and unrestored wetlands (Euliss et al., 2006; Gleason et al., 2008), to variable by site without a clear driving pattern (Tangen et al., 2015), to larger in restored wetlands (Gleason et al., 2009; Tangen & Bansal, 2020). While results from these studies sightly suggest that restoring hydrologic functioning through increased flooding may have beneficial climate mitigation outcomes, additional long-term studies in different ecosystems and soil types examining all fluxes and pools are needed to provide more conclusive evidence.

A subset of studies compared carbon fluxes and pools in restored wetlands to intact wetlands (Euliss et al. 2006; Gleason et al., 2008; Marton et al., 2014; Richards & Craft, 2015; Daniel et al., 2019). These studies offer clues about the potential of these restored wetlands to function like intact wetlands and provide more information about what this might mean for carbon balance in restored systems. In most cases GHG flux was similar in intact and restored wetlands [Daniel et al., 2019 (CH_4, N_2O, CO_2); Richards & Craft, 2015 (CH_4, N_2O)]. In one case, CH_4 release was higher in intact wetlands than restored wetlands (Daniel et al., 2019). This suggests that if restored wetlands achieve similar GHG fluxes as intact wetlands, they might either release similar amounts of these gases, or in the case of CH_4, potentially more. For SOC sequestration, studies suggest SOC levels were mostly higher in intact wetlands (Euliss et al., 2006; Marton et al., 2014; Richards & Craft, 2015), with two studies finding these levels to be indistinguishable from restored wetlands (Tangen et al., 2015 and 2020). Three space-for-time studies further suggest that SOC concentrations increase with age since restoration (Euliss et al., 2006; Gleason et al., 2008; Reed et al., 2022), while a fourth found no relationship (Gleason et al., 2009). This provides some evidence that restored areas may become larger carbon sinks over time, although further research will need to confirm if these restored sites will ever reach levels of SOC found in intact wetlands.

Key Finding #2. *Our findings provide some evidence that restored areas may become larger carbon sinks over time, although further research will need to confirm if these restored sites will ever reach levels of SOC found in intact wetlands.*

Time Since Restoration Studies: The potential for climate benefits through SOC sequestration was high within three out of four space-for-time studies, where authors looked at SOC levels across a series of restored wetlands spanning a gradient of time since restoration (Euliss et al., 2006; Reed et al., 2022; Tangen & Bansal, 2020). The Reed et al. 2022 study additionally examined increases in root biomass, which also increased over time across the time since restoration gradient. The stronger relationship in the space-for-time studies than the paired studies may be due to the longer timelines these studies encompassed, with time since restoration extending up to 35 years. In contrast, the age of restored wetlands in paired control versus restored plots was 1 year (1 study) or undefined (4 studies).

Inundation Studies: Results from inundation studies highlight the pronounced role of wetting-drying cycles in driving GHG fluxes in wetlands. Studies suggest the re-wetting process generally led to a release in CH_4, N_2O, and CO_2. Because of this, wetlands restored from land drained for agriculture released GHGs upon rewetting. After initial restoration, our results suggest that ongoing GHG release will depend on management of water levels. Maintaining a steady inundation level often reduced GHG release compared with wetlands having dynamic wetting and drying cycles. However, other ecosystem services including habitat maintenance for wildlife may depend on seasonal fluctuations of water.

While wetting and drying cycles appear to lead to release of GHGs from restored wetlands, it is possible that when hydrology is restored emissions may be offset either through carbon stored in vegetative biomass or within soils. We found only one inundation study looked at SOC storage in addition to GHG emissions (Mushet et al., 2022). When simply comparing nearby wet vs dry areas, Mushet et al. (2022) found no difference between SOC pools. However, when results were scaled up to the catchment scale, some types of wetlands -- temporarily hydrated wetlands, had two times larger SOC pools than semi-permanently pooled wetlands. This study suggests that the scale at which data was collected could influence inferences drawn from data.

Key Finding #3. *Our findings suggest that ongoing GHG release will depend on management of water levels with a steady inundation level having reduced GHG release compared to wetlands with more dynamic wetting and drying cycles.*

The Mushet et al. (2022) study also highlights the potential role of water table levels in the carbon cycle outcomes of restored wetlands. In these studies, temporarily hydrated wetlands had 2x larger CH_4 fluxes as well as 2x larger SOC pools than semi-permanently pooled wetlands. This indicates that parsing wetlands not only by soil types (histosol vs non histosol), but potentially by cyclical water regimes might be important for determining climate mitigation outcomes due to different patterns of carbon cycling. The importance of water table/level is further emphasized in a study by Zou et al. (2022), who found that as the water table declined, CH_4 emissions declined, while CO_2 and N_2O emissions increased, resulting in overall **net GHG** exchange increase as the water table declined. Importantly, as has been noted for other studies, Zou et al. did not consider CO_2 storage in soils.

No inundation studies examined how increased inundation might lead to greater plant cover/biomass, which in turn might remove CH_4 (through soil bacteria) and CO_2 from the atmosphere. One plant restoration study (Pfeifer-Meister et al., 2018), however, did find that planted wetland areas released less CO_2 than adjacent non-restored areas. A second plant restoration study (Means et al., 2016), additionally found that some plants used in wetland restoration have the potential to store increased amounts of carbon in their tissue.

These few studies suggest several key points when considering the relationship between wetland restoration and **climate forcing**. First, studies that measure all GHG fluxes, plus coinciding changes to plant communities, and changes to processes such as storage in soil and plant tissues, are needed to capture the full GHG cycle. Second, studies that occur at the same scale at which managers complete wetland restoration projects – i.e. the watershed and landscape scale, are likely the most relevant for determining whether these wetland restoration projects have beneficial climate mitigation outcomes.

Hydrological Function: Factors that Impact Carbon Mitigation Benefits

We hypothesized that several factors including the specific U.S. region where a study occurred, soil texture, and time since restoration might influence GHG flux and SOC storage. Our regional hypothesis was based on work by Nahlik and Fennessy (2016), who divided the conterminous U.S. into wetland regions (Tidal saline, Coastal Plains, Eastern Mountains & Upper Midwest, Interior Plains, West), and indicated that some of the regions had greater densities of organic carbon in their wetland soils (**Figure 9**). Some of these differences are driven by the presence of histosol wetlands, for example the Eastern Mountains and Upper Midwest wetlands, which have the highest carbon density and the highest percentage of histosol wetlands by area. However, even in regions where mineral soils are more common (Interior Plains, West), Nahlik et al. (2016) indicate that SOC density varies by region.

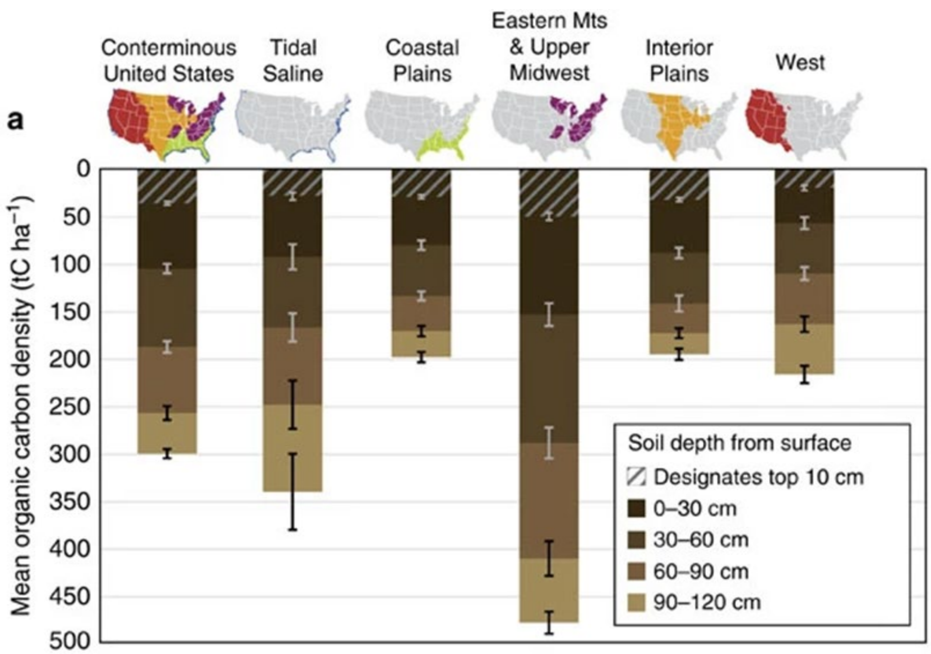

Figure 8. Carbon densities (tC ha^{-1}) are shown for the nation and in five regions: Tidal Saline (coastal and estuarine), Coastal Plains, Eastern Mountains and Upper Midwest, Interior Plains, and West. **On average, organic carbon density is largest in U.S. Eastern M Mountains and Upper Midwest soils; these wetlands contain a greater abundance of histosol soils than other regions**. The grey hatch within the bars represents the top 10 cm of the soil profile (within the 0–30 cm depth increment), followed by progressively lighter shading to represent 0–30, 30–60, 60–90 and 90–120 cm soil depths from the surface. Error bars (both white and black) represent s.e.m. (Figure credit: Nahlik & Fennessy 2016)

We expected soil texture might affect direction/magnitude of wetland restoration impact because it affects soil's ability to hold water and bind sequestered carbon through chemical bonds. Clay and silt both have a higher water holding capacity and bind significantly more carbon than sand (Jenny, 1994; Tan, 2009). As such, texture might strongly influence outcomes for GHG cycling and carbon storage in restored wetlands by influencing plant community dynamics (cover, biomass) and soil storage capacity.

Ecological evidence beyond the U.S. suggests that GHG fluxes and C storage vary among wetlands with different hydro-geomorphological characteristics (Moreno-Mateos, 2012; Goyette et al., 2022). For example, one study found areas with temporarily hydrated wetlands had 2x higher CH_4 fluxes as well as 2x larger SOC pools than semi-permanently pooled wetlands (Mushet et al., 2022). A second study found that restored wetlands with water tables below ground-level were net emitters of GHGs compared with those whose water table was at the soil surface (Zou et al., 2022). The Zou et al. study is a synthesis of data beyond the U.S. and did not focus on solely non-histosol soils. Despite this, these two studies together indicate that parsing wetlands not only by soil types (histosol vs non histosol), but potentially by annual water regimes might be important for determining the climate mitigation outcomes due to different patterns of carbon cycling.

Finally, we expected time since restoration to affect wetland carbon cycling outcomes because we expect that any restoration that alters ecosystem processes in wetlands will take time to translate into outcomes for plant communities, soil carbon accrual, and GHG cycling.

Key Finding #4. *Our findings only found evidence that the time since restoration showed climate mitigation benefits, and only for storage within SOC and root biomass. Other factors (region and soil texture) were not connected to climate mitigation benefits.*

Our collected studies spanned U.S. regions, soil types, and time frames (**Appendix A Table 9**). Despite this, we only found evidence that the time since restoration showed climate mitigation benefits, and only for storage within SOC and root biomass. This result is likely partially due to low replication of projects by location and soil texture (**Appendix A Table 9**), which prevented us from identifying any trends. An additional factor adding variability to results was the fact that very few studies collected all fluxes/pools of GHGs and carbon within single systems. One factor that may limit the existence of these types of studies is the lack of focus on carbon sequestration in older research. Ultimately, a series of replicated studies across different U.S. regions, that encompass replicated soil types, and which measure the full suite of GHG fluxes (CH_4, N_2O, CO_2) and carbon pools (SOC, above and below ground plant biomass) would best addresses how these factors affect GHG emissions and carbon storage in wetland restoration.

Hydrological Function: Gaps & Future Directions

Through the course of conducting this literature review, including conversations with subject experts Dr. Emily Ury (Environmental Defense Fund) and Dr. Kristin Byrd (USGS) (interviews with Dr. Corrie Knapp, December 2024), it became clear that studies that incorporate inputs and outputs of all carbon fluxes and pools would be helpful. One factor that may limit the existence of these types of studies is the lack of focus on carbon sequestration in older research. A series of replicated studies across different U.S. regions, that encompass replicated soil types, and which measure the full suite of GHG fluxes (CH_4, N_2O, CO_2) and carbon pools (SOC, above and below ground plant biomass) would best addresses how these factors affect GHG emissions and carbon storage in wetland restoration.

Hydrological Function: Synergistic Techniques

Beaver dam analogues (BDAs). BDAs are growing in popularity as a restoration tool across the U.S. West and elsewhere (Pearce et al., 2021). They are currently determined by the NRCS to be an important conservation enhancement activity (E643D). They are temporary-to-semi-permanent structures made to mimic beaver dams that slow water, promote pooling, and can lead to overbank flow. This has the potential to alter hydraulic connection between streams and surrounding landscapes, resulting in increased streamside complexity that includes permanent and semipermanent wetlands in non-histosol soils. While BDAs may be most applicable to Riparian Zone restoration, they may also influence wetland restoration.

BDAs are relatively low-cost when compared to restoration undertaken with large machinery. Areas restored with BDAs also have the potential to create beaver habitat. Beaver colonization can result in a self-sustaining maintenance and restoration of wetlands within stream networks. Although some studies have examined the influence of beaver dams and ponds on wetland carbon (typically reported as yielding net carbon gains; e.g., Wohl, 2013; Johnston, 2014), we did not encounter any studies directly examining the effect of BDAs on GHG fluxes or carbon sequestration. However, due to increasing interest in BDAs across the U.S. West, we recommend that future literature reviews specifically search for such studies.

Amendment additions. One study examined restoration of hydrological function in conjunction with the addition of organic amendments (Scott et al., 2022). This study examined a wetland mitigation project, which did not meet the criteria of our literature review. However, it highlighted that amendments may be used at times to influence soil properties (organic matter, bulk density, hydric indicators), vegetation (root and shoot biomass, composition), and CH_4 emissions. The amendments used included cow manure, composted wood chips, and hay. After one growing season, only wood chips increased SOC compared to unamended soils. Authors concluded that amendments may not be necessary to support vegetation and hydric soil development and might increase CH_4 release and invasive weed spread. This may be a technique that could be considered in tandem with wetland restoration provided sufficient sidebars are included to protect water quality and other habitat goals (e.g. Gravuer et al., 2019).

Restoring Microtopography

Restoring microtopography involves creating surface relief that re-establishes surface features with the goal of enhancing water and nutrient retention and increasing nutrient heterogeneity (Moser et al., 2009). Microtopography is known to be reduced when heavy machinery is used for restoration. Disking is the primary technique used to increase microtopography within restored wetlands.

The main way that microtopography might influence climate mitigation potential is through the creation of complex soil surfaces. Such surfaces influence microsite hydraulic conditions by creating wetter and drier areas and influencing overland water flow vs. retention during periods of flooding (Choi & Harvey, 2013). Because soil inundation can affect GHG fluxes and SOC sequestration (see **Appendix A Table 11**), this can potentially influence overall climate mitigation potential of these areas. Increased soil microtopography also has the potential to influence plant communities in restored wetlands, by creating a diverse suite of moisture and nutrient conditions that favor different species. It is possible that different plant community metrics including composition, diversity, and cover could affect climate mitigation potential since plants contribute to SOC accrual, store carbon within their tissues, and influence GHG cycling.

Microtopography: Literature Review Results

We did not find any studies relating the restoration of wetland microtopography to GHG fluxes, whereas five studies examined how restoration of microtopography might affect carbon storage (**Appendix A Table 11**). A variety of carbon and carbon-related pools were represented across studies. Carbon was directly measured in two types of soil pools: TOC and SOC. Carbon-related pools were represented through multiple plant metrics including: cover, diversity measured as Shannon diversity, richness measured as number of species, and composition. Note that while changes in plant pools are reported, authors did not actually measure carbon accrual in these pools.

Three studies examined the effects of disking to create microtopography (Moser et al., 2007, 2009; Ahn & Dee, 2011). The remaining two studies examined areas within restored wetlands that had different elevations (swale versus hummock) to determine how microtopography, if created, might influence SOC storage (Sleeper & Ficklin, 2016; Rossell et al., 2009).

Microtopography: Literature Review Synthesis

Soil carbon storage: There is slight evidence that creating microtopography in restored wetlands has climate mitigation benefits by increasing SOC storage. While disking compared to not disking did not appear to directly affect TOC in the 1 to 5 years following restoration (Moser et al., 2009), general studies of microtopography's correlation with TOC and SOC (via swales, flats, hummocks) found surface SOC varied depending on these features (Rossell et al., 2009; Sleeper & Ficklin, 2016). Studies did not agree on which features created areas of higher storage, with one study indicating this occurred in flats and hummocks rather than swales (Sleeper & Ficklin, 2016), and the second finding more SOC in lower areas (Rossell et al., 2009). This result is likely driven by different hydrological conditions within these sites and may indicate that increasing microtopographical complexity increases the likelihood that some areas of restored wetlands will encompass conditions beneficial to SOC accrual.

Key Finding #5. *There is slight evidence that creating microtopography in restored wetlands has climate mitigation benefits by increasing SOC storage.*

Plant carbon storage: There is slight evidence that creating microtopography in restored wetlands can yield climate mitigation benefits by increasing the potential of these wetlands to support more plants and more diverse plant assemblages. One study found that disking increased total plant cover (Ahn & Dee, 2011), which in turn increases the amount of standing biomass sequestering carbon in the system. From our team's interviews with experts Emily Ury (Environmental Defense Fund) and Kristin Byrd (USGS) (interviews with Dr. Corrie Knapp, December 2024), they described how increasing plant diversity when restoring wetland areas might additionally increase SOC storage by increasing the capacity of above ground assemblages to include species with higher carbon storing potential, or by increasing the resilience of this assemblage to perturbations that could lead to reductions in cover and biomass. For example, monocultures may not be as resilient as a diverse plant community to a disease, an extreme weather event, or flooding. Any event that reduced plant population cover/biomass would reduce community held carbon and reduce the cycling of this carbon into soils. Studies were mixed whether disking increased diversity, with one finding it did (Moser et al., 2007; metrics = richness, Shannon diversity) and a second finding it did not (Ahn & Dee, 2011; metrics = Shannon diversity, wetland prevalence index, floristic quality assessment index). Finally, one study highlighted the role of disking in preventing unwanted dominance by generalist species (Moser et al., 2007). This may be important for maintaining diversity and potentially resilience to disturbance over time. Overall, creating microtopography through disking increases the complexity of restored areas, leading to the potential for more complex herbaceous plant communities (Rossell et al., 2009). Importantly, no studies found that disking reduced cover, diversity, or richness.

Key Finding #6. *There is slight evidence that creating microtopography in restored wetlands increases the complexity of restored areas and can lead to more complex herbaceous plant communities.*

Microtopography: Factors that impact carbon mitigation benefits

Creating microtopography in restored wetlands increased complexity of both soil surfaces and, at times, plant communities. For SOC there is evidence that this complexity created niches for carbon sequestration across a suite of hydraulic conditions, such as during flooding due to weather. For plants there was some evidence that this complexity could affect plant community composition, which theoretically could have benefits for community resilience to disturbance.

Microtopography: Gaps & Future Directions

We found very few studies focused on microtopography.

Links among microtopography, plant diversity metrics, and carbon storage: More studies that focus on whether diversity leads to carbon sequestration in soils and plants would support the argument of whether creating microtopography has climate mitigation benefits. There is a very large body of literature focused on linking biodiversity to ecosystem function. This literature was not examined during this review. We recommend looking at this literature to examine whether others have related wetland planting to GHG emissions and soil and plant carbon sequestration. If there are not studies looking specifically at that question, it should be considered a research gap.

Implementation Methods

Implementation options that reliably produce GHG reductions and/or increased carbon storage: Our findings indicate that restoring hydraulic functioning to wetlands has a high potential to sequester carbon. Additional exploration of the relationship between GHG flux and SOC storage will help managers understand tradeoffs among these across short and long timescales.

Differences in implementation considerations by region: There were not enough regionally replicated studies to determine how wetland restoration affected carbon sequestration by region.

Differences in implementation considerations by other relevant factors: The factor that had moderate to strong evidence of affecting carbon sequestration included time since restoration with longer time periods related to greater sequestration (Euliss et al., 2006; Reed et al., 2022). Factors that had low evidence included creating microtopography in restored areas. Factors without enough evidence to evaluate effect included U.S. regional location, soil type, water table/level.

Results and Discussion: Riparian Restoration

Our review found that most (if not all) studies that do measure carbon sequestration following riparian zone restoration focus primarily on woody, rather than herbaceous, plants. One such study (Matzek et al., 2020) examined the results of riparian restoration across 42 revegetated sites in northern California. Plantings included riparian trees (*Salix* spp.; *Alnus rhombifolia*) and shrubs (*Sambucus mexicana*; *Symphoricarpos alba*; *Baccharis pilularis*; *Rosa acicularis*). Where revegetation was successful, they found average SOC gains of 0.87 Mg C ha^{-2} yr^{-1} (3 U.S. ton CO_2 acre yr^{-1}) in floodplain portions of the sites, and 1.12 Mg C ha^{-1} yr^{-1} (3.87 U.S. ton CO_2 acre yr^{-1}) in upper banks.

An alternative form of wetland reclamation is the re-establishment of beavers and their dams, or creation of BDAs which are built to mimic the structure of beaver dams and follow similar ecological trajectories. The reintroduction of beavers is increasingly being employed across the U.S. to assist with the impoundment of water and retention of carbon-rich sediment (necessary to support wetland habitat). The numerous ecological benefits offered by beaver dams are outlined in a document co-created by the Association of State Wetland Managers (ASWM) and the BLM (NAWM 2024). Documented benefits of beaver meadows include habitat complexity and associated biodiversity, and biomass increases, as well as sediment retention and soil water retention, which in combination result in substantial above and belowground carbon accumulation. In one study, it was estimated that beaver dam and meadow restoration would vastly increase the total carbon in those ecosystems, from 8% to 23% of the total in a watershed (Wohl et al., 2023).

Key Finding #7. *Our review found moderate to high support for carbon sequestration benefits of riparian restoration assuming that the practices increase overall plant material.*

Based on the identified literature, and foundational ecological and carbon-cycle science principles, our confidence in the direction of carbon sequestration benefits for riparian zone (including through beaver dam restoration) is moderate to high over short-term (e.g., NRCS funding cycles) and long-term (50+ years) periods, assuming the practice increases plant cover (especially woody plant cover) and sediment accumulation in the local riparian and wetland zone. However, the magnitude of net GHG impact is uncertain and should be more carefully investigated through a combination of field data analysis and ecosystem GHG modeling. As with any practice, the actual amount of carbon gains, and change in GHG mitigation, will be ecosystem- and restoration project-specific.

2.4.5. HERBACEOUS WEED TREATMENT (315)

INTRODUCTION

Herbaceous Weed Treatment is defined (according to NRCS guidelines) as the removal or control of herbaceous weeds including invasive, noxious, prohibited or undesirable plants. This practice is hypothesized to be able to decrease wildfire risk, enhance plant communities for wildlife, protect soils from erosion, and/or improve quantity and/or quality of forage and/or browse.

There is limited literature directly measuring the impacts of these practices on GHG outcomes in rangeland settings. Most studies evaluating impacts to SOC involve site comparisons for areas that have burned and are cheatgrass dominated as compared to native rangelands (e.g. Bradley et al., 2006; Koteen et al. 2011; Rau et al., 2011). However, these studies do not directly measure any impacts post treatment of herbaceous weed treatment (neither the ability for a site to improve nor any associated GHG outcomes). It is important to note that even when evaluating loss of SOC during transition from native range to annual grass-dominated range, many sites indicate mixed results. For example, Koteen et al. (2011) report lower carbon storage on sites dominated by annual grasses as compared to sites of relatively pure perennial grasses in Coastal California. However, while they hypothesize this difference may be due to lower net primary production and a shift in rooting depth and water use, they do note there are other alternative explanations such as differences in soil texture. Further, they do not track changes over time, nor do they show any potential gains from reversing invasion. Rau et al. (2011) compare outcomes for SOC across cheatgrass invasion gradients specific to the Sagebrush Steppe. Their results suggest that SOC decreases as invasion progresses, but again they use a comparison of sites, and do not track results over time. Rau et al. (2011) also do not include any estimates of potential for reversal of invasives, or any associated GHG outcomes. Bradley et al. (2006) use a site comparison approach to compare outcomes from burned, annual grass invaded sites compared to native shrubland. They state that ecosystem conversion has led to an overall loss of carbon, but primarily due to the loss of woody biomass above ground. Their results related to SOC are inconsistent, and in fact show higher carbon concentration in one of the paired grassland sites compared to the native shrubland site. Maxwell et al. (2024) also measure carbon stocks across the sagebrush steppe and estimate that sites invasion by annual grasses and increased fire events can result in soil carbon decreases of 42-49%, primarily in deep horizons. However, they emphasize that detecting actual changes can be challenging due to heterogeneity in these sites – vertically, horizontally, and temporally. They also do not predict the amount or speed of reversal of these losses if annual grasses are controlled. Katz et al. (2025) show regional variability in cheatgrass impacts on SOC, specifically that SOC stocks can be higher in the top 10 cm of soils in colder sites (e.g. Wyoming) but that was not observed in warmer sites (e.g., Colorado

Plateau). They do state, however, that this increase in SOC is primarily due to increases in the less stable particulate organic carbon (POC) as opposed to the more stable mineral-associated organic carbon (MAOC), meaning that this additional carbon storage is not necessarily stable. Katz et al. (2025) specify that it will likely take decades, or longer, to fully understand cheatgrass impacts to SOC, and do not discuss potential reversals if annual cheatgrass is controlled or discuss impacts to other deep-rooted perennial grasses.

METHODOLOGY

The search terms “rangeland”, “grazing land”, “pasture”, in combination with “control”, or “treatment”, and “carbon sequestration”, “SOC”, “Soil Carbon”, “Greenhouse Gas”, or “GHG” and “U.S.” were applied to identify multiple publications through Web of Science and Google Scholar to collect data on the ecosystem carbon and greenhouse gas outcomes from herbaceous weed treatment on U.S. grazing lands. Data collection was restricted to the results of field studies (i.e., not modelling or simulation studies) or site comparison of invaded and non-invaded sites within the U.S.

There is scant literate that tracks any carbon balance post-treatment of invasive species on grazing lands. The studies that include quantitative data related to pool or flux of ecosystem study typically utilize a comparison of carbon outcomes on paired sites, without any applied treatment. Studies that report impacts of treatment primarily discuss likelihood of reduction of invasive species and related recovery of desirable species, and mostly over short-time frames.

For this assessment we also interviewed rangeland reclamation experts Dr. Mark Paschke (Colorado State University) and Clay Wood (H2E, Inc).

RESULTS AND DISCUSSION

Key Finding #1. *The data regarding both direction and magnitude is inconclusive regarding soil organic carbon and GHG outcomes directly associated with herbaceous weed treatment within U.S. grazing land systems, especially over the enhancement life of this practice*.

Key Finding #2. *Broadcast chemical treatment of herbaceous weeds is not recommended in rangeland settings, as unintended (detrimental) harm to native plant communities might occur. Instead, careful spot treatment is recommended to avoid eliminating the carbon storage of native plant communities.*

Key Finding #3. *Combining herbaceous weed (chemical) treatment with rangeland or pasture plantings may increase likelihood of favorable climate outcomes.*

There is little empirical evidence supporting a positive GHG outcome from herbaceous weed treatment, regardless of the treatment method, particularly chemical control approaches. It is possible that biological methods such as targeted grazing may be more beneficial from a GHG perspective relative to other approaches, though more research is needed to assess these possible benefits.

In theory, removal of annual invasive grasses should lead to decreased fire risk and/or soil erosion. If more productive species (especially deep-rooted perennials) replace invasive grasses, improved net primary production and root growth could lead to increases in carbon storage, however the magnitude and timing of such improvements are difficult to predict. For example, Wood and Mealor (2022) show that the level of cheatgrass (*Bromus tectorum*) invasion impacts the response of perennial grasses post-herbicide treatment. They suggest a level of cheatgrass-to-native perennial ratio below 4:1 will likely have little impact on native production post-treatment as this ratio of invasion is likely not sufficient to suppress perennial grass production. They also state that treatment above a 10:1 ratio will likely not experience full recovery due to insufficient perennial cover, and ratios above 15:1 will require more intensive restoration efforts to see a desired response. Another study (Mumford et al., 2025) indicates that the herbidice imazapic can be useful for controlling low-abundance populations of cheatgrass, but that the use of imazapic alone did not increase native plant richness, abundance nor productivity and that other treatment factors should also be included (e.g., reseeing). Further, including adjustments of livestock stocking rates, the placement of grazing infrastructure, and the minimization of other land disturbance factors are needed to reduce risk of weed spread.

Further, Kluender et al. (2025) discuss the results of two different chemical treatment options impact cheatgrass in the sagebrush steppe. They highlight that most chemical treatments may reduce cheatgrass levels over 1-2 years, but results are generally less conclusive over longer horizons. However, they do caution that herbicide application can have unintended consequences such as non-target response and decreases in litter and increases in bare ground post-application – all of which could have negative carbon impacts, at least in the short-run. A few other considerations are the heterogeneity of coverage, specifically when application occurs via aircraft, and the potential for other exotics to take advantage of decreased competition post-treatment. In order to increase the likelihood of effective control, they recommend treatment post-fire (partly to reduce the impact of litter intercepting herbicide) and the need for reseeding post-treatment. However, as noted by Maxwell et al. (2024), waiting to control until after a fire event may result in loss of carbon stocks, minimizing the GHG impact of treatment.

While neither Wood and Mealor (2022) nor Kluender et al. (2025) track GHG outcomes, any benefit to SOC will likely be on a decadal scale, well beyond the five-year enhancement life span. For example, Yang et al. (2019) show that restoring degraded and abandoned lands may see soil carbon storage rates increase after 13 years, but even then, requiring a century or longer to attain soil carbon levels from pre-agricultural times. Further, to initiate increased carbon sequestration, Herbaceous Weed Treatment may need to be stacked with other practices such as Pasture and Hay Plantings, Range Plantings and Prescribed Burning, but these recommendations are site, and context, dependent. Endress et al. (2012) suggest that herbicide application without subsequent native perennial seeding resulted in sites becoming dominated by exotic grasses. They suggest the integration of herbicide application and seeding can be an effective restoration strategy, but successful establishment of perennial grasses was not apparent until the sixth year after seeding. Masters et al. (1996) recommend herbicide application followed by burning then reseeding as a strategy to increase likelihood of successful reclamation of leafy spurge (*Euphorbia esula*) infested rangelands.

Given the results of previous research, both the direction and magnitude of impact are uncertain from herbaceous weed treatment alone. To minimize the GHG impact of herbicide application, spot treatment is recommended over broadcast application. Reseeding in conjunction with herbicide treatment is more likely to result in (re-)establishment of desirable species, especially deep-rooted perennials that, in theory, can help sequester carbon. Other control methods (e.g. mechanical or biological – namely grazing) would likely experience the same uncertainty in GHG outcomes related to GHG footprint, with additional concerns over direct emissions and potential carbon losses related to mechanical treatment. Treating weeds to minimize soil erosion would likely result in lower carbon losses (simply from reducing overall soil losses), however the results of weed treatment are site/context specific, and there is uncertainty if removing weeds will in fact lead to better ground cover/forage composition (see Wood and Mealor, 2022). Quantifying the magnitude of impact from removing fuel loads associated with weed species (especially annual grasses) is difficult due to the uncertainty related to frequency of fire at a given location.

3. Process-Based Model Use for MMRV

3.1 Methodology Considerations

To better understand current model capabilities and areas for improvement, we reviewed four current process-based models that are capable of simulating grassland carbon and nitrogen dynamics: DayCent, DNDC, APSIM, and IFSM. The review process focused on peer-reviewed literature and online model documentation to identify relevant models and summarize their relevant attributes, such as inputs, structural characteristics, management practices that can be simulated, outputs, and breadth and depth of evaluation. We considered how model structure, input requirements, and setup procedures might affect model performance and applicability for the management practices discussed above.

3.2 Modeling

Changes in SOC and soil nutrient cycling over time in grasslands are challenging to measure because the areas of interest are typically large and ecologically heterogeneous, and many of the processes of interest are also highly variable in space and time. Process-based models are useful tools to estimate these changes through time and have the added benefit of being able to simulate different scenarios to understand what could have happened under different circumstances. While models are desirable quantification tools for practitioners, program developers, or organizations such as the NRCS, they are imperfect and their shortcomings are important to consider.

Overall, common process-based models including DayCent, DNDC, IFSM, and APSIM are capable of simulating grazing lands and representing common management practices, especially grazing management and plantings, but in simplified ways which may lead to inaccuracy (Ma et al., 2019; Wang et al., 2020). First, many of the mechanistic processes that determine SOC and GHG outcomes of the practices of interest are not explicitly represented in current process-based models. For example, the microbial communities that drive N cycling are not explicitly represented, instead their functioning is treated as a “black box”. Soil erosion and hydrological interactions, processes which drive SOC losses and GHG fluxes at the site level, are also not included explicitly in most models. Similarly, aspects of grazing land management and ecological functioning such as cattle movement, plant species diversity, and spatial heterogeneity are simplified because of the computational and technical difficulties of representing highly complex systems. For example, many models can only represent one plant species at a time, or very simply represented mixtures, for example by combining properties of the species together into an average sward. Heterogeneously applied management strategies, such as spot weed treatment or shrub pruning, are also generally difficult to implement in a modeling context and often are applied uniformly across an entire group of plants. Taken together, misrepresentation of key aspects of ecosystem complexity and functioning means that model estimates of SOC and GHG flux impacts come with high uncertainty.

Input data that drives the model is another important source of error that adds to uncertainty in model outputs (Ma et al., 2019). Running common process-based models requires input data, such as weather, management practices, soil properties, plant nutrient content, and rooting depths, that can be difficult to acquire with high precision or at all. In many cases, model inputs must be estimated by the person running the model (the “model user”), and inaccuracies in input data fuel inaccuracies in model predictions. Even with the best models, shortcomings in availability of reliable input data will limit model accuracy.

A key concern is whether model estimates are biased (i.e., show systematic error) due to omission of key mechanisms, simplification of ecosystem complexity, and estimated input data. At the very least, they are imprecise, but if they are also biased such that errors do not cancel as the area being modeled increases, then management outcomes at regional and national scales would be severely over- or underestimated (Ehrhardt et al., 2018). Unfortunately, the degree of imprecision and bias is currently difficult to assess because there is very little directly measured data across various scenarios and environmental conditions to compare model results against. While models may show good agreement with measurements at single sites (notably when the models are calibrated for those sites or conditions), more general model performance at other sites is not being tested in such studies and therefore cannot be assumed (e.g., Shepherd et al., 2019). This is true for any type of model, including both process-based and statistical models being deployed at any scale. Further, many studies claim reasonable model performance without using standard evaluation approaches or thresholds, or worse without showing statistics such as r^2, root mean square error, or bias, making it difficult to reliably judge model performance (Garsia et al., 2023). The general lack of data and rigorous evaluation for grazing lands modeling is a major area for research and investment to increase reliability and applicability of process-based models for grazing lands outcome quantification.

There is an urgent need to collect data that can be used to monitor GHG and SOC responses to practice change, especially where process-based models fall short. Integration of remote sensing data streams in combination with artificial intelligence and machine learning tools is a promising pathway for rapid MMRV (Measuring, Monitoring, Reporting, and Verification) development, including model improvement (also see discussions in Xia et al., 2025b). Improvement of current MMRV capabilities requires a two-pronged approach: (1) Identification of model improvements are likely to have the biggest impacts on the accuracy of GHG emissions and SOC stock changes, paired with (2) Targeted collection of new data and integration of existing data streams to directly inform those areas of model development. Moving forward, data collection and model development efforts must be better integrated and coordinated, working together toward the same goals of improving MMRV and reducing uncertainty in SOC and GHG emissions prediction and accounting. Model improvement efforts will ideally span multiple types of models, from point-scale biogeochemical models to landscape-scale models capable of simulating integrated processes such as watershed hydrology across management units at larger spatial resolutions (e.g., <100 m). Findings across model types can be compared and used to identify shortcomings in specific models that inform further improvements or combined in ensemble approaches.

One potential advancement in the MMRV space is the development and testing of satellite remote sensing-informed hybrid process-based diagnostic models including RangeSTAR-Carbon (e.g., Xia et al., 2025a). Unlike more complex ecosystem process models capable of prognostic model runs and future scenario simulations, diagnostic models such as RangeSTAR retain mathematical representation of carbon cycle processes, allowing for estimation of rangeland productivity, NEE, and SOC changes at higher spatial (~ 30 m) resolutions. However, because these types of models are not fully contained, they require external gridded inputs driving climate and weather conditions, vegetation properties, etc. Further, they do not explicitly track N status and impacts of land management including animal grazing indirectly accounted for through the remote sensing vegetation greenness records. Future research investments in models for grazing lands should consider both process-based (prognostic) and hybrid-based (diagnostic) approaches.

4. Management Implications – Advancing Science and Practice

Moving Forward with Limited Knowledge

Grazing lands managers have always been tasked with managing landscapes in the face of multiple uncertainties (e.g., Wilmer et al., 2020). The ecological systems in which managers work are inherently complex, heterogeneous, and dynamic. As such, it will be difficult to gain complete knowledge about application of processes in contexts with dynamic and varied characteristics such as precipitation, soil type, and management histories. The information in this report is intended to highlight practices where we have more certainty, either for or against climate benefits. We also draw attention to where additional information gathering is necessary. While it is critical that we understand relative certainty and gaps, increased awareness of existing evidence can inform, rather than stymie, decision-making.

In this report, we assess scientific evidence-based data and ecological theory about these practices. Together, they provide a body of evidence that indicates the relative certainty (or uncertainty) for each practice. Data and monitoring provide evidence about the implications of these practices but are often limited in the number of variables they measure, the contextual elements they reflect, and the duration of monitoring. Ecological theory provides important insights into ecological dynamics and hypotheses about how the system may behave. However, outcomes in specific contexts may vary, and application of theory can be strengthened by monitoring. In addition to these assessed sources of information, local knowledge and lived experience can be valuable sources of information for providing social and ecological context, experiential evidence about co-benefits, and hypotheses to be tested through data collection (Knapp & Fernandez-Gimenez, 2009).

To understand conservation practices, it is helpful to understand both the state and dynamics of an ecosystem, and the response of an ecosystem to various conditions (e.g., weather, climate, disturbance, management). For U.S. and global grazing lands, data is severely lacking and is incomplete. Obtaining necessary data will require U.S. investment in carefully controlled field studies, having baseline information, control plots and treatment plots, and active monitoring of a suite of ecosystem conditions in years following treatment. Data collection should include monitoring of ecosystem carbon storage, GHG fluxes, and in the case of fire, quantification of GHG pulse emissions from the fire itself. For grazing practices, it is important to carefully quantify grazing intensity and forage use before and after a change in management, over a long-term observation period. Careful documentation of the ecosystem properties (e.g., weather/climate information, plant species and percent cover, bare ground, soil texture, SOC concentrations and bulk density, N, root density if possible) for grazing treatment versus nearby control sites is essential. For more information about monitoring requirements, see detailed discussions in Xia et al. 2025b. Ideally, monitoring frameworks would be relatively consistent across the

U.S., allowing for consistency in regional data, making quality quantitative analyses more feasible, while allowing for a strong database that model developers can draw from for model evaluation and testing.

Although process models remain the gold standard for simulating the potential carbon and GHG implications of management approaches, at present process models for grazing land environments are deficient in their development and testing to the point that we are unable to recommend their use for tracking impacts related to prescribed grazing, brush management, fire management, weed treatment, plantings and wetlands. Future development and testing of models to address existing deficiencies will require the field and lab-based collection of crucial data needed to identify ecological response and system feedback given variability in ecosystem and climate conditions.

The evidence of anthropogenic climate change is no longer debated (USGCRP, 2023), and the experience of its impacts is increasingly felt globally; in grazing lands contexts there are increased hazards, variability, and intensity and duration of drought that demand adaptation. Climate mitigation strategies on grazing lands have been suggested to have large benefits (de Steiguer et al., 2008), and incentivizing, educating, and scaling up these strategies is critical work to avoid future loss and damage (IPCC, 2022). In the face of these issues, it is important to carefully assess how to move forward, using practices that have solid evidence, and how to learn while doing with practices with more mixed evidence.

For some practices, a combination of ecological theory and monitoring data supports climate benefits, and local knowledge of how those practices are used in specific contexts may be enough to encourage wider adoption of a practice. For other practices, ecological theory is inconclusive, and existing data (often not reflecting long-term monitoring, nor monitoring with treatment vs control sites) provides conflicting evidence; therefore, it will be important to invest in high-quality monitoring. Active adaptive management involves simultaneous experiments that allow for more rapid learning, while passive adaptive management allows learning from a single strategy at a time (Williams, 2011). The key here is that impactful learning and sharing of knowledge will require taking the time to document changes and outcomes in a quantitative manner that allows for mathematical tracking of data.

The findings from this report suggest important lessons for restructuring scientific pursuits to be more actionable, informing policy and practice. Actionable science, which often entails engaging with end-users, can create science that is both useful and useable (Bamzai-Dodson et al., 2021). Take-homes from this assessment suggest the need to establish best practices to collect standardized data across projects, collect all variables necessary to inform understanding of climate benefits, consider utility for informing policy, and work with ranchers, agencies, non-profits and funders to collect data to inform practices with high sequestration potential and interest from producers in adoption. While the first two suggestions relate to the structure of research efforts to increase learning across projects, the second two relate to creating knowledge that is useful and useable through engagement with end users. While there are a range of types of engagement that can lead to actionable science (Bamzai-Dodson et al., 2021), knowledge co-production is interactive, pluralistic, goal-oriented and context-based (Norstrom et al., 2020). A recent paper suggests that science may be more actionable if it focuses on management decision-making, gives priority to practices and outcomes over products, and builds connections across sectors and disciplines (Beier et al., 2016)

While this assessment provides guidance about the state of our ecological knowledge of climate smart practices, it does not fully review social and contextual factors that may influence adoption (social context, economic implications, management histories). To prioritize conservation practice standards that will have a climate benefit, it is important to understand not only the ecological dynamics that would lead to potential carbon sequestration, but also the potential for adoption and scaling up. Factors such as values, tradeoffs between yields and climate benefits, alignment with existing management, and manager perceptions are as important to understand, as these factors will move the potential of a practice into actual carbon benefits.

5. Conclusions

Protect Intact Rangelands and Pastures from Severe Disturbance and Conversion

From a climate mitigation and ecological perspective, the protection of large carbon stocks (aboveground and belowground) and sequestration capacity that exist in U.S. rangelands and pastures should be prioritized. This requires identifying and protecting healthy landscapes that currently hold considerable amounts of carbon and provide substantial net CO_2 uptake through their intact plant and soil systems. Protecting these landscapes requires avoiding soil disturbances, land conversion to non-perennial systems, degradation through overgrazing, avoiding severe fires, etc. Whenever possible, intact native systems should be protected from the introduction of invasive species.

Grazing Can Be Beneficial; Avoiding Overgrazing is Crucial

The literature clearly indicates the long-lasting detrimental impacts on plant ecosystems, their CO_2 sequestration, and SOC storage following overgrazing events that lead to decreases in healthy plant systems, plant mortality, and increases in bare soil. Overgrazing of rangelands and pastures should be avoided at all costs. Preventing overgrazing requires the detailed and active monitoring of grazing systems to effectively identify how much grazing is viable given existing ecosystem and weather conditions, to confirm throughout the grazing period that plants remain healthy, and to quickly remove livestock when there are signs that plants are becoming too stressed. Doing so requires the thoughtful design, implementation, and adaptation of grazing management plans that are tied to detailed ecosystem monitoring plans. In non-humid rangeland settings, continuous grazing with high stocking rates should be avoided.

There is some evidence in the literature suggesting that low to moderate intensity grazing by livestock may increase plant productivity and the robustness of belowground root systems in rangelands, providing increased resilience to drought, relative to those ungrazed by livestock. However, at present, the literature does not provide evidence that a change from continuous grazing to rotational (adaptative grazing) in rangeland settings will yield increases in plant sequestration and SOC storage. In contrast, there is some evidence that these benefits might occur in more humid pasture settings. In both cases, more data, carefully obtained through treatment vs control methods and long-term monitoring of ecosystem conditions where a suite of metrics are assessed, is needed to further evaluate the potential of management shifts to yield desired ecological responses.

Shrubs Can be Important for Carbon and Sequestration

There is evidence in the literature that encroachment by woody species into U.S. grasslands, may provide carbon sequestration and carbon storage (aboveground and belowground) that should be considered in management plans for U.S. rangelands. Caution should be taken if management goals include the removal of woody biomass (removal of plants, not just trimming) with the intention of increasing grass and herbaceous plants for livestock forage. If this is the case, removal of woody biomass should be coupled with active replanting or reseeding efforts, followed by monitoring to ensure that grass/herb revegetation is successful. The planning activity should acknowledge that short-to-long term deficits in ecosystem-level carbon sequestration and SOC may occur.

Prescribed Burning Can Benefit Ecosystem Structure and Function, Though May Result in Carbon Deficits

Prescribed burning, historically through Indigenous management, has been fundamental in shaping ecological landscapes across the U.S. and elsewhere. Decades of fire suppression have resulted in increased fuel loads and the risk of severe fire. This has led to an increasing movement across the U.S. to reintroduce low severity prescribed fires as a landscape management tool. Although prescribed burning may undoubtedly benefit some aspects of ecosystem structure and function (e.g., biodiversity), the impact of burning on carbon and GHG budgets remains unclear. Future studies will benefit from quantifying fire emissions during the burn event, and the short-and longer-term (multi-year) impacts on plant carbon sequestration (i.e., NEE), aboveground and belowground carbon stocks, and GHG emissions (including CH_4 and N_2O if wet soils occur).

Plantings, Enhancements, and Restoration are Important Investments in Rangeland and Pasture Health

Many U.S. rangelands and pastures have become degraded after years of land use, overgrazing, weed infestation and the loss of intact native, biodiverse, environments. Enhancement efforts that include reseeding and replanting, and in some cases full restoration efforts, are necessary in many landscapes. A first step in this process might warrant the comprehensive evaluation of rangeland and pasture status, to identify and quantify areas of intact and healthy lands versus those moderately to severely degraded. Healthy lands should be prioritized for long-term protection using appropriate management approaches. Rehabilitation and restoration plans should be developed for degraded lands. Management plans should also be implemented to ensure that replanting and other enhancements yield the designed ecological changes.

Although there is a lack of data in existing literature regarding the potential magnitudes and rates of improvement in ecosystem carbon and GHG status under various environmental conditions following plantings and enhancements, ecological theory suggests that moving from degraded land conditions having much higher percentages of bare ground relative to similar, nearby, undegraded lands to land conditions having diverse, intact, plant communities should increase plant carbon sequestration and SOC storage.

Wetland Protection and Restoration Are Important for Climate Mitigation

For all wetlands (histosol and non histosol), avoiding conversion is the easiest way to avoid carbon emissions. Evidence within published literature moderately to strongly indicates that restoration of non histosol wetlands by restoring hydrology has more positive impacts than creating microtopography. When wetlands are restored, there is some evidence that they may benefit from increasing carbon sinks over time, although further research is required to confirm if restored sites will ever reach levels of SOC found in intact wetlands.

An important factor is the management of water levels. Maintaining a steady inundation level often reduced GHG release compared with wetlands having dynamic wetting and drying cycles. However, other ecosystem services including habitat maintenance for wildlife may depend on seasonal fluctuation of water. The literature on microtopography suggests that it can yield climate mitigation benefits by increasing the potential of these wetlands to support more plants and more diverse plant assemblages. Literature for riparian plantings indicates moderate to high evidence towards increases in carbon storage over short-term (e.g., NRCS funding cycles) and long-term (50+ years) periods, assuming the practice increases plant cover and sediment accumulation in the local riparian and wetland zone.

For wetland restoration, additional research is needed to examine all GHG fluxes (CO_2, CH_4, N_2O) and (aboveground and belowground) carbon pools simultaneously at single locations and over multiple timescales to improve our understanding of wetland restoration benefits. In addition, studies that occur at the same scale at which managers complete wetland restoration projects – i.e. the watershed and landscape scale, are likely the most relevant for determining whether these wetland restoration projects have beneficial climate mitigation outcomes.

More Data Are Needed to Develop Robust Ecological Models for U.S. Grazing Lands

There remains a lack of comprehensive data regarding ecological states, dynamics and feedbacks within complex and diverse rangelands and pastures within the U.S. Without these data, efforts to develop and test process models capable of accurately mimicking ecosystems and ecosystem responses to management (and climate, disturbance) will be futile. Investments in carefully designed studies and rigorous monitoring efforts that aim to deliver comprehensive, high quality, datasets detailing multiple metrics of ecosystem condition and change over time (relative to baseline and control sites; see Xia et al., 2025b) are urgently needed; as are efforts to coordinate between model developers and those designing and conducting field studies, and the groups/ranchers designing and conducting long-term monitoring of grazing land conditions. The use of geospatial technologies, including multi-spectral remote sensing, should be integrated with future modeling and monitoring efforts.

6. References

Abagandura, G.O., U. Sekaran, S. Singh, J. Singh, M.A. Ibrahim, S. Subramanian, V.N. Owens, and S. Kumar. "Intercropping Kura Clover with Prairie Cordgrass Mitigates Soil Greenhouse Gas Fluxes." *Scientific Reports* 10 (2020): 7334. doi:10.1038/s41598-020-64182-2.

Abdalla, M., A. Hastings, D. R. Chadwick, Davey L. Jones, C. D. Evans, Michael B. Jones, R. M. Rees, and P. E. T. E. Smith. "Critical Review of the Impacts of Grazing Intensity on Soil Organic Carbon Storage and Other Soil Quality Indicators in Extensively Managed Grasslands." *Agriculture, Ecosystems & Environment* 253 (2018): 62-81.

Abdallah, A.B., R. Mata-Gonzalez, J.S. Noller, C.G. Ochoa. "Ecosystem carbon in relation to woody plant encroachment and control: Juniper systems in Oregon, USA. *Agriculture, Ecosystems & Environment (2020)*, 290. https://doi.org/10.1016/j.agee.2019.106762.

Ahn, Changwoo, and Suzanne Dee. "Early Development of Plant Community in a Created Mitigation Wetland as Affected by Introduced Hydrologic Design Elements." *Ecological Engineering* 37, no. 9 (2011): 1324-1333.

Allen, V.G., C. Batello, E.J. Berretta, et al. "An International Terminology for Grazing Lands and Grazing Animals." *Grass Forage Science* 66 (2011): 2-28.

Amorim, H.C.S., A.J. Ashworth, P.A. Moore Jr., B.J. Wienhold, M.C. Savin, P.R. Owens, S. Jagadamma, T.S. Carvalho, and S. Xu. "Soil Quality Indices Following Long-Term Conservation Pasture Management Practices." *Agriculture, Ecosystems & Environment* 301 (2020). https://doi.org/10.1016/j.agee.2020.107060.

Anderson, J.E., and R.S. Inouye. "Landscape-Scale Changes in Plant Species Abundance and Biodiversity of a Sagebrush Steppe over 45 Years." *Ecological Monographs* 71 (2001): 531-556.

Andrews, Holly M., Peter M. Homyak, Patty Y. Oikawa, Jun Wang, and G. Darrel Jenerette. "Water-Conscious Management Strategies Reduce Per-Yield Irrigation and Soil Emissions of CO2, N2O, and NO in High-Temperature Forage Cropping Systems." *Agriculture, Ecosystems & Environment* 332 (2022): 107944.

Angerer, J.P., W.E. Fox, J.E. Wolfe, D.R. Tolleson, and T. Owen. "Chapter 20 -- Land Degradation in Rangeland Ecosystems." In *Biological and Environmental Hazards, Risks and Disasters*, 2nd ed., Hazards and Disasters Series, 395-434. 2023. doi.org/10.1016/B978-0-12-820509-9.00007-1.

Ansley, R.J., M.J. Castellao. "Strategies for savanna restoration in the Southern Great Plains: effects of fire and herbicides." *Restoration Ecology* 45: 420-428.

Anthony, Tyler L., and Whendee L. Silver. "Hot Spots and Hot Moments of Greenhouse Gas Emissions in Agricultural Peatlands." *Biogeochemistry* 167, no. 4 (2024): 461-477.

Apfelbaum, S.I., R. Thompson, F. Wang, S. Mosier, and R. Teague. "Vegetation, Water Infiltration, and Soil Carbon Response to Adaptive Multi-Paddock and Conventional Grazing in Southeastern USA Ranches." *Journal of Environmental Management* 308 (2022).

Archer, S. "Woody Plant Encroachment into Southwestern Grasslands and Savannas: Rates, Patterns and Proximate Causes." In *Ecological Implications of Livestock Herbivory in the West*, edited by M. Vavra, W.A. Laycock, and R.D. Piper, 13-68. Denver, CO: Society for Range Management, 1994.

Archer, S.R. "Rangeland conservation and shrub encroachment: new perspectives on an old problem." In: Wild Rangelands: conserving wildlife while maintaining livestock in semi-arid ecosystems. Eds. J.T. du Toit, R. Kock, J.C. Deutsch. Pp. 53-97. ISBN: 9781405177856.

Axelrod, D.I. "Rise of the Grassland Biome, Central North America." *Botanical Review* 51 (1985): 163-201.

Bai, Y., and M.F. Cotrufo. "Grassland Soil Carbon Sequestration: Current Understanding, Challenges, and Solutions." *Science* 377 (2022): 603-608. doi: 10.1126/science.abo2380.

Bailey, Derek W., Jeffrey C. Mosley, Richard E. Estell, Andres F. Cibils, Marc Horney, John R. Hendrickson, John W. Walker, Karen L. Launchbaugh, and Elizabeth A. Burritt. "Synthesis Paper: Targeted Livestock Grazing: Prescription for Healthy Rangelands." *Rangeland Ecology & Management* 72, no. 6 (2019): 865-877.

Balch, J. K., B. A. Bradley, C. M. D'Antonio, and J. Gómez-Dans. "Introduced Annual Grass Increases Regional Fire Activity across the Arid Western USA (1980–2009)." *Global Change Biology* 19, no. 1 (2013): 173-183. https://doi.org/10.1111/gcb.12046.

Bamzai-Dodson, A., A. E. Cravens, A. A. Wade, and R. A. McPherson. "Engaging with Stakeholders to Produce Actionable Science: A Framework and Guidance." *Weather, Climate, and Society* 13 (2021): 1027-1041. https://doi.org/10.1175/WCAS-D-21-0046.1.

Barger, Nichole N., S.R. Archer, J.L. Campbell, C. Huang, J.A. Morton, A.K. Knapp. "Woody plant proliferation in North American drylands: a synthesis of impacts on ecosystem carbon balance. *JGR Biogeosciences* 116. https://doi.org/10.1029/2010JG001506.

Barneze, Arlete S., Jeanette Whitaker, Niall P. McNamara, and Nicholas J. Ostle. "Legumes Increase Grassland Productivity with No Effect on Nitrous Oxide Emissions." *Plant and Soil* 446, no. 1 (2020): 163-177. https://doi.org/10.1007/s11104-019-04338-w.

Barsotti, Joy L., Upendra M. Sainju, Andrew W. Lenssen, Clifford Montagne, and Patrick G. Hatfield. "Net Greenhouse Gas Emissions Affected by Sheep Grazing in Dryland Cropping Systems." *Soil Science Society of America Journal* 77, no. 3 (2013): 1012-1025.

Bates, J. D., K. W. Davies, and R. N. Sharp. "Shrub-Steppe Early Succession Following Juniper Cutting and Prescribed Fire." *Environmental Management* 47, no. 3 (2011): 468-481. https://doi.org/10.1007/s00267-011-9629-0.

Beier, P., L. J. Hansen, L. Helbrecht, and D. Behar. "A How-to Guide for Coproduction of Actionable Science." *Conservation Letters* (2016). https://doi.org/10.1111/conl.12300.

Beltz, C.W., M.L. Mobley, and I.C. Burke. "Nitrogen Addition Pulse Has Minimal Effect in Big Sagebrush (Artemisia tridentata) Communities on the Pinedale Anticline, Wyoming (USA)." *PLoS One* 14, no. 5 (2019): e0206563. doi: 10.1371/journal.pone.0206563.

Bi, J., R. Hao, J. Li, and J. Qiao. "Identifying Ecosystem States with Patterns of Ecosystem Service Bundles." *Ecological Indicators* 131 (2021): 108195. https://doi.org/10.1016/j.ecolind.2021.108195.

Billings, S.A., K. Lajtha, A. Malhotra, A.A. Berhe, M-A. de Graaff, J. Fraterrigo, K. Georgiou, et al. "Soil Organic Carbon Is Not Just for Soil Scientists: Measurement Recommendations for Diverse Practitioners." *Ecological Applications* (2021): e02290. https://doi.org/10.1002/eap.2290

Blair, J. M. "Fire, N Availability, and Plant Response in Grasslands: A Test of the Transient Maxima Hypothesis." *Ecology* 78, no. 8 (1997): 2359-2368. https://doi.org/10.1890/0012-9658(1997)078[2359:FNAAPR]2.0.CO;2.

Bloom, A.A., P.I. Palmer, A. Fraser, D.S. Reay, and C. Frankenberg. "Large-Scale Controls of Methanogenesis Inferred from Methane and Gravity Spaceborne Data." *Science* 327 (2010): 322-325.

Buffington, L.C., and C. H. Herbel. “Vegetational Changes on a Semidesert Grassland Range from 1858 to 1963.” *Ecological Monographs*, 35: 139-164.

Bureau of Land Management (BLM). *Interpreting Indicators of Rangeland Health* [Version 5]. U.S. Department of the Interior Bureau of Land Management Technical Reference 1734-6. 2020. https://www.blm.gov/documents/national-office/blm-library/technical-reference/interpreting-indicators-rangeland-health-0

Bonin, C., J. Flores, R. Lal, and B. Tracy. "Root Characteristics of Perennial Warm-Season Grasslands Managed for Grazing and Biomass Production." *Agronomy* 3 (2013): 508-523. doi.org/10.3390/agronomy3030508.

Bouwman, A.F., L.J.M. Boumans, and N.H. Batjes. "Emissions of N2O and NO from Fertilized Fields: Summary of Available Measurement Data." *Global Biogeochemical Cycles* 16 (2002): 6-1-6-13. doi:10.1029/2001GB001811.

Bradley, Bethany A., R.A. Houghton, John F. Mustard, and Steven P. Hamburg. "Invasive Grass Reduces Aboveground Carbon Stocks in Shrublands of the Western US." *Global Change Biology* 12 (2006): 1815-1822. doi.org/10.1111/j.1365-2486.2006.01232.x.

Bridgham, Scott D., J. Patrick Megonigal, Jason K. Keller, Norman B. Bliss, and Carl Trettin. "The Carbon Balance of North American Wetlands." *Wetlands* 26, no. 4 (2006): 889-916.

Briske, D.D., ed. *Conservation Benefits of Rangeland Practices: Assessment, Recommendations, and Knowledge Gaps*. Washington, DC: USDA-NRCS, 2011.

Brown, Joel R., and Jeffrey E. Herrick. "Making Soil Health a Part of Rangeland Management." *Journal of Soil and Water Conservation Society* 71 (2016): 55-60. doi: 10.2489/jswc.71.3.55A.

Brunson, M.W., L. Huntsinger, G.R. Meredith, and N. Sayre. "The Future of Social Science Integration in Rangelands Research." *Rangelands* (2021).

Cao, D., J. Zhang, T. Zhang, F. Yao, R. Ji, S. Zi, H. Li, and Q. Cheng. "Spatiotemporal Variations and Driving Factors of Global Terrestrial Vegetation Productivity Gap under the Changing of Climate, CO2, Landcover, and N Deposition." *Science of the Total Environment* 880 (2023): 162753. https://doi.org/10.1016/j.scitotenv.2023.162753.

Carbone, M.S., and S.E. Trumbore. "Contribution of New Photosynthetic Assimilates to Respiration by Perennial Grasses and Shrubs: Residence Times and Allocation Patterns." *New Phytologist* 176 (2007): 124-135. https://doi.org/10.1111/j.1469-8137.2007.02153.x.

Caspi, Tal, Leo Estrada, Anna V. Dowling, Erin Su, Maxim Leshchinskiy, André RO Cavalcanti, Edward J. Crane III, Colin R. Robins, and Wallace M. Meyer III. "Carbon and Nitrogen in the Topsoils of Inceptisols and Mollisols under Native Sage Scrub and Non-Native Grasslands in Southern California." *Geoderma Regional* 14 (2018): e00172.

Chambers, A., et al. “Soil carbon sequestration potential of U.S. croplands and grasslands; Implementing the 4 per thousand initiative.” *Journal of Soil and Water Conservation* 71:68-74.

Chan, A.S.K., and T.B. Parkin. "Methane Oxidation and Production Activity in Soils from Natural and Agricultural Ecosystems." *Atmospheric Pollutant and Trace Gas* (2001). https://doi.org/10.2134/jeq2001.1896.

Chebet, S., L.J. Munkholm, and J.L. Jensen. "Rapid Increase in Soil Organic Carbon and Structural Stability in a Sandy Loam Soil Following Conversion from Long-Term Arable to Semi-Natural Grassland Irrespective of Initial Soil Conditions." *Geoderma* 438 (2023): 116646. doi:10.1016/j.geoderma.2023.116646.

Chi, K., and B. Molano-Flores. "Degradation of Habitat Disrupts Plant-Pollinator Interactions for Rare Self-Compatible Plant." *Plant Ecology* 216 (2015): 1275-1283.

Choi, Jungyill, and Judson W. Harvey. "Relative Significance of Microtopography and Vegetation as Controls on Surface Water Flow on a Low-Gradient Floodplain." *Wetlands* 34 (2013): 101-115.

Collier, Sarah M., Andrew P. Dean, Lawrence G. Oates, Matthew D. Ruark, and Randall D. Jackson. "Does Plant Biomass Manipulation in Static Chambers Affect Nitrous Oxide Emissions from Soils?" *Journal of Environmental Quality* 45, no. 2 (2016): 751-756.

Conant, R.T., K. Paustian, and E.T. Elliott. "Grassland Management and Conversion into Grassland: Effects on Soil Carbon." *Ecological Applications* 11 (2001): 343-355. doi:10.1890/1051-0761(2001)011[0343:GMACIG]2.0.CO;2.

Conant, R.T., C.E.P. Cerri, B.B. Osborne, and K. Paustian. "Grassland Management Impacts on Soil Carbon Stocks: A New Synthesis." *Ecological Applications* 27 (2017): 662-668. doi:10.1002/eap.1473.

Connell, R. Kent, Jesse B. Nippert, John M. Blair. “Three decades of divergent land use and plant community change alters soil C and N content in Tallgrass Prairie.” *JGR Biogeosciences* 125. (2020) https://doi.org/10.1029/2020JG005723.

Connin, S.L., R.A. Virginia, and C.P. Chamberlain. "Carbon Isotopes Reveal Soil Organic Matter Dynamics Following Arid Land Shrub Expansion." *Oecologia* 110 (1997): 374-386.

Coultrap, D.E., K.O. Fulgham, D.L. Lancaster, J. Gustafson, D.F. Lile, and M.R. George. "Relationships between Western Juniper (Juniperus occidentalis) and Understory Vegetation." *Invasive Plant Sciences and Management* 1 (2008): 3-11.

Craft, Christopher. *Creating and Restoring Wetlands: From Theory to Practice*. Cambridge, MA: Elsevier, 2022.

Dai, X. T.W. Boutton, M. Hailemichael, R.J. Ansley, K.E. Jessup. “Soil carbon and nitrogen storage in response to fire in a temperate mixed-grass savanna.” *J. Environmental Quality* 35: 1620-1628.

Daly, E.J., G. Hernandez-Ramirez, K.A. Congreves, T. Clough, C. Voigt, E. Harris, and R. Ruser. "Soil Organic Nitrogen Priming to Nitrous Oxide: A Synthesis." *Soil Biology and Biochemistry* 189 (2024): 109254. doi:10.1016/j.soilbio.2023.109254.

Dangi, S.R., P.D. Stahl, E. Pendall, M.B. Cleary, J.S. Buyer. “Recovery of soil microbial community structure after fire in a sagebrush-grassland ecosystem.” *Land Degradation & Development*, 21: 423-432.

Daniel, Dale W., Loren M. Smith, Scott T. McMurry, Brian A. Tangen, Charles F. Dahl, Ned H. Euliss Jr., and Ted LaGrange. "Effects of Land Use on Greenhouse Gas Flux in Playa Wetlands and Associated Watersheds in the High Plains, USA." 2019.

Daryanto, Stefani, David J. Eldridge, and Heather L. Throop. "Managing Semi-Arid Woodlands for Carbon Storage: Grazing and Shrub Effects on Above- and Belowground Carbon." *Agriculture, Ecosystems & Environment* 169 (2013): 1-11.

Davies, K.W., J.D. Bates, and R.F. Miller. "Short-Term Effects of Burning Wyoming Big Sagebrush Steppe in Southeast Oregon." *Rangeland Ecology & Management* 60 (2007): 515-522.

De Deyn, Gerlinde B., Robert S. Shiel, Nick J. Ostle, Niall P. McNamara, Simon Oakley, Iain Young, Christopher Freeman, Nathalie Fenner, Helen Quirk, and Richard D. Bardgett. "Additional Carbon Sequestration Benefits of Grassland Diversity Restoration." *Journal of Applied Ecology* 48, no. 3 (2011): 600-608.

DeLonge, Marcia, and Andrea Basche. "Managing Grazing Lands to Improve Soils and Promote Climate Change Adaptation and Mitigation: A Global Synthesis." *Renewable Agriculture and Food Systems* 33 (2017): 267-278. doi:10.1017/S1742170517000588.

DeMarco, Jennie, Timothy Filley, and Heather L. Throop. "Patterns of Woody Plant-Derived Soil Carbon Losses and Persistence after Brush Management in a Semi-Arid Grassland." *Plant and Soil* 406 (2016): 277-293. doi.org/10.1007/s11104-016-2880-7.

Derner, Justin D., Gerald E. Schuman, Ronald F. Follett, and George F. Vance. "Plant and Soil Consequences of Shrub Management in a Big Sagebrush-Dominated Rangeland Ecosystem." *Environment and Natural Resources Research* 4 (2014). doi: 10.5539/enrr.v4n1p19.

Derner, Justin D., Bob Budd, Grady Grissom, Emily J. Kachergis, David J. Augustine, Hailey Wilmer, J. Derek Scasta, and John P. Ritten. "Adaptive Grazing Management in Semiarid Rangelands: An Outcome-Driven Focus." *Rangelands* 44 (2022): 111-118.

Dillon, Jasmine, and Megan Machmuller. *Regenerative Grazing, Carbon, and Climate*. Report prepared by Colorado State University, 2021. https://pastureproject.org/wp-content/uploads/2021/11/Regenerative-Grazing-and-Carbon.pdf.

DiTomaso, J.M., R.A. Masters, and V.F. Peterson. "Rangeland Invasive Plant Management." *Rangelands* 32 (2010): 43-47. doi.org/10.2111/rangelands-d-09-00007.1.

D'Odorico, Paolo, Gregory S. Okin, and Brandon T. Bestelmeyer. "A Synthetic Review of Feedbacks and Drivers of Shrub Encroachment in Arid Grasslands." *Ecohydrology* 5 (2011): 520-530.

Dubeux Jr., J.C.B., L.E. Sollenberger, N.B. Comerford, J.M. Scholberg, A.C. Ruggieri, J.M.B. Vendramini, S.M. Interrante, and K.M. Portier. "Management Intensity Affects Density Fractions of Soil Organic Matter from Grazed Bahiagrass Swards." *Soil Biology & Biochemistry* 38 (2006): 2705-2711.

Ehrhardt, F., J. Soussana, G. Bellocchi, P. Grace, R. McAuliffe, S. Recous, R. Sándor, et al. "Assessing Uncertainties in Crop and Pasture Ensemble Model Simulations of Productivity and N2O Emissions." *Global Change Biology* 24 (2018): e603-e616. doi:10.1111/gcb.13965.

Endress, B. A., C. G. Parks, B. J. Naylor, S. R. Radosevich, and M. Porter. "Grassland Response to Herbicides and Seeding of Native Grasses 6 Years Posttreatment." *Invasive Plant Science and Management* 5, no. 3 (2012): 311-316.

Environmental Protection Agency (EPA). "Understanding Global Warming Potentials." 2025a. https://www.epa.gov/ghgemissions/understanding-global-warming-potentials.

Environmental Protection Agency (EPA). "Greenhouse Gas Equivalencies Calculator." 2025b. https://www.epa.gov/energy/greenhouse-gas-equivalencies-calculator.

Euliss Jr., Ned H., R.A. Gleason, A. Olness, R.L. McDougal, H.R. Murkin, R.D. Robarts, R.A. Bourbonniere, and B.G. Warner. "North American Prairie Wetlands Are Important Nonforested Land-Based Carbon Storage Sites." *Science of the Total Environment* 361, no. 1-3 (2006): 179-188.

Fischer, J., and D. B. Lindenmayer. "Landscape Modification and Habitat Fragmentation: A Synthesis." *Global Ecology and Biogeography* 16, no. 3 (2007): 265-280. https://doi.org/10.1111/j.1466-8238.2007.00287.x.

Flechard, C.R., P. Ambus, U. Skiba, R.M. Rees, A. Hensen, A. van Amstel, A. van den P. Dasselaar, et al. "Effects of Climate and Management Intensity on Nitrous Oxide Emissions in Grassland Systems across Europe." *Agriculture, Ecosystems & Environment* 121 (2007): 135-152. doi:10.1016/j.agee.2006.12.024.

Fletcher, Brittany Lippy. "Nutrition and Management of Woody Plant Encroachment Utilizing Cattle and Goats." PhD diss. Oklahoma State University, 2024.

Fluet-Chouinard, Etienne, Benjamin D. Stocker, Zhen Zhang, Avni Malhotra, Joe R. Melton, Benjamin Poulter, Jed O. Kaplan, Kees Klein Goldewijk, Stefan Siebert, and Tatiana Minayeva. "Extensive Global Wetland Loss over the Past Three Centuries." *Nature* 614, no. 7947 (2023): 281-286.

Folke, C., S. Carpenter, B. Walker, M. Scheffer, T. Elmqvist, L. Gunderson, and C.S. Holling. "Regime Shifts, Resilience, and Biodiversity in Ecosystem Management." *Annual Review of Ecology, Evolution, and Systematics* 35 (2004): 557-581. https://doi.org/10.1146/annurev.ecolsys.35.021103.105711.

Follett, R.F., and D.A. Reed. "Soil Carbon Sequestration in Grazing Lands: Societal Benefits and Policy Implications." *Rangeland Ecology & Management* 63 (2010): 4-15. doi.org/10.2111/08-225.1.

Foster, Jamie L., Megan K. Clayton, Meagan M. Lesak, Kimberly McCuistion, and Trent Teinert. "Combining Management Techniques for Short-Term Reduction of Introduced Old World Bluestems in South Texas Rangelands." *Rangeland Ecology & Management* 99 (2025): 50-57.

Fuchs, Kathrin, Lutz Merbold, Nina Buchmann, Gianni Bellocchi, Marco Bindi, Lorenzo Brilli, Richard T. Conant, et al. "Evaluating the Potential of Legumes to Mitigate N2O Emissions from Permanent Grassland Using Process-Based Models." *Global Biogeochemical Cycles* 34, no. 12 (2020): e2020GB006561. https://doi.org/10.1029/2020GB006561.

Fuhlendorf, S.D., D.M. Engle, R.D. Elmore, R.F. Limb, and T.G. Bidwell. "Conservation of Pattern and Process: Developing an Alternative Paradigm of Rangeland Management." *Rangeland Ecology & Management* 65 (2011): 579-589.

Fynn, Richard W.S., David J. Augustine, and Samuel D. Fuhlendorf. "Managing Browsing and Grazing Ungulates." In *The Ecology of Browsing and Grazing II*, edited by Iain J. Gordon and Herbert H.T. Prins, 321-338. Springer, 2019. doi.org/10.1007/978-3-030-25865-8_14.

Fultz, Lisa M., Jennifer Moore-Kucera, et al. “Forest wildfire and grassland prescribed fire effects on soil biogeochemical processes and microbial communities: Two case studies in the semi-arid Southwest.” *Applied Soil Ecology*, 99: 118-128.

Gann, G.D., T. McDonald, B. Walder, J. Aronson, C.R. Nelson, J. Jonson, J.G. Hallett, et al. "International Principles and Standards for the Practice of Ecological Restoration." *Restoration Ecology* 27, no. S1 (2019): S1-S46. doi: 10.1111/rec.13035.

Garsia, A., A. Moinet, C. Vazquez, R.E. Creamer, and G.Y.K. Moinet. "The Challenge of Selecting an Appropriate Soil Organic Carbon Simulation Model: A Comprehensive Global Review and Validation Assessment." *Global Change Biology* (2023): 1-15. doi:10.1111/gcb.16896.

Geesing, Dieter, Peter Felker, Ralph L. Bingham. “Influence of mesquire (Prosopis glandulosa) on soil nitrogen and carbon development: implications for global carbon sequestration”. *J. Arid Environ*. 46: 157-180. https://doi.org/10.1006/jare.2000.0661.

Gill, R.A., and Ingrid C. Burke. "Ecosystem Consequences of Plant Life Form Changes at Three Sites in the Semiarid United States." *Oecologia* 121 (1999): 551-563. doi.org/10.1007/s004420050962.

Gilmanov, Tagir G., Tony J. Svejcar, Douglas A. Johnson, Raymond F. Angell, Nicanor Z. Saliendra, and Bruce K. Wylie. "Long-Term Dynamics of Production, Respiration, and Net CO2 Exchange in Two Sagebrush-Steppe Ecosystems." *Rangeland Ecology & Management* 59, no. 6 (2006): 585-599. doi: 10.2111/05-198R1.1.

Gleason, Robert A., Murray K. Laubhan, Ned H. Euliss Jr., Brian A. Tangen, and Kevin E. Kermes. *Ecosystem Services Derived from Wetland Conservation Practices in the United States Prairie Pothole Region with an Emphasis on the US Department of Agriculture Conservation Reserve and Wetlands Reserve Programs*. Reston, Virginia: US Geological Survey, 2008.

Gleason, R.A., B.A. Tangen, B.A. Browne, and N.H. Euliss Jr. "Greenhouse Gas Flux from Cropland and Restored Wetlands in the Prairie Pothole Region." *Soil Biology and Biochemistry* 41 (2009): 2501-2507.

Godde, C.M., I.J.M. de Boer, E. zu Ermgassen, M. Herrero, C.E. van Middelaar, A. Muller, E. Roos, C. Schader, P. Smith, H.H.E. van Zanten, and T. Garnett. "Soil Carbon Sequestration in Grazing Systems: Managing Expectations." *Climatic Change* 161 (2022): 385-391.

Gómez-Aparicio, L. "The Role of Plant Interactions in the Restoration of Degraded Ecosystems: A Meta-Analysis across Life-Forms and Ecosystems." *Journal of Ecology* 97, no. 6 (2009): 1202-1214.

Gomez-Casanovas, N., N.J. DeLucia, C.J. Bernacchi, E.H. Boughton, J.P. Sparks, S.D. Chamberlain, and E.H. DuLucia. "Grazing Alters Net Ecosystem C Fluxes and the Global Warming Potential of a Subtropical Pasture." *Ecological Applications* 28 (2018): 557-572.

Govers, G., R. Merckx, K. Van Oost, and B. Wesemael. *Managing Soil Organic Carbon for Global Benefits: A STAP Technical Report*. Washington, D.C.: Global Environment Facility, 2013. http://www.stapgef.org.

Goyette, Jean-Olivier, Stephane Savary, Marianne Blanchette, Alain N. Rousseau, Stephanie Pellerin, and Monique Poulin. "Setting Targets for Wetland Restoration to Mitigate Climate Change Effects on Watershed Hydrology." *Environmental Management* 71 (2023): 365-378.

Gravuer, Kelly, Sasha Gennet, and Heather L. Throop. "Organic Amendment Additions to Rangelands: A Meta-Analysis of Multiple Ecosystem Outcomes." *Global Change Biology* 25, no. 3 (2019): 1152-1170. https://doi.org/10.1111/gcb.14535.

Guo, L.B., and R.M. Gifford. "Soil Carbon Stocks and Land Use Change: A Meta Analysis." *Global Change Biology* 8 (2002): 345-360. doi:10.1046/j.1354-1013.2002.00486.x.

Harris, Lucas, Alan H. Taylor. “Previous burns and topography limit and reinforce fire severity in a large wildfire.” *Ecosphere*, 8. https://doi.org/10.1002/ecs2.2019.

Hasler, N., C.A. Williams, V.C. Denney, P.W. Ellis, S. Shrestha, D.E.T. Hart, N.H. Wolff, S. Yeo, T.W. Crowther, L.K. Werden, and S.C. Cook-Patton. "Accounting for Albedo Change to Identify Climate-Positive Tree Restoration." *Nature Communications* 15 (2024). doi.org/10.1038/s41467-024-46577-1.

Havstad, K.M., D.P.C. Peters, R. Skaggs, J. Brown, B. Bestelmeyer, E. Fredrickson, J. Herrick, and J. Wright. "Ecological Services to and from Rangelands of the United States." *Ecological Economics* 64 (2007): 261-268. https://doi.org/10.1016/j.ecolecon.2007.08.005.

Hayes, D.J., D.P. Turner, G. Stinson, A.D. McGuire, Y. Wei, T.O. West, L.S. Heath, et al. "Reconciling Estimates of the Contemporary North American Carbon Balance among Terrestrial Biosphere Models, Atmospheric Inversions, and a New Approach for Estimating Net Ecosystem Exchange from Inventory-Based Data." *Global Change Biology* 18 (2012): 1282-1299. doi.org/10.1111/j.1365-2486.

Heinemeyer, A., M. Bahn, and W.L. Kutsch, eds. *Soil Carbon Dynamics: An Integrated Methodology*. Cambridge University Press, 2010.

Heitschmidt, R.K., R.D. Schultz, and C.J. Scifres. "Herbaceous Biomass Dynamics and Net Primary Production Following Chemical Control of Honey Mesquite." *Rangeland Ecology & Management* 39 (1986): 67-71.

Henderson, Benjamin B., Pierre J. Gerber, Tom E. Hilinski, Alessandra Falcucci, Dennis S. Ojima, Mirella Salvatore, and Richard T. Conant. "Greenhouse Gas Mitigation Potential of the World's Grazing Lands: Modeling Soil Carbon and Nitrogen Fluxes of Mitigation Practices." *Agriculture, Ecosystems & Environment* 207 (2015): 91-100. https://doi.org/10.1016/j.agee.2015.03.029.

Henry, H.A.L., N.R. Chiariello, P.M. Vitousek, H.A. Mooney, C.B. Field. “Interactive effects of fire, elevated carbon dioxide, nitrogen deposition, and precipitation on a California annual grassland.” *Ecosystems* 9 (2006): 1066-1075.

Hernandez-Blanco, M., R. Costanza, H. Chen, D. deGroot, et al. "Ecosystem Health, Ecosystem Services, and the Well-Being of Humans and the Rest of Nature." *Global Change Biology* 28 (2022): 5027-5040. https://doi.org/10.1111/gcb.16281.

Hernadez, Maria E., William J. Mitsch. “Denitrification in created riverine wetlands: Influence of hydrology and season.” *Ecological Engineering* 30 (2006): 78-88.

Herrick, J. E., K.M. Havstad, and A. Rango. "Remediation Research in the Jornada Basin: Past and Future." In *Structure and Function of a Chihuahuan Desert Ecosystem: The Jornada Basin Long-Term Ecological Research Site*, edited by K.M. Havstad, L.F. Huenneke, and W.H. Schlesinger, 278-304. 2006.

Hewins, D. B., M. P. Lyseng, D. F. Schoderbek, M. Alexander, W. D. Willms, C. N. Carlyle, and E. W. Bork. "Grazing and Climate Effects on Soil Organic Carbon Concentration and Particle-Size Association in Northern Grasslands." *Scientific Reports* 8, no. 1 (2018): 1336.

Hibbard, K. A., S. A. Archer, D. S. Schimel, and D. V. Valentine. "Biogeochemical Changes Accompanying Woody Plant Encroachment in a Subtropical Savanna." *Ecology* 82 (2001): 1999-2011.

Hibbard, K.A., D.S. Schimel, S. Archer, D.S. Ojima, and W. Parton. "Grassland to Woodland Transitions: Integrating Changes in Landscape Structure and Biogeochemistry." *Ecological Applications* 13 (2003): 911-926.

Hillenbrand, M., R. Thompson, F. Wang, S. Apfelbaum, and R. Teague. "Impacts of Holistic Planned Grazing with Bison Compared to Continuous Grazing with Cattle in South Dakota Shortgrass Prairie." *Agriculture, Ecosystems & Environment* 279 (2019): 156-168.

Hobbs, N. T., D. S. Schimel, C. E. Owensby, and D. S. Ojima. "Fire and Grazing in the Tallgrass Prairie: Contingent Effects on Nitrogen Budgets." *Ecology* 72, no. 4 (1991): 1374-1382. https://doi.org/10.2307/1941109.

Holechek, Jerry L., and Dee Galt. "Grazing Intensity Guidelines." *Rangelands* 22 (2000). USDA Range Technical Note No. NM-113. https://efotg.sc.egov.usda.gov/references/public/NM/RangeTechNote113.pdf.

Hondula, Kelly L., C. Nathan Jones, Margaret A. Palmer. “Effects of seasonal inundation on methane fluxes from forested wetlands.” Environmental Research Letters (2021). DOI: 10.1088/1748-9326/ac1193

Horton, J.L., and S.C. Hart. "Hydraulic Lift: A Potentially Important Ecosystem Process." *Trends in Ecology & Evolution* 13, no. 6 (1998): 232-235.

House, J. I., S. Archer, D. D. Breshears, R. J. Scholes, and NCEAS Tree–Grass Interactions Participants. "Conundrums in Mixed Woody–Herbaceous Plant Systems." *Journal of Biogeography* 30, no. 11 (2003): 1763-1777. https://doi.org/10.1046/j.1365-2699.2003.00873.x.

Hui, D., Q. Deng, H. Tiam, and Y. Luo. "Global Climate Change and Greenhouse Gas Emissions in Terrestrial Ecosystems." In *Handbook of Climate Change Mitigation and Adaptation*, 3rd ed., edited by M. Lackner, B. Sajjadi, and W-Y. Chen, 23-76. Cham: Springer International Publishing, 2022.

Hunt, E. Raymond Jr., Robert D. Kelly, William K. Smith, Jace T. Fahnestock, Jeffrey M. Welker, and William A. Reiners. "Estimation of Carbon Sequestration by Combining Remote Sensing and Net Ecosystem Exchange Data for Northern Mixed-Grass Prairie and Sagebrush-Steppe Ecosystems." *Environmental Management* 33 (2004): 432-441. doi.org/10.1007/s00267-003-9151-0.

Hursh, A., A. Ballantyne, L. Cooper, M. Maneta, J. Kimball, and J. Watts. "The Sensitivity of Soil Respiration to Soil Temperature, Moisture, and Carbon Supply at the Global Scale." *Global Change Biology* 23, no. 5 (2017): 2090-2103.

Huston, M.A., and G. Marland. "Carbon Management and Biodiversity." *Journal of Environmental Management* 67 (2003): 77-86. doi.org/10.1016/S0301-4797(02)00190-1.

Ingram, L.J., G.E. Schuman, T.B. Parkin, and M. Mortenson. "Trace Gas Fluxes from a Northern Mixed-Grass Prairie Interseeded with Alfalfa." *Plant and Soil* 386 (2015): 285-301. doi:10.1007/s11104-014-2264-9.

Ingram, L. J., P. D. Stahl, G. E. Schuman, J. S. Buyer, G. F. Vance, G. K. Ganjegunte, and J. D. Derner. "Grazing Impacts on Soil Carbon and Microbial Communities in a Mixed-Grass Ecosystem." *Soil Science Society of America Journal* 72, no. 4 (2008): 939-948.

IPCC. "Summary for Policymakers." In *Climate Change 2022: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*, edited by H.-O. Pörtner, D.C. Roberts, M. Tignor, E.S. Poloczanska, K. Mintenbeck, A. Alegría, M. Craig, et al., 3-33. Cambridge, UK: Cambridge University Press, 2022. doi:10.1017/9781009325844.001.

Jackson, Robert B., Jay L. Banner, Esteban G. Jobbagy, William T. Pockman, and Diana H. Wall. "Ecosystem Carbon Loss with Woody Plant Invasion of Grasslands." *Nature* 418 (2002): 623-626. https://doi.org/10.1038/nature00610.

Jackson, R.B., K. Lajtha, S.E. Crow, G. Hugelius, M.G. Kramer, and G. Pineiro. "The Ecology of Soil Carbon: Pools, Vulnerabilities, and Biotic and Abiotic Controls." *Annual Review of Ecology, Evolution and Systematics* 48 (2017): 419-445.

Janzen, H.H. "Soil Carbon: A Measure of Ecosystem Response in a Changing World." *Canadian Journal of Soil Science* 85 (2005). https://doi.org/10.4141/S04-081.

Jariel, D.M. Jr., R.J. Ansley, B.A. Kramp, D.L. Jones. “Soil chemical responses to fire seasonality and frequency in a Texas grassland.” The Prairie Naturalist (2014): 98-109.

Jellinek, S., K.M. Parris, D.A. Driscoll, and P.D. Dwyer. "Are Incentive Programs Working? Landowner Attitudes to Ecological Restoration of Agricultural Landscapes." *Journal of Environmental Management* 127 (2013): 69-76. doi: 10.1016/j.jenvman.2013.04.034.

Jenny, Hans. *Factors of Soil Formation: A System of Quantitative Pedology*. Courier Corporation, 1994.

Jisha, K.C., and Jos T. Puthur. "Ecological Importance of Wetland Systems." In *Wetlands Conservation: Current Challenges and Future Strategies*, 40-54. 2021.

Johnson, Jane M.F., Sharon L. Weyers, David W. Archer, and Nancy W. Barbour. "Nitrous Oxide, Methane Emission, and Yield-Scaled Emission from Organically and Conventionally Managed Systems." *Soil Science Society of America Journal* 76, no. 4 (2012): 1347-1357.

Johnston, A.E., P.R. Poulton, and K. Coleman. "Soil Organic Matter: Its Importance in Sustainable Agriculture and Carbon Dioxide Fluxes." In *Advances in Agronomy*, Vol. 101, edited by D.L. Sparks. 2009. doi: 10.1016/S0065-2113(08)00801-8.

Johnston, Carol A. "Beaver Pond Effects on Carbon Storage in Soils." *Geoderma* 213 (2014): 371-378.

Jones, R., J. C. Chambers, D. W. Johnson, R. R. Blank, and D. I. Board. "Effect of Repeated Burning on Plant and Soil Carbon and Nitrogen in Cheatgrass (Bromus tectorum) Dominated Ecosystems." *Plant and Soil* 386, no. 1-2 (2015): 47-64. https://doi.org/10.1007/s11104-014-2242-2.

Jordon, Matthew W., Jean-Charles Buffet, Jennifer A. J. Dungait, Marcelo V. Galdos, Tara Garnett, Michael R. F. Lee, John Lynch, et al. "A Restatement of the Natural Science Evidence Base Concerning Grassland Management, Grazing Livestock and Soil Carbon Storage." *Proceedings of the Royal Society B: Biological Sciences* 291, no. 2015 (2024): 20232669. https://doi.org/10.1098/rspb.2023.2669.

Katz, S., T. Maxwell, M. A. de Graaff, and M. J. Germino. "Invasion of Perennial Sagebrush Steppe by Shallow-Rooted Exotic Cheatgrass Reduces Stable Forms of Soil Carbon in a Warmer but Not Cooler Ecoregion." *Environmental Research Communications* 7, no. 3 (2025): 031001.

Kell, D.B. "Large-Scale Sequestration of Atmospheric Carbon via Plant Roots in Natural and Agricultural Ecosystems: Way and How." *Philosophical Transactions of the Royal Society B* 367 (2012). https://doi.org/10.1098/rstb.2011.0244.

Kimball, Sarah, Megan Lulow, Quinn Sorenson, Kathleen Balazs, Yi-Chin Fang, Steven J. Davis, Michael O'Connell, and Travis E. Huxman. "Cost-Effective Ecological Restoration." *Restoration Ecology* 23, no. 6 (2015): 800-810. https://doi.org/10.1111/rec.12261.

Kleppel, Gary S., and Douglas A. Frank. "Structure and Functioning of Wild and Agricultural Grazing Ecosystems: A Comparative Review." *Frontiers in Sustainable Food Systems* 6 (2022). doi.org/10.3389/fsufs.2022.945514.

Kluender, C. R., M. J. Germino, B. E. Lazarus, and T. Matthews. "Patchy Response of Cheatgrass and Nontarget Vegetation to Indaziflam and Imazapic Applied After Wildfire in Sagebrush Steppe." *Rangeland Ecology & Management* 98 (2025): 432-440.

Knapp, A. K., S. L. Conard, and J. M. Blair. "Determinants of Soil CO2 Flux from a Sub-Humid Grassland: Effect of Fire and Fire History." *Ecological Applications* 8, no. 3 (1998): 760-770. https://doi.org/10.1890/1051-0761(1998)008[0760:DOSCFF]2.0.CO;2.

Knapp, A.K., John M. Briggs, et al. “Shrub encroachment in North American grasslands: shifts in growth form dominance rapidly alters control of ecosystem carbon inputs.” *Global Change Biology* 14: 615-623. https://doi.org/10.1111/j.1365-2486.2007.01512.x

Knapp, C.N., and M.E. Fernandez-Gimenez. "Knowledge in Practice: Documenting Rancher Local Knowledge in Northwest Colorado." *Rangeland Ecology and Management* 62 (2009): 500-509.

Knight, R. "Beef -- Sector at a Glance." USDA Economic Research Service, August 30, 2023. https://www.ers.usda.gov/topics/animal-products/cattle-beef/sector-at-a-glance/.

Kochy, M., R. Hiederer, and A. Freibauer. "Global Distribution of Soil Organic Carbon -- Part 1: Masses and Frequency Distributions of SOC Stocks for the Tropics, Permafrost Regions, Wetlands and the World." *SOIL* 1 (2015): 351-365.

Korfmacher, J. L., J. C. Chambers, R. J. Tausch, B. A. Roundy, S. E. Meyer, and S. Kitchen. "Technical Note: A Technique for Conducting Small-Plot Burn Treatments." *Journal of Range Management* 56, no. 3 (2003): 251. https://doi.org/10.2307/4003814.

Koteen, L. E., D. D. Baldocchi, and J. Harte. "Invasion of Non-Native Grasses Causes a Drop in Soil Carbon Storage in California Grasslands." *Environmental Research Letters* 6, no. 4 (2011): 044001.

Kramer, R.J., H. He, B.J. Soden, L. Oreopoulos, G. Myhre, P.M. Forster, and C.J. Smith. "Observational Evidence of Increasing Global Radiative Forcing." *Geophysical Research Letters* 48 (2021). doi.org/10.1029/2020GL091585.

Kray, Julie A., Dana M. Blumenthal, Mitchell B. Stephenson, Lauren M. Porensky, David J. Augustine, Seth J. Romero, Matt C. Mortenson, and Justin D. Derner. "Using Targeted Grazing to Close the Phenological Niche Exploited by Invasive Annual Bromes." *Rangeland Ecology & Management* 98 (2025): 441-453.

Lal, R. "Carbon Sequestration." *Philosophical Transactions of the Royal Society B: Biological Sciences* 363 (2008): 815-830.

LANDFIRE. "Existing Vegetation Type Layer, LANDFIRE 2.0.0." U.S. Department of the Interior, Geological Survey, and U.S. Department of Agriculture, 2016. http://www.landfire/viewer.

Lang, M.W., J.C. Ingebritsen, and R.K. Griffin. *Status and Trends of Wetlands in the Conterminous United States 2009 to 2019*. Washington, D.C.: U.S. Department of the Interior; Fish and Wildlife Service, 2024.

Li, W., X. Zuo, J.M.H. Knops. “Different fire frequency impacts over 27 years on vegetation sucession in an infertile old-field grassland.” *Rangeland Ecology & Mangement* 66 (2013): 267-273.

Li, W., P. Ciais, B. Guenet, S. Peng, J. Chang, V. Chaplot, S. Khudyaev, A. Peregon, S. Piao, Y. Wang, and C. Yue. "Temporal Response of Soil Organic Carbon after Grassland-Related Land-Use Change." *Global Change Biology* 24 (2018): 4731-4746. doi:10.1111/gcb.14328.

Li, Zheng, Jay P. Angerer, and X. Ben Wu. "Temporal Patterns of Large Wildfires and Their Burn Severity in Rangelands of Western United States." *Geophysical Research Letters* 48 (2021). doi.org/10.1029/2020GL091636.

Li, Zheng, Jay P. Angerer, and X. Ben Wu. "The Impacts of Wildfires of Different Burn Severities on Vegetation Structure across the Western United States Rangelands." *Science of the Total Environment* 845 (2022). doi.org/10.1016/j.scitotenv.2022.157214.

Liao, J.D., T.W. Boutton, and J.D. Jastrow. "Storage and Dynamics of Carbon and Nitrogen in Soil Physical Fractions Following Woody Plant Invasion of Grassland." *Soil Biology and Biochemistry* 38 (2006): 3184-3196. doi.org/10.1016/j.soilbio.2006.04.003.

Limb, R. F., D. M. Engle, A. L. Alford, and E. C. Hellgren. "Tallgrass Prairie Plant Community Dynamics Along a Canopy Cover Gradient of Eastern Redcedar (Juniperus virginiana L.)." *Rangeland Ecology & Management* 63, no. 6 (2010): 638-644. https://doi.org/10.2111/REM-D-09-00056.1.

Limb, R. F., S. D. Fuhlendorf, D. M. Engle, and R. F. Miller. "Synthesis Paper: Assessment of Research on Rangeland Fire as a Management Practice." *Rangeland Ecology & Management* 69, no. 6 (2016): 415-422. https://doi.org/10.1016/j.rama.2016.07.013.

Limpens, J., F. Berendse, C. Blodau, J. G. Canadell, C. Freeman, J. Holden, N. Roulet, H. Rydin, and G. Schaepman-Strub. "Peatlands and the Carbon Cycle: From Local Processes to Global Implications -- A Synthesis." *Biogeosciences* 5 (2008): 1475-1491. https://doi.org/10.5194/bg-5-1475-2008.

Ma, L., J.D. Derner, R.D. Harmel, J. Tatarko, A.D. Moore, C.A. Rotz, D.J. Augustine, et al. "Application of Grazing Land Models in Ecosystem Management: Current Status and Next Frontiers." In *Advances in Agronomy*, 173-215. Elsevier, 2019. doi:10.1016/bs.agron.2019.07.003.

MacNeil, M. D., M. R. Haferkamp, L. T. Vermeire, and J. M. Muscha. "Prescribed Fire and Grazing Effects on Carbon Dynamics in a Northern Mixed-Grass Prairie." *Agriculture, Ecosystems & Environment* 127, no. 1-2 (2008): 66-72. https://doi.org/10.1016/j.agee.2008.02.015.

Maestre, Fernardo T., Matthew A. Bowker, et al. “Shrub encroachment can reverse desertification in semi-arid Mediterranean grasslands.” *Ecology Letters* 12 (2009). Doi.org/10.1111/j.1461-0248.2009.01352.x

Maher, A.T., N.E. Ashwell, K.A. Maczko, D.T. Taylor, J.A. Tanaka, and M.C. Reeves. "An Economic Valuation of Federal and Private Grazing Land Ecosystem Services Supported by Beef Cattle Ranching in the United States." *Translational Animal Science* 5 (2021): 1-15. https://doi.org/10.1093/tas/txab054.

Mahood, Adam L., Rachel O. Jones, David I. Board, Jennifer K. Balch, and Jeanne C. Chambers. "Interannual Climate Variability Mediates Changes in Carbon and Nitrogen Pools Caused by Annual Grass Invasion in a Semiarid Shrubland." *Global Change Biology* 28, no. 1 (2022): 267-284.

Manning, G.C., S.G. Baer, and J.M. Blair. "Effects of Grazing and Fire Frequency on Floristic Quality and Its Relationship to Indicators of Soil Quality in Tallgrass Prairie." *Environmental Management* 60 (2017): 1062-1075. doi.org/10.1007/s00267-017-0942-0.

Martinez-Martinez, Edwin, A. Pouyan Nejadhashemi, Sean A. Woznicki, and Bradley J. Love. "Modeling the Hydrological Significance of Wetland Restoration Scenarios." *Journal of Environmental Management* 133 (2014): 121-134.

Marton, John M., Rinku Roy Chowdhury, and Christopher B. Craft. "A Comparison of the Spatial Variability of Denitrification and Related Soil Properties in Restored and Natural Depression Wetlands in Indiana, USA." *International Journal of Biodiversity Science, Ecosystem Services & Management* 11, no. 1 (2015): 36-45. doi: 10.1080/21513732.2014.950981.

Masters, R. A., S. J. Nissen, R. E. Gaussoin, D. D. Beran, and R. N. Stougaard. "Imidazolinone Herbicides Improve Restoration of Great Plains Grasslands." *Weed Technology* 10, no. 2 (1996): 392-403.

Matzek, Virginia, David Lewis, Anthony O'Geen, Michael Lennox, Sean D. Hogan, Shane T. Feirer, Valerie Eviner, Kenneth W. Tate. "Increases in soil and woody biomass carbon stocks as a result of rangeland riparian restoration." *Carbon Balance and Management* 15 (2020).

Mauritz, M., and D.A. Lipson. "Plant Community Composition Alters Moisture and Temperature Sensitivity of Soil Respiration in Semi-Arid Shrublands." *Oecologia* 197 (2021): 1003-1015. doi.org/10.1007/s00442-021-04961-4.

Maxwell, T. M., H. E. Quicke, S. J. Price, and M. J. Germino. "Annual Grass Invasions and Wildfire Deplete Ecosystem Carbon Storage by > 50% to Resistant Base Levels." *Communications Earth & Environment* 5, no. 1 (2024): 669.

McArthur, E. Durant, and Stanley G. Kitchen. "Shrubland ecosystems: Importance, distinguishing characteristics, and dynamics. In: Sosebee, Ronald E., et al., comps. Proceedings: Shrubland dynamics – fire and water; 2004 August 10-12; Lubbock, TX. Proceedings RMS-P-47. Fort Collins, CO: U.S. Department of Agriculture, Forest Service (2007). https://research.fs.usda.gov/treesearch/28353

McClaran, Mitchel P., Jennifer Moore-Kucera, Dean A. Martens, Joost van Haren, and Stuart E. Marsh. "Soil Carbon and Nitrogen in Relation to Shrub Size and Death in a Semi-Arid Grassland." *Geoderma* 145 (2008): 60-68. doi.org/10.1016/j.geoderma.2008.02.006.

McClure, Glennis A., and Jim A. Jansen. *Nebraska Farm Custom Rates, 2024*. 2024. https://extensionpubs.unl.edu/publication/ec823/2024/pdf/view/ec823-2024.pdf.

McCord, S.E., and D.S. Pilliod. "Adaptive Monitoring in Support of Adaptive Management in Rangelands." *Rangelands* 44 (2022): 1-7. doi.org/10.1016/j.rala.2021.07003.

McCulley, R.L., S.R. Archer, T.W. Boutton, F.M. Hons, D.A. Zuberer. "Soil respiration and nutrient cycling in wooded communities developing in grassland." *Ecology* 85: 2804-2817 (2004).

McGregor, S., J.P.G.M. Cromsigt, M. te Beest, J. Chen, D.P. Roy, H.-J. Hawkins, and G.I.H. Kerley. "Grassland Albedo as a Nature-Based Climate Prospect: The Role of Growth Form and Grazing." *Environmental Research Letters* 19 (2024). doi.org/10.1088/1748-9326/ad8765.

McKinley, Duncan C., Mark D. Norris, John M. Blair, Loretta C. Johnson. "Altered ecosystem processes as a consequence of Juniperus virginiana L. encroachment into North American tallgrass prairie." In: Van Auken, O.W. (eds) Western North American Juniperus Communities. Ecological Studies, vol 196. (2008). Doi.org/10.1007/978-387-34003-6_9

McLellan, E.L., K.G. Cassman, A.J. Eagle, P.B. Woodbury, S. Sela, C. Tonitto, R.D. Marjerison, and H.M. van Es. "The Nitrogen Balancing Act: Tracking the Environmental Performance of Food Production." *BioScience* 68 (2018): 194-203. doi:10.1093/biosci/bix164.

Means, Mary M., Changwoo Ahn, Alicia R. Korol, and Lisa D. Williams. "Carbon Storage Potential by Four Macrophytes as Affected by Planting Diversity in a Created Wetland." *Journal of Environmental Management* 165 (2016): 133-139.

Mehre, J., K. Schneider, S. Jayasundara, A. Gillespire, and C. Wagner-Riddle. "Adaptive Multi-Paddock Grazing Increases Soil Carbon Stocks and Decreases the Carbon Footprint of Beef Production in Ontario, Canada." *Journal of Environmental Management* 371 (2024). https://doi.org/10.1016/j.jenvman.2024.123255.

Melton, J.R., R. Wania, E.L. Hodson, B. Poulter, B. Ringeval, R. Spahni, T. Bohn, et al. "Present State of Global Wetland Extent and Wetland Methane Modelling: Conclusions from a Model Intercomparison Project (WETCHIMP)." *Biogeosciences Discussions* 9 (2012): 11577-11654.

Miller, R.F., S.T. Knick, D.A. Pyke, C.W. Meinke, S.E. Hanser, M.J. Wisdom, and A.L. Hild. "Characteristics of Sagebrush Habitats and Limitations to Long-Term Conservation." In *Greater Sage-Grouse: Ecology and Conservation of Landscape Species and Its Habitats*, edited by S.T. Knick and J.W. Connelly, 146-184. Berkeley, CA: University of California Press, 2011.

Mitsch, William J., and James G. Gosselink. "The Value of Wetlands: Importance of Scale and Landscape Setting." *Ecological Economics* 35, no. 1 (2000): 25-33.

Miwa, C. T., and R. J. Reuter. "Persistence of Western Juniper (Juniperus occidentalis) Resource Islands Following Canopy Removal." *Northwest Science* 84, no. 4 (2010): 361-368. https://doi.org/10.3955/046.084.0405.

Mohr, Kylie. "Bureau of Land Management: There Are Millions of Acres of 'Failing' Rangelands, Data Shows." *High Country News*, 2022. hcn.org/issues/54-5/.

Monaco, T.A., T.A. Jones, and T.L. Thurow. "Identifying Rangeland Restoration Targets: An Appraisal of Challenges and Opportunities." *Rangeland Ecology & Management* 65 (2012): 599-605. doi.org/10.2111/REM-D-12-00012.1.

Moreno-Mateos, David, Mary E. Power, Francisco A. Comin, and Roxana Yockteng. "Structural and Functional Loss in Restored Wetland Ecosystems." *PLoS Biology* (2012). Doi.org/10.1371/journal.pbio.1001247.

Morford, S.L., B.W. Allred, D. Twidwell, M.O. Jones, J.D. Maestas, C.P. Roberts, D.E. Naugle. “Herbaceous production lost to tree encroachment in United States rangelands”. *J. Applied Ecology*. Doi.org/10.1101/2021.04.02.438282.

Morgan, J. A., W. Parton, J. D. Derner, T. G. Gilmanov, and D. P. Smith. "Importance of Early Season Conditions and Grazing on Carbon Dioxide Fluxes in Colorado Shortgrass Steppe." *Rangeland Ecology & Management* 69, no. 5 (2016): 342-350.

Morris, Craig D. "How Biodiversity-Friendly Is Regenerative Grazing?" *Frontiers in Ecology and Evolution* 9 (2021): 816374.

Mortenson, M.C., G.E. Schuman, L.J. Ingram, V. Nayigibugu, and B.W. Hess. "Forage Production and Quality of a Mixed-Grass Rangeland Interseeded with Medicago sativa ssp. Falcata." 2005.

Moser, Kurt, Changwoo Ahn, and Gregory Noe. "Characterization of Microtopography and Its Influence on Vegetation Patterns in Created Wetlands." *Wetlands* 27, no. 4 (2007): 1081-1097.

Moser, Kurt F., Changwoo Ahn, and Gregory B. Noe. "The Influence of Microtopography on Soil Nutrients in Created Mitigation Wetlands." *Restoration Ecology* 17, no. 5 (2009): 641-651.

Mosier, S., S. Apfelbaum, P. Byck, F. Calderon, R. Teague, R. Thompson, and M.F. Cotrufo. "Adaptive Multi-Paddock Grazing Enhances Soil Carbon and Nitrogen Stocks and Stabilization through Mineral Association in Southeastern Grazing Lands." *Journal of Environmental Management* 288 (2021). https://doi.org/10.1016/j.jenvman.2021.112240.

Mount, Dallas. "Recognizing Overgrazing -- How Do I Know?" University of Wyoming, Barn Backyard, 2009. https://www.uwyo.edu/barnbackyard/_files/documents/magazine/2009/summer/recognizing-overgrazing-summer-bb-2009.pdf.

Mumford, C., J. Mangold, J. Winnie Jr., K.A. Cutting, C. Zabinski, and L.J. Rew. "Response of Cheatgrass and Other Vegetation to Proactive Management Using Two Consecutive Fall Herbicide Applications in High-Elevation Sagebrush Steppe." *Rangeland Ecology & Management* 101 (2025): 43-53.

Mushet, David M., Ned H. Euliss Jr., Donald O. Rosenberry, James W. LaBaugh, Sheel Bansal, Zeno F. Levy, Owen P. McKenna, Kyle McLean, Christopher T. Mills, and Brian P. Neff. *Lessons Learned from Wetlands Research at the Cottonwood Lake Study Area, Stutsman County, North Dakota, 1967–2021*. US Geological Survey, 2022.

Nagy, R. Chelsea, Emily J. Fusco, Jennifer K. Balch, John T. Finn, Adam Mahood, Jenica M. Allen, and Bethany A. Bradley. "A Synthesis of the Effects of Cheatgrass Invasion on US Great Basin Carbon Storage." *Journal of Applied Ecology* 58, no. 2 (2021): 327-337.

Nahlik, A.M., and M.S. Fennessy. "Carbon Storage in US Wetlands." *Nature Communications* 7 (2016): 13835.

National Research Council (NRC). *Rangeland Health: New Methods to Classify, Inventory, and Monitor Rangelands*. Washington, D.C.: National Academy Press, 1994.

North American Wetlands Management Plan (NAWM). "Beaver-Related Wetland and Stream Restoration, Supporting Resources." 2024. https://www.nawm.org/pdf_lib/beaver_supporting_1.pdf.

Natural Resources Conservation Service (NRCS). "Chapter 5: Management of Grazing Lands." *National Range and Pasture Handbook*. U.S. Department of Agriculture, Natural Resources Conservation Service, 2022. https://www.nrcs.usda.gov/sites/default/files/2022-09/Chapter%205%20-%20Management%20of%20Grazing%20Lands.pdf.

Natural Resources Conservation Service (NRCS). *Grazing Management and Soil Health: Keys to Better Soil, Plant, Animal and Financial Health*. 2017. https://www.nrcs.usda.gov/sites/default/files/2022-09/Grazing%20Management_SoilHealth_0.pdf.

Natural Resources Conservation Service (NRCS). "Ecological Site Descriptions." U.S. Department of Agriculture, Natural Resources Conservation Service. n.d. https://www.nrcs.usda.gov/getting-assistance/technical-assistance/ecological-sciences/ecological-site-descriptions.

Norström, A.V., C. Cvitanovic, M.F. Löf, et al. "Principles for Knowledge Co-Production in Sustainability Research." *Nature Sustainability* 3 (2020): 182-190. https://doi.org/10.1038/s41893-019-0448-2.

Norton, Urszula, A.R. Mosier, J.A. Morgan, J.D. Derner, L.J. Ingram, and P.D. Stahl. "Moisture Pulses, Trace Gas Emissions and Soil C and N in Cheatgrass and Native Grass-Dominated Sagebrush-Steppe in Wyoming, USA." *Soil Biology and Biochemistry* 40, no. 6 (2008): 1421-1431. doi: 10.1016/j.soilbio.2007.12.021.

Ogle, D., and B. Brazee. "Estimating Initial Stocking Rates." *USDA-NRCS Technical Note* 3 (2009): 519.

Ojima, D.S., D.S. Schimel, W.J. Parton, and C.E. Owensby. "Long-Term and Short-Term Effects of Fire on Nitrogen Cycling in Tallgrass Prairie." *Biogeochemistry* 24 (1994): 67-84.

Pannu, Manmeet W., Kelley A. Meinhardt, Anthony Bertagnolli, Steven C. Fransen, David A. Stahl, and Stuart E. Strand. "Nitrous Oxide Emissions Associated with Ammonia-Oxidizing Bacteria Abundance in Fields of Switchgrass with and without Intercropped Alfalfa." *Environmental Microbiology Reports* 11, no. 5 (2019): 727-735.

Papanastasis, V.P. "Restoration of Degraded Grazing Lands through Grazing Management: Can It Work?" *Restoration Ecology* 17 (2009): 441-445. doi.org/10.1111/j.1526-100X.2009.00567.x.

Pearce, Casey, Laura Lautz, Christa Kelleher, and Julianne Davis. "Impact of Beaver Dam Analogues on Hydrology in a Semi-Arid Floodplain." *Hydrological Processes* 35 (2021). doi.org/10.1002/hyp.14275.

Pedrini, S., and K. W. Dixon. "International Principles and Standards for Native Seeds in Ecological Restoration." *Restoration Ecology* 28 (2020): S286-S303. doi: 10.1111/rec.13155.

Pellant, M. L. *Interpreting Indicators of Rangeland Health: Version 4*. US Department of the Interior, Bureau of Land Management, National Science and Technology Center, Division of Science Integration, Branch of Publishing Services, 2005.

Perkins, Steven R., and Kirk C. McDaniel. "Infiltration and Sediment Rates Following Creosotebush Control with Tebuthiuron." *Rangeland Ecology & Management* 58 (2005): 605-613. doi.org/10.2111/05-048R1.1.

Pfeifer-Meister, Laurel, Bitty A. Roy, Bart R. Johnson, Jeff Krueger, and Scott D. Bridgham. "Dominance of Native Grasses Leads to Community Convergence in Wetland Restoration." *Plant Ecology* 213 (2012): 637-647.

Phillips-Mao, Laura, et al. "Model-Based Scenario Planning to Develop Climate Change Adaptation Strategies for Rare Plant Populations in Grassland Reserves." *Biological Conservation* 193 (2016): 103-114.

Pierce, Nathan A., Steven R. Archer, Brandon T. Bestelmeyer, and Darren K. James. "Grass-Shrub Competition in Arid Lands: An Overlooked Driver in Grassland-Shrubland State Transition." *Ecosystems* 22 (2019): 619-628.

Plastina, Alejandro, Ann Johanns, and Kwanele Magwaba. *2024 Iowa Farm Custom Rate Survey*. Iowa State University, 2024. https://store.extension.iastate.edu/fm1698-pdf.

Poeplau, C., A. Don, L. Vesterdal, J. Leifeld, B. Van Wesemael, J. Schumacher, and A. Gensior. "Temporal Dynamics of Soil Organic Carbon after Land-Use Change in the Temperate Zone -- Carbon Response Functions as a Model Approach." *Global Change Biology* 17 (2011): 2415-2427. doi:10.1111/j.1365-2486.2011.02408.x.

Potts, D. L., K. N. Suding, G. C. Winston, A. V. Rocha, and M. L. Goulden. "Ecological Effects of Experimental Drought and Prescribed Fire in a Southern California Coastal Grassland." *Journal of Arid Environments* 81 (2012): 59-66. https://doi.org/10.1016/j.jaridenv.2012.01.007.

Prater, M. R., and E. H. DeLucia. "Non-Native Grasses Alter Evapotranspiration and Energy Balance in Great Basin Sagebrush Communities." *Agricultural and Forest Meteorology* 139, no. 1-2 (2006): 154-163. https://doi.org/10.1016/j.agrformet.2006.08.014.

Prokopy, L.S., K. Floress, J.G. Arbuckle, S.P. Church, F.R. Eanes, Y. Gao, B.M. Gramig, P. Ranjan, and A.S. Singh. "Adoption of Agricultural Conservation Practices in the United States: Evidence from 35 Years of Quantitative Literature." *Journal of Soil and Water Conservation* 74, no. 5 (2019): 520-534. https://doi.org/10.2489/jswc.74.5.520.

Ratcliff, F., D.R. Rao, S.J. Barry, S. Dewees, L. Macaulay, R. Peterson, M.A. Moritz, and L.C. Forero. "Cattle Grazing Reduces Fuel Load and Leads to More Manageable Fire Behavior." *U.S. Agriculture and Natural Resources* 73 (2022): 2-3. doi.org/10.3733/ca.2022a0011.

Rau, B.M., D.W. Johnson, J.C. Chambers, R.R. Blank, A. Lucchesi. “Estimating root biomass and distribution after fire in a Great Basin woodland using cores and pits.” Western North American Naturalist 69 (2009): 459-468.

Rau, B. M., D. W. Johnson, R. R. Blank, A. Lucchesi, T. G. Caldwell, and E. W. Schupp. "Transition from Sagebrush Steppe to Annual Grass (Bromus tectorum): Influence on Belowground Carbon and Nitrogen." *Rangeland Ecology & Management* 64, no. 2 (2011): 139-147.

Reed, C.C., A.A. Berhe, K.C. Moreland, J. Wilcox, and B.W. Sullivan. "Restoring Function: Positive Responses of Carbon and Nitrogen to 20 Years of Hydrologic Restoration in Montane Meadows." *Ecological Applications* 32, no. 7 (2022): e2677. doi: 10.1002/eap.2677.

Ren, Shuai, César Terrer, Juan Li, Yingfang Cao, Shanshan Yang, and Dan Liu. "Historical Impacts of Grazing on Carbon Stocks and Climate Mitigation Opportunities." *Nature Climate Change* 14, no. 4 (2024): 380-386.

Rhoades, C. C., A. J. Meier, and A. J. Rebertus. "Soil Properties in Fire-Consumed Log Burnout Openings in a Missouri Oak Savanna." *Forest Ecology and Management* 192, no. 2-3 (2004): 277-284. https://doi.org/10.1016/j.foreco.2004.01.012.

Richards, Brianna, and Christopher B. Craft. "Greenhouse Gas Fluxes from Restored Agricultural Wetlands and Natural Wetlands, Northwestern Indiana." In *The Role of Natural and Constructed Wetlands in Nutrient Cycling and Retention on the Landscape*, 17-32. 2015.

Rinehart, L. *Pasture, Rangeland and Grazing Management*. ATTRA -- National Sustainable Agriculture Information Service, 2008. https://rangemanagement.extension.colostate.edu/wp-content/uploads/sites/42/2020/07/Pasture_Range_Grazing_Management.pdf.

Robertson, G.P., E.A. Paul, and R.R. Harwood. "Greenhouse Gases in Intensive Agriculture: Contributions of Individual Gases to the Radiative Forcing of the Atmosphere." *Science* 289 (2000): 1922-1925. doi:10.1126/science.289.5486.1922.

Robinson, N.P., B.W. Allred, D.E. Naugle, and M.O. Jones. "Patterns of Rangeland Productivity and Land Ownership: Implications for Conservation Management." *Ecological Applications* 29, no. 3 (2019). https://doi.org/10.1002/eap.1862.

Rossell, Irene M., Kevin K. Moorhead, Huma Alvarado, and Robert J. Warren. "Succession of a Southern Appalachian Mountain Wetland Six Years Following Hydrologic and Microtopographic Restoration." *Restoration Ecology* 17, no. 2 (2009): 205-214.

Rouquette, Francis Monte Jr., Lynne E. Sollenberger, and Joao M. B. Vendramini. "Grazing Management and Stocking Strategy Decisions for Pasture-Based Beef Systems: Experimental Confirmation vs Testimonials and Perceptions." *Translational Animal Science* 7 (2023). doi.org/10.1093/tas/txad069.

Rutledge, S., A.M. Wall, P.L. Mudge, B. Troughton, D.I. Campbell, J. Pronger, C. Joshi, and L.A. Schipper. "The Carbon Balance of Temperate Grasslands Part I: The Impact of Increased Species Diversity." *Agriculture, Ecosystems & Environment* 239 (2017): 310-323. https://doi.org/10.1016/j.agee.2017.01.039.

Sala, O.E., L. Yahdjian, K. Havstad, and M.R. Aguiar. "Rangeland Ecosystem Services: Nature's Supply and Humans' Demand." In *Rangeland Systems*, edited by D. Briske. Springer Series on Environmental Management. Cham: Springer, 2017. https://doi.org/10.1007/978-3-319-46709-2_14.

Sanderman, Jonathan, Tomislav Hengl, and Gregory J. Fiske. "Soil Carbon Debt of 12,000 Years of Human Land Use." *PNAS* 114 (2017): 9575-9580.

Sanderman, Jonathan. "Low Quality Evidence Dominates Discussion of Carbon Benefits of Alternative Grazing Strategies." In prep for *Nature Climate Change*, 2025.

Sankey, J. B., S. Ravi, C. S. A. Wallace, R. H. Webb, and T. E. Huxman. "Quantifying Soil Surface Change in Degraded Drylands: Shrub Encroachment and Effects of Fire and Vegetation Removal in a Desert Grassland." *Journal of Geophysical Research: Biogeosciences* 117, no. G2 (2012): 2012JG002002. https://doi.org/10.1029/2012JG002002.

Sayre, N.F. *The New Ranch Handbook: A Guide to Restoring Western Rangelands*. 2011. http://quiviracoalition.org/images/global/303-NRH_Opening%20Pages.pdf.

Schantz, M.C., D.R. Smith, D. Harmel, D.J. Goodwin, D.R. Tolleson, J.M.O. Leyton, K.C. Flynn, J.L. Yost, K.R. Thorp, J.G. Arnold, M.J. White, and K. Adhikari. "The LTAR Integrated Grazing Land Common Experiment at the Texas Gulf." *Journal of Environmental Quality* (2024). doi: 10.1002/jeq2.20573.

Scott, Brian, Andrew H. Baldwin, and Stephanie A. Yarwood. "Quantification of Potential Methane Emissions Associated with Organic Matter Amendments Following Oxic-Soil Inundation." *Biogeosciences* 19, no. 4 (2022): 1151-1164.

Seyfried, M.S., and B.P. Wilcox. "Soil Water Storage and Rooting Depth: Key Factors Controlling Recharge on Rangelands." *Hydrological Processes: An International Journal* 20, no. 15 (2006): 3261-3275.

Scharenbroch, B.C., M.L. Flores-Mangual, B. Lepore, J.G. Bockheim, B. Lowery. “Tree encroachment impacts carbon dynamics in a sand prairie in Wisconsin.” *Soil Science Society of America Journal* 74 (2010): 956-968.

Shepherd, A., M.D. Hartman, N. Fitton, C.A. Horrocks, R.M. Dunn, A. Hastings, and L.M. Cardenas. "Metrics of Biomass, Live-Weight Gain and Nitrogen Loss of Ryegrass Sheep Pasture in the 21st Century." *Science of The Total Environment* 685 (2019): 428-441. doi:10.1016/j.scitotenv.2019.05.038.

Sigua, Gilbert C., Samuel W. Coleman, J. Albano. “Beef cattle pasture to wetland reconversion: impact on soil organic carbon and phosphorus dynamics. *Ecological Engineering* 35 (2009): 1231-1236.

Silva, L.S., M.K. Mullen, C. Prevatt, and J.J. Tucker. "Perceptions of Adoption of Alfalfa Plantings by Forage–Livestock Producers in the Southern United States." *Applied Animal Science* 37, no. 6 (2021): 665-669.

Silveira, Maria L., Priscilia J. Rodrigues da Cruz, J. Mauricio Bueno Vendramini, Elizabeth Boughton, Rosvel Bracho, Abmael da Silva Cardoso. “Opportunities to increase soil carbon sequestration in grazing lands in the southeastern United States.” *Grassland Research* 3 (2024): 69-78.

Skinner, Howard. "Winter Carbon Dioxide Fluxes in Humid-Temperate Pastures." *Agricultural and Forest Meteorology* 144, no. 1-2 (2007): 32-43.

Skinner, R. Howard. "Nitrogen Fertilization Effects on Pasture Photosynthesis, Respiration, and Ecosystem Carbon Content." *Agriculture, Ecosystems & Environment* 172 (2013): 35-41.

Skinner, R. Howard, and Curtis J. Dell. "Yield and Soil Carbon Sequestration in Grazed Pastures Sown with Two or Five Forage Species." *Crop Science* 56, no. 4 (2016): 2035-2044. https://doi.org/10.2135/cropsci2015.11.0711.

Sleeper, Benjamin E., and Robert L. Ficklin. "Edaphic and Vegetative Responses to Forested Wetland Restoration with Created Microtopography in Arkansas." *Ecological Restoration* 34, no. 2 (2016): 117-123.

Smith, D. *The Tallgrass Prairie Center Guide to Prairie Restoration in the Upper Midwest*. University of Iowa Press, 2010.

Smith, N.G., T.F. Keenan, I.C. Prentice, H. Wang, I.J. Wright, U. Niinemets, K.Y. Crous, et al. "Global Photosynthetic Capacity Is Optimized to the Environment." *Ecology Letters* 22 (2019): 506-517. doi.org/10.1111/ele.13210.

Sollenberger, Lynn E., Carmen T. Agouridis, Eric S. Vanzant, Alan J. Franzluebbers, and Lloyd B. Owens. "Prescribed Grazing on Pasturelands." In *Conservation Outcomes from Pasturelands and Hayland Practices: Assessments, Recommendations, and Knowledge Gaps*, edited by C. Jerry Nelson, 111-204. 2012. https://uknowledge.uky.edu/cgi/viewcontent.cgi?article=1039&context=bae_facpub.

Song, K., S. H. Lee, W. J. Mitsch, and H. Kang. "Different Responses of Denitrification Rates and Denitrifying Bacterial Communities to Hydrologic Pulsing in Created Wetlands." *Soil Biology and Biochemistry* 42, no. 10 (2010): 1721-1727.

Stanley, P., L. Roche, and T. Bowles. "Amping Up Soil Carbon: Soil Carbon Stocks in California Rangelands under Adaptive Multi-Paddock and Conventional Grazing Management." *International Journal of Agricultural Sustainability* 23 (2025). doi.org/10.1080/14735903.2025.2461826.

Stavi, IIan. “Wildfires in grasslands and shrublands: a review of impacts on vegetation, soil, hydrology, and geomorphology.” *Water* 11 (2019). https://doi.org/10.3390/w11051042

de Steiguer, J. E., Joel R. Brown, and Jim Thorpe. "Contributing to the Mitigation of Climate Change Using Rangeland Management." *Rangelands* 30, no. 3 (2008): 7-11. https://doi.org/10.2111/1551-501X(2008)30[7:CTTMOC]2.0.CO;2.

Stephens, Elizah Z., and Peter M. Homyak. "Post-Fire Soil Emissions of Nitric Oxide (NO) and Nitrous Oxide (N2O) across Global Ecosystems: A Review." *Biogeochemistry* 165, no. 3 (2023): 291-309.

Sulc, R.M., and A.J. Franzluebbers. "Exploring Integrated Crop-Livestock Systems in Different Ecoregions of the United States." *European Journal of Agronomy* 57 (2014): 21-30. doi.org/10.1016/j.eja.2013.10.007.

Svejcar, Tony, Raymond Angell, James A. Bradford, William Bugas, William Emmerich, Albert B. Frank, Tagir Gilmanov, Marshall Haferkamp, Douglas A. Johnson, et al. "Carbon Fluxes on North American Rangelands." *Rangeland Ecology & Management* 61 (2008): 465-474.

Tan, Kim H. *Environmental Soil Science*. CRC Press, 2009.

Tangen, Brian A., Raymond G. Finocchiaro, and Robert A. Gleason. "Effects of Land Use on Greenhouse Gas Fluxes and Soil Properties of Wetland Catchments in the Prairie Pothole Region of North America." *Science of the Total Environment* 533 (2015): 391-409.

Tangen, Brian A., and Sheel Bansal. "Soil Organic Carbon Stocks and Sequestration Rates of Inland, Freshwater Wetlands: Sources of Variability and Uncertainty." *Science of the Total Environment* 749 (2020): 141444.

Tankosić, Jelena Vapa, Svetlana Ignjatijević, Nemanja Lekić, Nataša Kljajić, Miloš Ivaniš, Slobodan Andžić, and Dejan Ristić. "The Role of Environmental Attitudes and Risk for Adoption with Respect to Farmers' Participation in the Agri-Environmental Practices." *Agriculture* 13, no. 12 (2023): 1-19.

Taylor, K. "Decadal Soil Health Responses to Adaptive Multipaddock (AMP) Rotational Grazing vs Season-Long Grazing in a Semiarid Rangeland." USDA-ARS Rangeland Resources and Systems Research Unit; CARM Stakeholder Group presentation, 2025. Manuscript in preparation.

Teague, R., and U. Kreuter. "Managing Grazing to Restore Soil Health, Ecosystem Function, and Ecosystem Services." *Frontiers in Sustainable Food Systems* 4 (2020): 28. doi.org/10.3389/fsufs.2020.534187.

Teague, W.R., S.L. Dowhower, R.J. Ansley, W.E. Pinchak, J.A. Waggoner. "Integrated grazing and prescribed fire restoration strategies in a mesquite savanna: 1. vegetation responses." *Rangeland Ecology & Management* 63 (2010): 275-285.

Throop, Heather L., Lara G. Reichmann, Osvaldo E. Salva, and Steven R. Archer. "Response of Dominant Grass and Shrub Species to Water Manipulation: An Ecophysiological Basis for Shrub Invasion in a Chihuahuan Desert Grassland." *Oecologia* 169 (2011): 373-383. https://doi.org/10.1007/s00442-011-2217-4.

Throop, Heather L., Steven R. Archer, and Mitchel P. McClaran. "Soil Organic Carbon in Drylands: Shrub Encroachment and Vegetation Management Effects Dwarf Those of Livestock Grazing." *Ecological Applications* 30 (2020). doi.org/10.1002/eap.2150.

Throop, Heather L. "Response of dominant grass and shrub species to water manipulation: an ecophysiological basis for shrub invasion in a Chihuanhuan Desert Grassland." Oecologica, 169: 373-383 (2012).

Tilhou, N.W., R.L.G. Nave, S. Jagadamma, N. Eash, and J.T. Mulliniks. "Forage Species and Summer Management Impacts on Soil Carbon and Nitrogen in Winter Stockpiled Grazing Systems." *Agroecosystems, Geoscience & Environment* 4 (2021). doi.org/10.1002/agg2.20132.

Tian, H., G. Chen, C. Lu, X. Xu, W. Ren, B. Zhang, K. Banger, B. Tau, S. Pan, M. Liu, C. Zhang, L. Bruhwiler, and S. Wofsy. "Global Methane and Nitrous Oxide Emissions from Terrestrial Ecosystems Due to Multiple Environmental Changes." *Ecosystem Health and Sustainability* 1 (2017). doi: 10.1890/EHS14-0015.1.

Twidwell, D., W. E. Rogers, C. L. Wonkka, C. A. Taylor, and U. P. Kreuter. "Extreme Prescribed Fire during Drought Reduces Survival and Density of Woody Resprouters." *Journal of Applied Ecology* 53, no. 5 (2016): 1585-1596. https://doi.org/10.1111/1365-2664.12674.

Twidwell, Dirac, William E. Rogers, Samuel D. Fuhlendorf, Carissa L. Wonkka, David M. Engle, John R. Weir, Urs P. Kreuter, and Charles A. Taylor Jr. "The Rising Great Plains Fire Campaign: Citizens'

Response to Woody Plant Encroachment." *Frontiers in Ecology and the Environment* 11 (2013): 64-71. https://doi.org/10.1890/130015.

Ueckert, D.N., T.L. Whigham, B.M. Spears. "Effect of burning on infiltration, sediment, and other soil properties in a mesquite-tobosagrass community." J*ournal of Range Management* 31 (1978): 420-425.

Ukhurebbor, K.E., U.O. Aigbe, R.B. Onyancha, J.N. Ndunagu, O.A. Osibote, J.O. Emegha, V.A. Balogun, H.S. Kusuma, H. Darmokoesoemo. "An overview of the emergence and challenges of land reclamation: issues and prospect." *Applied and Environmental Soil Science* (2022). https://doi.org/10.1155/2022/5889823.

U.S. Army Corps of Engineers. *Corps of Engineers Wetlands Delineation Manual*. Environmental Laboratory U.S. Army Corps of Engineers, Waterways Experiment Station, Wetlands Research Program Technical Report Y-87-1. Vicksburg, MS, 1987.

U.S. Forest Service (USFS). "Indicators of Grazing Intensity." 5352543. https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5352543.pdf.

USDA. "Range Technical Note No: NM-113." 2017. https://efotg.sc.egov.usda.gov/references/public/NM/RangeTechNote113.pdf.

USDA. "Practice Specification, Prescribed Grazing Appendix A (Code 528)." Natural Resources Conservation Service, 2019. https://efotg.sc.egov.usda.gov/api/CPSFile/23671/528_ND_PS_Prescribed_Grazing_2019.

USDA. "Conservation Practice Standard, Pasture and Hay Planting Code 512." Natural Resources Conservation Service, 2020a. https://www.nrcs.usda.gov/sites/default/files/2022-09/Pasture_and_Hay_Planting_512_NHCP_CPS_2020.pdf.

USDA. "Conservation Practice Standard, Range Planting Code 550." Natural Resources Conservation Service, 2020b. https://www.nrcs.usda.gov/sites/default/files/2022-09/Range_Planting_550_NHCP_CPS_2022.pdf.

USDA. "Guide to Pasture Condition Scoring." Natural Resources Conservation Service, 2020. https://www.nrcs.usda.gov/sites/default/files/2022-06/National_Pasture_Condition_Scoring_Guide_and_Score_Sheet_-_January_2020.pdf.

USDA. "Grazing Management and Soil Health: Keys to Better Soil, Plant, Animal and Financial Health." 2022. https://www.nrcs.usda.gov/sites/default/files/2022-09/Grazing%20Management_SoilHealth_0.pdf.

USDA. "Major Land Resource Area (MLRA)." 2022. https://www.nrcs.usda.gov/resources/data-and-reports/major-land-resource-area-mlra.

USGCRP. *Fifth National Climate Assessment*. Edited by A.R. Crimmins, C.W. Avery, D.R. Easterling, K.E. Kunkel, B.C. Stewart, and T.K. Maycock. Washington, DC: U.S. Global Change Research Program, 2023. https://doi.org/10.7930/NCA5.2023.

Van Andel, J., and J. Aronson. *Restoration Ecology: The New Frontier*. 2nd ed. West Sussex, UK: Wiley-Blackwell, 2012.

Van Auken, O.W. "Shrub invasions of North American Semiarid Grasslands". *Annual Review of Ecology, Evolution, and Systematics*, 31: 197-215. https://doi.org/10.1146/annurev.ecolsys.31.197

Vargas, R., S. L. Collins, M. L. Thomey, J. E. Johnson, R. F. Brown, D. O. Natvig, and M. T. Friggens. "Precipitation Variability and Fire Influence the Temporal Dynamics of Soil CO2 Efflux in an Arid Grassland." *Global Change Biology* 18, no. 4 (2012): 1401-1411. https://doi.org/10.1111/j.1365-2486.2011.02628.x.

Vivoni, Enrique R., Eli R. Perez-Ruiz, Russel L. Scott, Adam T. Naito, Steven R. Archer, Joel A. Biederman, and Nicole P. Templeton. "A Micrometeorological Flux Perspective on Brush Management in a Shrub-Encroached Sonoran Desert Grassland." *Agricultural and Forest Meteorology* 313 (2022). doi.org/10.1016/j.agrformet.2021.108763.

Walsh, K.B., and J. Rose. "A Review of Restoration Techniques and Outcomes for Rangelands Affected by Oil and Gas Production in North America." *Ecological Restoration* 40 (2022). doi: 10.3368/er.40.4.259.

Wang, H., Z. Yang, X. Ju, X. Song, J. Zhang, S. Li, and X. Zhu-Barker. "Quantifying Nitrous Oxide Production Rates from Nitrification and Denitrification under Various Moisture Conditions in Agricultural Soils: Laboratory Study and Literature Synthesis." *Frontiers in Microbiology* 13 (2022). https://doi.org/10.3389/fmicb.2022.1110151.

Wang, J., Y. Li, E.W. Bork, G.M. Richter, H.-I. Eum, C. Chen, S.H.H. Shah, and S. Mezbahuddin. "Modelling Spatio-Temporal Patterns of Soil Carbon and Greenhouse Gas Emissions in Grazing Lands: Current Status and Prospects." *Science of The Total Environment* 739 (2020): 139092. doi:10.1016/j.scitotenv.2020.139092.

Wang, Y., W.X. Duan, C. Tu, S. Washburn, L. Cheng, and S. Hu. "Soil Carbon, Nitrogen and Microbial Dynamics of Pasturelands: Impacts of Grazing Intensity and Planting Systems." *Pedosphere* 24 (2014): 408-416. https://doi.org/10.1016/S1002-0160(14)60027-8.

Wang, G., J. Li, D. Dukes, H.B. Gonzales, J.B Sankey. "Post-fire redistribution of soil carbon and nitrogen at a grassland-shrubland ecotone. *Ecosystems* 22 (2018): 174-188.

Watson, P.A., H.D. Alexander, and J.D. Moczygemba. "Coastal Prairie Recovery in Response to Shrub Removal Method and Degree of Shrub Encroachment." *Rangeland Ecology & Management* 72 (2019): 275-282.

Westberg, H., B. Lamb, K.A. Johnson, and M. Huyler. "Inventory of Methane Emissions from U.S. Cattle." *Journal of Geophysical Research: Atmospheres* 106 (2001): 12633-12642. doi.org/10.1029/2000JD900808.

Wheeler, C. Winston, Steven R. Archer, Gregory P. Asner, and Chad R. McMurtry. "Climate/Edaphic Controls on Soil Carbon/Nitrogen Response to Shrub Encroachment in Desert Grassland." *Ecological Applications* 17 (2007): 1911-1928. doi.org/10.1890/06-1580.1.

Whitehead, David. "Management of Grazed Landscapes to Increase Soil Carbon Stocks in Temperate, Dryland Grasslands." *Frontiers in Sustainable Food Systems* 4 (2020): 585913. https://doi.org/10.3389/fsufs.2020.585913.

Williams, B. "Passive and Active Adaptive Management: Approaches and an Example." *Journal of Environmental Management* 92, no. 5 (2011): 1371-1378. doi: 10.1016/j.jenvman.2010.10.039.

Williams, M.I., G.E. Schuman, A.L. Hild, and L.E. Vicklund. "Wyoming Big Sagebrush Density: Effects of Seeding Rates and Grass Competition." *Restoration Ecology* 10, no. 2 (2002): 385-391. doi: 10.1046/j.1526-100X.2002.01025.x.

Wilmer, H., M.E. Fernández-Giménez, S. Ghajar, et al. "Managing for the Middle: Rancher Care Ethics under Uncertainty on Western Great Plains Rangelands." *Agriculture and Human Values* 37 (2020): 699-718. https://doi.org/10.1007/s10460-019-10003-w.

Wilson, David, Dominique Blain, J. Couwenberg, Chris D. Evans, Daniel Murdiyarso, Susan Page, Florence Renou-Wilson, Jack Rieley, A. Sirin, and Maria Strack. "Greenhouse Gas Emission Factors Associated with Rewetting of Organic Soils." 2016.

Wohl, Ellen. "Landscape-Scale Carbon Storage Associated with Beaver Dams." *Geophysical Research Letters* 40 (2013): 3631-3636.

Wood, C. W., and B. A. Mealor. "Identifying Structural Thresholds in Annual Grass–Invaded Rangelands." *Rangeland Ecology & Management* 83 (2022): 1-9.

Wu, Z., W. Wang, W. Zhu, P. Zhang, R. Chang, G. Wang. “Shrub ecosystem structure in response to anthropogenic climate change: a global synthesis.” *Science of the Total Environment* 953. https://doi.10.1016/j.scitotenv.2024.176202

Xia, Y., J. Sanderman, J.D. Watts, M.B. Machmuller, A.L. Mullen, et al. "Coupling Remote Sensing with a Process Model for the Simulation of Rangeland Carbon Dynamics." *ESS Open Archive* (2025a). doi:10.22541/essoar.171269303.34921116/v1.

Xia, Yushu, Jonathan Sanderman, Jennifer D. Watts, Stephanie A. Ewing, Alex Parisien, Chris Burdett, Todd Hogrefe, John Lamoreux, and Holly Bambord. "Robust Carbon Monitoring, Reporting, and Verification for Grassland Management in the Great Plains." Report for the National Fish and Wildlife Foundation, 2025b. doi: 10.1575/1912/71403.

Xu, S., N. Eisenhauer, A. F. A. Pellegrini, J. Wang, G. Certini, C. A. Guerra, and D. Y. F. Lai. "Fire Frequency and Type Regulate the Response of Soil Carbon Cycling and Storage to Fire across Soil Depths and Ecosystems: A Meta-Analysis." *Science of The Total Environment* 825 (2022): 153921. https://doi.org/10.1016/j.scitotenv.2022.153921.

Xu, S., M. Delgado-Baquerizo, Y. Kuzyakov, Y. Wu, L. Liu, Y. Yang, Y. Li, Y. Yu, B. Zhu, and H. Yao. "Positive Soil Priming Effects Are the Rule at a Global Scale." *Global Change Biology* 30 (2024): e17502. doi:10.1111/gcb.17502.

Yang, Y., D. Tilman, G. Furey, and C. Lehman. "Soil Carbon Sequestration Accelerated by Restoration of Grassland Biodiversity." *Nature Communications* 10, no. 1 (2019): 718.

Yang, P., L. Zhang, D. Y. F. Lai, H. Yang, L. Tan, L. Luo, et al. "Landscape Change Affects Soil Organic Carbon Mineralization and Greenhouse Gas Production in Coastal Wetlands." *Global Biogeochemical Cycles* 36 (2022): e2022GB007469. https://doi.org/10.1029/2022GB007469.

Yao, J., W. Yuan, Z. Gao, H. Liu, X. Chen, Y. Ma, E. Arntzen, and D. McFarland. "Impact of Shifts in Vegetation Phenology on the Carbon Balance of a Semiarid Sagebrush Ecosystem." *Remote Sensing* 14, no. 23 (2022): 5924. doi: 10.3390/rs14235924.

Yu, L., Y. Huang, W. Zhang, T. Li, and W. Sun. "Methane Uptake in Global Forest and Grassland Soils from 1981 to 2010." *Science of the Total Environment* 607-608 (2017): 1163-1172. doi.org/10.1016/j.scitotenv.2017.07.082.

Yu, T., R. Sun, Z. Xiao, Q. Zhang, G. Liu, T. Cui, and J. Wang. "Estimation of Global Vegetation Productivity from Global Land Surface Satellite Data." *Remote Sensing* 10 (2018): 327. https://doi.org/10.3390/rs10020327.

Zaady, E., D. J. Eldridge, and M. A. Bowker. "Effects of Local-Scale Disturbance on Biocrusts." In *Biological Soil Crusts: An Organizing Principle in Drylands*, 429-449. 2016.

Zhou, L., X. Zhou, J. Shao, Y. Nie, Y. He, L. Jiang, Z. Wu, and S.H. Bai. "Interactive Effects of Global Change Factors on Soil Respiration and Its Components: A Meta-Analysis." *Global Change Biology* 22 (2016): 3157-3169. https://doi.org/10.1111/gcb.13253.

Zhou, Yong, Thomas W. Boutton, and X. Ben Wu. "Carbon Response to Woody Plant Encroachment: Importance of Spatial Heterogeneity and Deep Soil Storage." *Journal of Ecology* (2017). doi.org/10.1111/1365-2770.

Zou, Junyu, Alan D. Ziegler, Deliang Chen, Gavin McNicol, Philippe Ciais, Xin Jiang, Chunmiao Zheng, Jie Wu, Jin Wu, and Ziyu Lin. "Rewetting Global Wetlands Effectively Reduces Major Greenhouse Gas Emissions." *Nature Geoscience* 15, no. 8 (2022): 627-632.

Zuazo, V.H.D., and C.R.R. Pleguezuelo. "Soil-Erosion and Runoff Prevention by Plant Covers: A Review." In *Sustainable Agriculture*, 785-811. 2009.

7. Glossary

Abiotic: Physical and non-living component of an ecosystem that shapes its environment

Adaptive Management: A structured, iterative process of decision-making that allows for flexible responses to changing environmental conditions and new scientific knowledge

Adaptive, Multi-Paddock Rotational Grazing: A form of rotational grazing that applies high intensity grazing (typically with high stocking rates) followed by a long period of rest. Also referred to as “intensive grazing” and considered a form of “regenerative” grazing

Afforested Rangelands: Rangelands or pasture where tree encroachment or planting has taken place (lands have > tree cover)

Biological Crust: A complex community of living organisms, including cyanobacteria, algae, lichens, mosses, and fungi, that forms a thin layer on the soil surface in arid and semi-arid ecosystems

Carbon Sequestration: The process by which plants absorb carbon dioxide (CO2) from the atmosphere and store it in biomass and soils

Carbon Sink: Natural or artificial process that removes carbon from the atmosphere

Climate Forcing: A factor that changes the climate system and can be caused by natural (ex. volcanos) or human (ex. GHG emissions) activities

CO2 Equivalent: A metric used to compare the global warming potential (GWP) of different greenhouse gases. It is calculated by multiplying a gas’s mass in metric tons by its GWP

Conventional Grazing: See continuous grazing

Continuous Grazing: A grazing system where livestock graze in a pasture or land parcel for an extended period of time (e.g. xx months) without rotation (i.e., periods of time where livestock are removed from the land unit to allow plant rest and recovery). Also commonly referred to as “traditional” or “conventional” grazing

Dissolved Organic Carbon (DOC): Small fragments of organic material small enough to pass through a fine filter; dissolved in water

Enteric Methane: A greenhouse gas produced by the natural digestive process occurring in wild and domesticated ruminant animals

Global Warming Potential (GWP): A measure of how much heat a greenhouse gas traps in the atmosphere over a unit of time in comparison with the same unit of carbon dioxide (CO2). Measures the potency of contributions to warming relative to carbon dioxide

Grazing Deferment: A delay of grazing in an area to provide adequate time for plant reproduction, establishment of new plants, and regrowth of existing plants

Grazing Intensity: A function of grazer intensity and duration. Refers to how much vegetation is removed by grazing within a given area and timeframe

Grazing Recovery Period: The length of time between grazing periods on rotationally stocked pastures; this period should be long enough to allow for recovery of green, photosynthetically active, biomass

Grazing Rest: The absence of grazing by livestock to benefit plants for regrowth between grazing periods, for plant growth, development, and establishment

Grazing Lands: Lands capable of providing forage for livestock

Greenhouse Gases (GHG): Gases, such as carbon dioxide (CO2), methane (CH4), and nitrous oxide (N2O)-that trap heat from the sun in the earth's atmosphere and contribute to global warming

Histosol: A type of soil dominated by organic material

Hydric Soils: Soils that are saturated by water

Intensive Grazing: See definition for "adaptive, multi-paddock rotational grazing"

Ley Pasture: Temporary pastures that are integrated into crop sequences

Megaton: One million tons

Microtypography: Topographic variability of the soil at small scales

Monitoring: A structured approach used to assess land health, grazing impacts, and conservation effectiveness over time

Net GHG: The full balance of greenhouse gas emissions including soil organic carbon sequestration balanced with emissions of CO2, CH4, and N2O, typically expressed in **CO2 equivalent**

Net primary productivity (NPP): Carbon sequestered through photosynthesis minus CO2 loss through cell respiration

Pasture: More agronomically managed lands, relative to rangelands, often with domesticated forage species and used for hay production and/or grazing

Petragram: 1015 grams

Prescribed Grazing: According to the NRCS this management type is where: "the timing, intensity, degree of use, frequency, duration, and season of grazing will be manipulated to promote ecologically healthy and economically stable plant communities"

Rangelands: Natural ecosystems that are predominately grasses, grass-like species, forbs or shrubs suitable for livestock grazing or browsing.

Reclamation: The process of rehabilitating degraded land through plantings and other improvements, to restore function of core ecosystem cycles (i.e., carbon, water, energy nutrient cycles), biodiversity, etc.

Regenerative Grazing: A livestock management approach that prioritizes restoring and improving soil health, plant diversity and ecosystem function

Restoration: The process of restoring land to its natural state, or state before land degradation or conversion

Soil Organic Carbon (SOC): The component of soil organic matter that consists of carbon, critical for soil health, water retention, and nutrient cycling

Soil Mineralization: The process of breaking down organic matter by microbes, fungi, etc. to access energy and nutrients. A product of mineralization is CO2

Teragram (Tg): One metric ton (Mt)

Traditional Grazing: See continuous grazing definition

Transitional Rangelands: Rangelands or pasture where, if left unmanaged (e.g., ungrazed, or without biomass removal), they would transition into shrubland or forest

Appendix A: Supporting Tables for Practice Standard Review

Table 1 NRCS grazing lands Conservation Practice Standards (CPS) considered for ecosystem service co-benefits, including climate mitigation potential.

CONSERVATION PRACTICE STANDARD (CPS) NAME	CPS ID	PRACTICE DESCRIPTION SUMMARY	LINK TO FULL PRACTICE DESCRIPTION
Prescribed Grazing	528	*Managing the harvest of vegetation with grazing and/or browsing animals with intent to achieve ecological, economic and management objectives.*	https://www.nrcs.usda.gov/sites/default/files/2022-09/Prescribed_Grazing_528_CPS.pdf
Brush Management	314	*The management or removal of woody plants including those that are invasive and noxious.*	https://www.nrcs.usda.gov/sites/default/files/2022-09/Brush_Management_314_CPS-3-17Final.pdf
Prescribed Burning	338	*Applying a planned fire to a predetermined area of land to manage undesirable vegetation, to reduce risks of wildfire, to improve forage quality and habitat diversity.*	https://www.nrcs.usda.gov/sites/default/files/2022-09/Prescribed_Burning_338_Overview_10_2020.pdf
Pasture and Hay Planting	512	*Establishing adapted and compatible species, varieties, or cultivars of*	https://www.nrcs.usda.gov/sites/default/files/2022-

CONSERVATION PRACTICE STANDARD (CPS) NAME	CPS ID	PRACTICE DESCRIPTION SUMMARY	LINK TO FULL PRACTICE DESCRIPTION
		perennial herbaceous plants suitable for pasture or hay production.	09/Pasture_and_Hay_Planting_512_NHCP_CPS_2020.pdf
Range Planting	550	*The seeding and establishment of herbaceous and woody species for the improvement of vegetation composition and productivity of the plant community to meet management goals.*	https://www.nrcs.usda.gov/sites/default/files/2022-09/550_NHCP_CPS_Range_Planting_2021_0.pdf
Wetland Restoration	657	*The re-establishment of abiotic conditions (hydrology, topographic features, and substrate) on filled or partially, effectively, or fully drained wetlands to a close approximation of pre-disturbed conditions.*	https://www.nrcs.usda.gov/resources/guides-and-instructions/wetland-restoration-ac-657-conservation-practice-standard
Riparian Herbaceous Cover	390	*Establishment of vegetation tolerant of flooding and saturated soils within the transitional zone between upland and aquatic habitats.*	https://www.nrcs.usda.gov/sites/default/files/2022-11/390-NHCP-CPS-Riparian-Herbaceous-Cover-2022.pdf
Herbaceous Weed Treatment	315	*The removal or control of herbaceous weeds including invasive, noxious, prohibited or undesirable plants.*	https://www.nrcs.usda.gov/sites/default/files/2022-09/Herbaceous_Weed_Treatment_315_CPS_10_2020.pdf

Table 2 North American (U.S. and Canada) research studies identified by report authors, that examine impacts of grazing management on rangeland carbon cycle components (carbon stocks, greenhouse gas (GHG) fluxes).

Study Description										Carbon & Greenhouse Gas Impact			
Study Number; Reference	**Site Name; State or Provence**	**Mean Annual Temp. (MAT; °C)**	**Mean Annual Precip. (MAP; mm)**	**Plant Type**	**Soil Texture**	**Years**	**Treatment Type**	**What Was Measured?**	**Soil Sampling Depth (cm)**	**Belowground Organic Carbon Pool**	**Aboveground Organic Carbon Pool (Biomass)**	**GHG Flux**	**Study Conclusion**
1. Morgan et al. (2016)	USDA-ARS CPER ; Nunn, CO	15.6	341	Native shortgrass	Loam	2001-2003; 2004-2006	[1]No grazing; Cont. Moderate grazing (0.6 AU/mon); Cont. Heavy grazing (0.9 AUM/mon)	Tower CO_2 fluxes; Aboveground biomass	NA	NA	Severely reduced live biomass with heavy grazing.	[2] NEE was carbon source to neutral with heavy grazing; NEE was annual [3]sink with no to moderate grazing.	Heavy grazing can be detrimental to range productivity and carbon cycling; Appropriate grazing (e.g., moderate) may increase ecosystem resilience to drought.
2. Taylor et al. (2025)	USDA-ARS CPER ; Nunn, CO	15.6	341	Native shortgrass	Loam	2013-2024	Cont. Moderate grazing; adaptive (high intensity) rotational grazing	SOC	0-15	[4] SOC stock decreased in both treatments. No significant difference between treatments.	NA	NA	No significant difference in SOC stock between continuous and rotational grazing treatments.
3. Ingram et al. (2008)	Cheyenne, WY	5.6	425	Native mixed grass	Loam/Sandy Loam	[5]1982-2003	No grazing; Cont. Light grazing (0.16-23 AU ha^{-1}); Cont. Heavy (0.56 AU ha^{-1})	SOC; aboveground biomass; microbial community	0-5, 5-15, 15-30, 30-60	[4] SOC stock decreased in all treatments. Significantly > SOC lost in ungrazed and heavy grazed, relative to lightly grazed system.	Largest decreases in live plant biomass occurred under heavy grazing.	NA	Evidence that appropriate grazing (e.g., light) may increase ecosystem carbon resilience to drought. Heavy grazing is detrimental to SOC and aboveground biomass.
4. Hillenbrand et al. (2019)	Hermosa, SD	6.8	500	Native shortgrass	Clay/Clay Loam/Loam	[6]1980-2016	Rotational bison grazing (13 AU 100 ha^{-1}) vs Cont. Light (14 AU 100 ha^{-1}) and Heavy (51 AU 100 ha^{-1}) at nearby ranches.	SOC; aboveground biomass	0-100	No difference in SOC stock between bison and Light cattle grazing sites; both had significantly > SOC relative to Heavy Grazing site.	Significantly higher live plant biomass at bison site, compared to High and Light cattle grazed sites.	NA	Heavy continuous grazing substantially reduced SOC stock and plant biomass, relative to light and rotational grazing. Heavy grazing substantially increased bare ground.

5. Hewins et al. (2018)	[7]Alberta, CAN	1.9 to 5.1	336 to 637	Mixed grass prairie	Variable	Soils sampled 30-60 yrs following exclosures	Grazing exclosures vs Light to Moderate growing season cont. cattle grazing	SOC	0-15; 15-30	SOC concentrations significantly > in grazed vs ungrazed	NA	NA	SOC concentrations significantly > in grazed vs ungrazed; however, interpret with caution as SOC stocks were not quantified.
6. Schantz et al. (2024)[8]	Riesel, Central TX	19.4	900	Native tallgrass prairie + bermudagrass & kleingrass	Clay	2012-2016	[8] Year-round cont. Moderate (50 cow/calf vs rotational grazing (50 cow/calf followed by rest)	SOC; aboveground biomass	0-5	SOC stocks > under continuous relative to rotational management	NA	NA	SOC stocks > under continuous moderate grazing relative to rotational management; interpret with caution given very shallow soil sampling.
7. Apfelbaum et al. (2022)[b]	KY, TN, AL, MS	>14	>1000	Mixed native grass & weeds	Silt Loam/ Silt Clay Loam/	SOC sampling in 2018	[9] Five paired ranches; Cont. Low Density vs Rotational Low Density	SOC	0-30; 30-50; 50-100	On average (however, not in all ranch pairs) SOC stocks significantly > for rotational grazing vs cont.	NA	NA	On average (however, not in all ranch pairs) SOC stocks significantly > for rotational grazing vs cont. Interpret with caution as multiple variables including management history are unknown.
8. Gomez-Casanovas (2018)[c]	Lake Placid, FL	20	1310	Semi-native grass + bahiagrass	Loam	2013-2015	[10] No grazing vs continuously grazed (light to moderate; 0.3 to 0.9 AU/ha)	Tower CO_2 & CH_4 fluxes; chamber soil CO_2 & CH_4; aboveground biomass; SOC	0-55	Grazing significantly < root biomass 0-10 cm; > root biomass below 10 cm. SOC stock > in grazing vs ungrazed at 0-5 cm	Grazing significantly reduced live and dead biomass, relative to ungrazed	[11] Grazing significantly increased NEE sink over time; NEE at ungrazed was NEE (CO_2) neutral. Global warming potential (GWP) less in grazed site.	Continuous (light to moderate) grazing management showed positive benefits to SOC, NEE, and GWP, relative to ungrazed.
9. Stanley et al. (2025)	Coastal California	~9.5	>800 to >1500	Semi-native grassland	Loam, Loam/Sand, Silt Loam, Clay	2020	[12] Paired ranches; multi-density continuously grazed vs rotational.	SOC and plant cover	0-10, 10-30, 0-100	Adaptive grazing plots had significantly > SOC relative to continuously grazed sites. SOC accumulation	All adaptive management sites had less bare ground relative to continuously grazed plots.	NA	NA

										> at sites having > precipitation. SOC accumulation low-to-none at sites with high clay content.			

[1] Before treatment all parcels were moderately grazed for 50 years. [2] Net ecosystem CO2 exchange (NEE) observed through year-round monitoring of gas flux using the eddy covariance method. [3] The first experiment (2001-2023) occurred during severe drought and included heavy grazing (NEE = 192 gCO_2 m^{-2} yr^{-1}), no grazing (NEE = -142 gCO_2 m^{-2} yr^{-1}), moderate grazing (NEE = -186 gCO_2 m^{-2} yr^{-1}); second experiment (2004-2006) included moderate grazing (NEE = -267 gCO_2 m^{-2} yr^{-1}) and severe grazing (NEE = -2 gCO_2 m^{-2} yr^{-1}). [4] Severe decrease in MAP over treatment years. [5] No grazing before 1982. [6] Continuous grazed with cattle until 1980s. Adaptive bison management started in 1980s; soil sampling in 2016. [7] Within Alberta Environment & Parks Rangeland Reference Area management units; 30-60 year old grazing exclosures within light to moderately cattle grazed units (AU not reported); 108 paired control/grazed sites in total. [8] Aimed for 50% forage utilization rate with 50 cow/calf, regardless of continuous grazing vs rotational grazing; noted severe drought during study. [9] AU not reported for the evaluated ranches; past management history not reported. [10] Both sites continuously moderately grazed (0.62 AU/ha) for > 30 years before treatment. [11] Ungrazed NEE was ~ 50 to -25 gC m^{-2} yr^{-1} vs -150 to -200 gC m^{-2} yr^{-1} for ungrazed (183 to -91 gCO_2 m^{-2} yr^{-1} vs -550 to -733 gCO_2 $m^{-2}yr^{-1}$). [b] Also see Johnson et al. (2022). [c]Also see Wilson et al. (2017) and Wade et al. (2022). [12] Four paired ranches having conventional continuous grazing (varying stocking density, ~ 0.11 to 0.28 AU/acre/year, annual average) versus adaptive multi-paddock grazing (i.e., a form of rotational grazing) with varying stocking densities ~ 0.05 to 0.5 AU/acre/year. Adaptive grazing practiced for at least three years.

Table 3 North American (U.S. and Canada) research studies identified by report authors, that examine impacts of grazing on carbon cycle components (carbon stocks, greenhouse gas (GHG) fluxes) in managed (previously cultivated, planted, and/or fertilized) pastures.

Study Description										Carbon & Greenhouse Gas Impact			
Study Number; Reference	**Site Name; State or Provence**	**Mean Annual Temp. (MAT; °C)**	**Mean Annual Precip. (MAP; mm)**	**Plant Type**	**Soil Texture**	**Years**	**Treatment Type**	**What Was Measured?**	**Soil Sampling Depth (cm)**	**Belowground Organic Carbon Pool**	**Aboveground Organic Carbon Pool (Biomass)**	**GHG Flux**	**Study Conclusion**
1. Mehre et al. (2024)	[1] Southern Ontario, CAN	6.2 to 8	892 to 1184	Pasture mixed grass/legume; previously cultivated[1]	Loam/Silt Loam	Sampling in 2021, 10 years after starting rotational grazing treatment	Cont. cattle grazing (AU ha^{-1} 6 to 19) vs Rotational grazing (AU ha^{-1} 7 to 13.5)	SOC	0-60	[2] SOC stock significantly > in rotational grazing (across-site average) vs cont. grazing.	NA	NA	SOC stock significantly > in rotational grazing (across-site average) vs cont. grazing. However, interpret with caution as other possibly conflicting variables (e.g., differences in plant species, management history, etc. were not accounted for.
2. Wang et al. (2014)	Goldsboro, NC	21	1220	Planted pasture in 2003; ryegrass; sundangrass/sorghum; bermudagrass; previously disturbed soil	Sandy	Sampling in 2007 (4 years after study start)	Cont. cattle grazing; Low stocking (2.47 cows ha^{-1}) vs high stocking (3.7 cows ha^{-1})	SOC	0-10; 10-30	SOC stock significantly > under high intensity grazing relative to low intensity, for ryegrass/sorghum/sudangrass plantings	NA	NA	SOC stock significantly > under high intensity grazing relative to low intensity, for ryegrass/sorghum/sudangrass plantings. Note the mechanism(s) driving > SOC under some species is unclear.
3. Debuex et al. (2006) [a]	Gainesville, FL	23	1244	Planted pasture (pensacola bahiagrass) > 10 years old; nitrogen fertilized	Sandy	Sampling in 2004	Cont. grazing at low (1.4 AU ha-1) vs moderate (2.8 AU ha-1) vs high (4.2 AU ha-1) vs	SOC; aboveground biomass	0-8	No significant difference in SOC stock between treatments. Increased intensity increased light density, possibly indicating faster SOC turnover (yielding	NA	NA	No significant difference in SOC stock between treatments. Increased intensity increased light density, possibly indicating faster SOC turnover (yielding

							rotational grazing with high stocking rates.			increase in plant nutrients)			increase in plant nutrients). Note: past management history unclear.
4. Amorim et al. (2020)[b]	Boonville, AR [3]	16	1266	Non-native mountain pasture (bermudagrass).	Silt Loam	2003-2017	[3]Continuously cattle grazed; hayed; Rotationally grazed on artificial buffer strip; riparian strip	SOC	0-15	No significant differences in SOC stock between treatments.	NA	NA	No significant differences in SOC stock between treatments. Interpret with caution given lack of detail of past management history, grazing density, other land characteristics, etc.
5. Tilhou et al. (2021	Spring Hill, TN	14	1372	[4] Seeded red clover; native mix.	Silt Loam	2013-2016	[4] Cont. light cattle grazing vs hayed pasture	SOC	0-5; 5-15	Grazed had significantly > SOC concentrations (by 12%) at 5 cm depth.	NA	NA	Grazed had significantly > SOC concentrations (by 12%) at 5 cm depth. Interpret with caution as SOC stocks not reported.
6. Mosier et al. (2021)	[5]Woodville, MS	13 to 19	1310 to 1649	Planted pasture; previously cultivated grain crop	Loam/Silt	Sampling in 2018	[5] Paired rotational cattle grazing vs cont. grazing units	SOC; root biomass	0-85	Significantly (average 13%) > SOC stock (to depth) in rotational grazing sites relative to continuous. No difference in root biomass.	NA	NA	Significantly (average 13%) > SOC stock (to depth) in rotational grazing sites relative to continuous. Interpret with caution given variable past management history.

[1] Five paired ranches selected, located within 15 km of each other; 20 fields in total. All sites previously cultivated cropland (20+ years prior). [2] 120 MgC ha^{-1} vs 106.6 MgC ha^{-1}. [a] Also see Silveira et al. (2013); another study at site; did not report significant differences in SOC. [3] Plots were 15 watersheds (0.14 ha each). No information about past land use, nor grazing intensity. Assumed previously cultivated/disturbed land given plantings and creation of artificial buffer strip. Lands fertilized with poultry manure. [4] Past land history unknow. Assumed previously cultivated given seeing of red clover and native mix. AU unknown. [5] Five paired rotational grazing vs continuous grazing units; variable stocking rates spanning 0.67 to 3.78 AU/ha. Many of the treatment sites previously cultivated (in row crop) before pasture conversion and grazing. [b] Also see Xu et al. (2021).

Table 4 North American (U.S. and Canada) research studies identified by report authors, that examine impacts of brush management on ecosystem carbon storage and GHG exchange.

Study Description										Carbon & Greenhouse Gas Impact			
Study Number	**Reference**	**Study Type**	**Study Site Name**	**State**	**Ecotype**	**MAT (°C)**	**MAP (mm)**	**Soil Texture**	**Brush Species Managed**	**Management Type**	**Belowground Carbon Pool**	**Aboveground Carbon Pool**	**Carbon & GHG Flux**	**Ecosystem Carbon**
1	Miwa & Reuter, 2010	Brush management (treatment)	NA; Privately-owned ranch	Oregon	Western juniper encroached shrubland	8.1	250	Clay loam/loam	Western juniper (trees)	Mechanical; tree cutting occurred 8-15 years ago	**Negative impact of cutting**; Higher total SOC concentrations where juniper had been located, relative to soils further from grass interspaces	NA	NA	NA
2	Derner et al., 2014	Brush management (treatment)	Seedskadee BLM Allotment; Cow Hollow BLM Allotment	Wyoming	Sagebrush-steppe	3.6	277	Loam; sandy loam	Wyoming big sagebrush	[2]Chemical (Spike® 20P [tebuthiuron]); Mechanical (mowing)	**Neutral:** No differences in SOC stock between treatment and control at sandy loam site [3]**Positive impact of treatment**; Increase in surface (0 – 5 cm) and subsurface soil (5 – 15 cm) organic carbon at loam site having herbicide treatment	**Negative**: Loss of annual aboveground carbon biomass from shrub mowing.	[4]**Negative**: Loss of annual CO_2 uptake (plant productivity) from shrub treatment, especially from chemical treatment.	NA
3	Connin et al., 1997	Brush encroachment	Jornada Basin	New Mexico	Semidesert grassland / mesquite shrubland complex	17	345	Sandy (with high calcium carbonate)	Velvet mesquite	NA; comparing brush encroached with grass plots	[5]**Postive impact of brush**; SOC higher in velvet mesquite plots than in grass plots	NA	NA	NA
4	Perkins et al., 2005	Brush management	Las Cruces and Socorro	New Mexico	Semidesert grassland / mesquite	17	345	Sandy	Creosote	[6]Chemical (aerial application of Tbuthiuron; 0.56 to 0.84 kg ha^{-1})	**Neutral impact of treatment**; Surface SOC stocks (0 – 5 cm) were not affected by	NA	NA	NA

		(treatment)	BLM districts		shrubland complex						creosote management >10 years post-treatment, despite increase in grass cover			
5	Throop et al., 2012	Brush encroachment (water manipulation experiment)	Santa Rita Experimental Range	Arizona	Semidesert grassland / mesquite shrubland complex	19.3	345	Sandy loam	Velvet mesquite	NA	**Positive to neutral impact of shrub;** SOC stocks were larger under velvet mesquite than grass in areas that received low – moderate MAP; SOC did not differ between grass and velvet mesquite in areas that received high MAP	NA	NA	NA
6	Abdallah et al., 2020	Brush management (treatment)	Camp Creek Paired Watershed	Oregon	Western juniper encroached shrubland	6.1	358	Loam; silt loam	[7]Western juniper (trees)	Mechanical (chainsaw; leaving only old growth juniper)	**Neutral – Positive impact of treatment**; SOC stock higher near stumps and live woody biomass in both treated and control sites, relative to grass-dominated interspaces SOC stock in grass interspaces was higher in treated plots relative to grass interspaces in control	**Negative impact of treatment**; Western juniper and herbaceous aboveground carbon stocks were much smaller in treated area (13 years post-treatment) relative to untreated, despite slightly higher grass cover in treated.	NA	NA
7	McClaran et al. 2008	Brush management	Santa Rita Experimental Range	Arizona	Semidesert grassland / mesquite shrubland complex	18	358	Sandy loam	Velvet mesquite	Chemical spray 40 years prior study	**Negative impact of treatment**; Surface (0 – 5 cm) and subsurface (5 – 10 cm) soil organic SOC lower under grass	NA	NA	NA

		(treatment)									(where shrub death occurred) relative to velvet mesquite control.			
8	DeMarco et al., 2016	Brush management (treatment)	Santa Rita Experimental Range	Arizona	Semidesert grassland / mesquite shrubland complex	18	370	Sandy loam	Velvet mesquite	Chemical; Mechanical	**Negative impact of treatment**; Surface (0 – 5 cm) and subsurface l (5 – 20 cm) SOC lower under grass than velvet mesquite 8-years post-treatment; Long-term impacts of velvet mesquite (52 years post treatment) was associated with a decrease in SOC	NA	NA	NA
9	Throop et al., 2020	Brush management (treatment)	Santa Rita Experimental Range	Arizona	Semidesert grassland / mesquite shrubland complex	18	370	Sandy loam	Velvet mesquite	Chemical	**Negative impact of treatment**; Treatment of velvet mesquite was associated with a decrease in SOC (0 – 20 cm)	NA	NA	NA
10	Coultrap et al. (2008)	Brush management (treatment)	Multiple Sites	California	Mixed shrub and grass	0	410	NA	Western Juniper; Big sagebrush	Primarily mechanical (through also some fire and chemical)	NA	**Neutral impact of treatment:** Decreased shrub and grass productivity; increased herbaceous productivity; increased weed (cheatgrass) productivity	NA	NA
11	Hunt et al. (2004)	Brush encroachment	Chugwater; Shirley Basin	Wyoming	Mixed native grass	16.4; 11.2	240 (sagebrush); 390 (grass)	NA	Sagebrush	NA; grass compared to shrub encroached land	NA	NA	**Positive impact of brush.** Multi-year monitoring of the site	NA

					prairie vs sagebrush								showed significantly larger annual NEE sink in sagebrush system (avg. -30 gC) whereas the grassland ranged from a small sink to small source (avg. 0.11 gC) in very dry years.	
12	Wheeler et al., 2007	Brush encroachment	Santa Rita Experimental Range	Arizona	Semidesert grassland / mesquite shrubland complex	18	330; 430	Sandy loam	Velvet mesquite	NA; grass compared to shrub encroached land	**Positive impact of brush**; SOC stocks (0 – 20 cm) smaller under grass than velvet mesquite	NA	NA	NA
13	Vivoni et al., 2022	Brush management (treatment)	Santa Rita Experimental Range	Arizona	Semidesert grassland / mesquite shrubland complex	18	275 - 450	Sandy loam; loamy sand	Velvet mesquite	[8] Chemical (aerial spray)	NA	**Neutral impact of treatment**; Decreased grass cover; increased bare ground in interspaces; increased shrub foliage	**Positive impact of treatment**; Ecosystem carbon uptake was enhanced by higher gross primary productivity (associated with shrub recovery and improved water use efficiency) and lower ecosystem respiration	NA

													two years post-treatment	
14	Jackson et al., 2002	Brush encroachment	Multiple sites	Colorado; New Mexico; Texas	Semidesert grassland / mesquite shrubland complex	NA	230; 277; 322; 660; 840	NA	Honey mesquite / Velvet mesquite / creosote	NA; grass compared to shrub encroached land	**Positive to negative impact of brush**; SOC stocks increased with brush encroachment at sites with drier soils, and decreased at wet soil sites	NA	NA	NA
15	Heitschmidt et al., 1986	Brush management (treatment)	W. T. Waggoner Estate	Texas	Temperate savanna	17	652	Clay loam	Honey mesquite (trees)	[9] Chemical (aerial)	NA	**Neutral impact of treatment:** No significant differences in herbaceous biomass 5-years post-treatment (total biomass including shrubs unknown)	**Neutral impact of treatment;** Herbaceous NPP was not affected by management of honey mesquite (total productivity including shrubs is unknown)	NA
16	Gill & Burke, 1999	Brush encroachment	La Copita Research Area; Sevilleta National Wildlife Refuge	Texas; New Mexico	Subtropical thorn woodland	22.4; 7.1	716	NA	Creosote; Honey mesquite	NA; grass compared to shrub encroached land	**Positive impact of brush**; SOC concentrations higher under creosote and honey mesquite than grass	NA	NA	NA
17	Hibbard et al., 2001; Hibbard et al. 2003	Brush encroachment	La Copita Research Area	Texas	Subtropical thorn woodland	22.4	716	Sandy loam	Honey mesquite	NA; grass compared to shrub encroached land	**Positive impact of brush**; SOC stock higher under honey mesquite relative to grass	NA	NA	NA

18	Liao et al., 2006	Brush encroachment	La Copita Research Area	Texas	Subtropical thorn woodland	22.4	716	Sandy loam	Honey mesquite	NA; grass compared to shrub encroached land	**Positive impact of brush**; SOC lower in grass plots relative to honey mesquite	NA	NA	NA
19	Scharenbroch et al., 2010	Brush encroachment	Lower Wisconsin River Valley	Wisconsin	Dry sand prairie grassland	6.6	790	Sandy	Red pine (tree)	NA; Comparison of native prairie with encroached shrubland	**Neutral:** No difference in SOC stocks between sites	NA	NA	**Positive impact of shrub**; Ecosystem carbon stock (woody + understory + soil carbon) was much larger in areas with red pine encroachment
20	McKinley et al., 2008	Brush encroachment	Konza Prairie Biological Station	Kansas	Tallgrass prairie	12.7	835	Silty clay loam	Eastern red cedar	NA	NA	NA	NA	**Positive impact of shrub**; Ecosystem carbon was larger in areas with eastern red cedar encroachment than without
21	Connell et al., 2020	Brush encroachment	Konza Prairie Biological Station	Kansas	Tallgrass prairie	12.7	835	Silty clay loam	Roughleaf dogwood	NA; Native tallgrass prairie vs dogwood shrub encroachment	**Positive impact of shrub**; Soil organic carbon stocks increased over time as dogwood and other woody species cover increased	NA	NA	NA
22	McCulley et al., 2004	Brush encroachment	Texas A&M AgriLife La Copita	Texas	Subtropical savanna parkland	22.4	680	Sandy loam	Honey mesquite	NA; Grassland compared to mesquite encroached land	**9Positive impact of shrub**; Soil organic carbon stocks and rates of increase were smaller under grass	NA	NA	NA

	; Zhou et al., 2017		Research Area								than honey mesquite at multiple depths (0 – 5 cm, 5 – 15 cm, 15 – 30 cm)			
23	Geesing et al., 2000	Brush encroachment	Multiple sites	Texas	Temperate savanna	~11	645; 698; 701; 706; 850	NA	Velvet mesquite (tree)	NA; Grassland compared to mesquite encroached land	**Positive impact of shrub;** Soil organic carbon stocks were smaller under grass than velvet mesquite	NA	NA	NA

[1] Moderate canopy mature (80-90 year old) juniper, categorized as "tree" instead of shrub. Trees were fully removed to create open spaces for grass growth. Results from this study should be used with caution. Impacts from larger juniper trees (having more of a canopy effect) are likely different from shrubs. [2] Plots were sampled 12 years following treatment. Chemical treatment yielded more dead biomass, however also produced lower density plots with taller (live) sagebrush. [3]. Estimated annual SOC sequestration rates of 0.16 and 0.14 Mg C ha^{-1} yr^{-1} for the 0-5 and 5-15 cm soil depths at loam site (likely because of increased grass cover). [4] Estimated loss of 0.13 Mg C ha^{-1} yr^{-1} in CO_2 uptake for chemical treatment plot at loam site due to plant death. [5] Through isotope analysis (13C and 14C) the study found lower carbon turnover associated with the mesquite-influenced soils, relative to the grass-influenced soils. This and shrub root carbon stored in deeper soils likely contribute to higher SOC in shrub sites. [6] Treatment resulted in 85% creosote brush mortality. [7] Mature juniper watershed where juniper was tree height (< 3 meters) and semi-open canopy. Mechanical pruning removed 90% of trees in the treatment watershed; residue was left on the ground. [8] Two flux towers at this site. Herbicide treatment was by helicopter (18 ha area) using clopyralid, aminopyralid, triclopyr at rates of 1157, 512, 1170 ml ha-1. [9] 1:1 mixture of 2,4,5-trichlorophenoxy and picloram in diesel oil water; 0.6 kg/ha. 9 Estimated SOC accumulation rates were 12-18 Mg C ha^{-1} higher in areas with encroached woody shrub relative to adjacent grassland.

Table 5 North American (U.S. and Canada) research studies identified by report authors that examine impacts of prescribed fire management on ecosystem carbon storage and GHG exchange.

Study Description				Carbon & Greenhouse Gas Impact			
Study Number; Reference	**Study Site Name**	**State**	**Ecosystem**	**Belowground Carbon**	**Aboveground Carbon**	**Carbon & GHG Flux**	**Ecosystem Carbon**
1. Potts et al., 2012	Irvine Ranch Conservancy	California	Mediterranean grassland	NA	NA	**Neutral**; Aboveground net primary productivity did not change following prescribed burning	**Neutral**; Net ecosystem exchange did not change following prescribed burning
2. Ueckert et al., 1978	Spade Ranch	Texas	Mesquite-tobosagrass grassland	**Negative; Neutral;** Soil organic carbon either decreased or did not change following prescribed burning, depending on antecedent soil moisture	NA	NA	NA
3. MacNeil et al., 2008	Fort Keogh Livestock & Range Research Laboratory	Montana	Grassland	NA	NA	**Positive**; CO_2 fluxes decreased following prescribed burning	NA
4. Wang et al., 2018	Sevilleta National Wildlife Refuge	New Mexico	Grassland	**Neutral**; Soil organic carbon did not change following prescribed burning	NA	NA	NA
5. Fultz et al., 2016	Erskine Native Rangeland Research Station	Texas	Shortgrass prairie	**Neutral;** Soil total carbon did not change following prescribed burning	NA	**Negative**; CO_2 flux increased following prescribed burning in the short-term	NA
7. Jariel et al., 2014	NA	Texas	Mixed-grass savanna	**Negative;** Soil organic carbon decreased following prescribed burning in the short-term	NA	NA	NA

8. Knapp et al., 1998	Konza Prairie Biological Station	Kansas	Tallgrass prairie	NA	NA	**Negative**; CO_2 flux increased following prescribed burning	NA
9. Rau et al., 2009	Underdown Canyon	Nevada	Sagebrush-steppe	**Positive**; Soil total carbon increased following prescribed burning in the short-term	NA	NA	NA
10. Ansley et al., 2006	NA	Texas	Mixed-grass savanna	**Neutral; Positive;** Soil organic carbon increased following summer prescribed burning; Soil organic carbon didn't change following winter prescribed burning	NA	NA	NA
11. Dai et al., 2006	W. T. Waggoner Estate	Texas	Temperate savanna	**Positive**; Surface soil organic carbon increased following summer prescribed burning, and summer and winter prescribed burning	NA	NA	NA
Dangi et al., 2010	Savory Creek Watershed	Wyoming	Sagebrush-steppe	**Negative;** Soil organic carbon decreased following prescribed burning	NA	NA	NA
12. Harris et al., 2008	W. T. Waggoner Ranch	Texas	Temperate savanna	**Negative;** Soil organic carbon decreased following prescribed burning	NA	NA	NA
13. Li et al., 2014	Cedar Creek Ecosystem Science Reserve	Minnesota	Grassland	NA	NA	**Neutral;** Aboveground net primary productivity did not change following prescribed burning	NA
14. Rhoades et al., 2004	AB Savanna	Missouri	Oak savanna	**Negative**; Soil total carbon decreased following prescribed burning	NA	NA	NA
15. Sankey et al., 2012	Sevilleta National Wildlife Refuge	New Mexico	Desert grassland	**Negative**; Soil total and organic carbon decreased following prescribed burning	NA	NA	NA

16. Teague et al., 2010	W. T. Waggoner Estate	Texas	Temperate savanna	**Neutral;** Soil organic carbon did not change following prescribed burning	NA	NA	NA
17. Vargas et al., 2012	Sevilleta National Wildlife Refuge	New Mexcio	Desert grassland	NA	NA	**Positive;** CO_2 flux decreased following prescribed burning	NA
18. Henry et al., 2006	Jasper Ridge Biological Preserve	California	Mediterranean grassland	NA	NA	**Negative;** Net primary production decreased following prescribed burning	NA

Table 6 Summary of synthesis of U.S.-based research studies that examined the effects of prescribed burning on ecosystem carbon storage and GHG exchange.

Study Description	**Carbon & Greenhouse Gas Impact**			
Reference	**Belowground Carbon**	**Aboveground Carbon**	**Carbon & Greenhouse Gas Flux**	**Ecosystem Carbon**
1. Limb et al., 2016	**Negative; Neutral; Positive**; Soil organic carbon either decreased, didn't change, or increased following prescribed burning, depending on the ecosystem	NA	**Negative; Positive**; CO_2 flux either decreased or increased following prescribed burning, depending on the ecosystem	NA
2. Stephens and Homyak, 2023	NA	NA	**Negative**; N_2O flux increased following prescribed burning regardless of ecosystem	NA
3. Stavi, 2019	**Negative**; Soil organic carbon tends to decrease following prescribed burning regardless of ecosystem	NA	NA	NA
4. Silveira et al., 2024	**Neutral**; Soil organic carbon didn't change following prescribed burning in Florida pine flatwood ecosystems	NA	NA	NA
5. Xu et al., 2022	**Negative**; Soil organic carbon decreased following prescribed burning regardless of ecosystem	NA	NA	NA

Table 7 U.S. research studies identified by report authors that examine impacts of pasture and hay plantings, or similarly relevant treatment comparisons, on ecosystem carbon storage and GHG exchange.

Study Description										Carbon & Greenhouse Gas Impact			
Study Number; Reference	**Site Name; State or Provence**	**Mean Annual Temp. (MAT; ˚C)**	**Mean Annual Precip. (MAP; mm)**	**Plant Type**	**Soil Texture**	**Years**	**Treatment Type**	**What Was Measured?**	**Soil Sampling Depth (cm)**	**Belowground Carbon Pool**	**Aboveground Carbon Pool (Biomass)**	**GHG Flux**	**Study Conclusion**
1. Pannu et al., 2019	WSU Irrigated Agriculture and Research Station; Prosser, WA	17	175	Switchgrass biofuel production on	Coarse silt	2012-2013	Switchgrass : alfalfa 70:30, 50:50. Switchgrass only. No fertilizer applied to switchgrass: alfalfa mixes, control and N fertilizer applied to switchgrass only plots.	Soil N_2O emissions and yields	NA	NA	Fertilized switchgrass plots had highest biomass, followed (descending order) by switchgrass: alfalfa 70:30, 50:50, and unfertilized switchgrass.	N_2O emissions in plots seeded with switchgrass: alfalfa 70:30 were lower than 50:50 plots. N_2O emissions were also lower than plots growing only unfertilized and fertilized switchgrass crops.	Intercropping alfalfa with switchgrass reduced N_2O emissions relative to fertilized switchgrass. However, yields were lower for switchgrass with alfalfa than fertilized switchgrass.
2. Johnson et al., 2012	Swan Lake Research Farm; MN		645	Hay cropping, row cropping	Silty clay loam	2006-2008	N and P fertilizer applied to conventional crops, manure applied to	Chamber CH_4 and N_2O emissions; aboveground biomass		NA	Organic alfalfa had a higher biomass in first year. Conventional alfalfa had higher biomass	CH_4 was sequestered in organic alfalfa plots during all	Incorporating alfalfa into annual cropping rotation can sequester CH_4,

							organic alfalfa crop. Conventional and organic alfalfa grown with and without wheat. Corn and soy examined				pools in following years.	three years and more of a sink than conventional alfalfa or. N_2O emissions were positive for both cropping systems.	particularly for organic alfalfa.
3. Anthony and Silver, 2024	CA			Hay cropping, pasture, row cropping		2017-2021.	Alfalfa, grazed native pasture, conventional corn. N fertilizer applied to corn only	Chamber CO2, CH4, and N_2O emissions		NA	NA	CO_2 and CH_4 were sequestered in alfalfa and grazed native pasture, while N_2O emissions were slightly positive.	Alfalfa system was a greater GHG sink than pasture, while corn system was a net source.
4. Robertson 2000 Sci Reports	MI			Hay cropping, row cropping, restored grassland, forest, poplar plantation		1991-1999	Alfalfa, conventional and organic corn, soy, wheat rotation plus legume cover crop. N fertilizer applied to conventional rotation. No fertilizer applied to alfalfa plot.	Chamber N_2O and CH_4 emissions; SOC.	0-7.5 cm	Soil C was sequestered in alfalfa plots.	Aboveground net primary production was slightly greater in conventional cropping rotation.	Alfalfa plot had a drawdown in CH_4 and released N_2O with net-decrease in GHG emissions. Cropping systems with legume cover	Alfalfa and legume covers were effective at decreasing fertilizer inputs and overall GHG impact.

												crops had lower emissions than conventional.	
5. Barsotti et al., 2013	Fort Ellis Research and Extension Center, Montana State University. Bozeman, MT		453	Perennial alfalfa vs. Annual cropping	Silt loam	2010-2011	Alfalfa, continuous wheat, wheat/ pea-barley rotation. N fertilizer applied to spring wheat and spring wheat/ pea-barley rotation; no N fert on alfalfa.	Chamber CO_2, N_2O, and CH_4 fluxes. SOC.	0-15cm	Spring wheat/ pea-barley rotation sequestered SOC at a greater rate than alfalfa.	Alfalfa system had higher biomass than spring wheat/ pea-barley rotation. Alfalfa system had higher biomass than spring wheat system when grazed but not chemically treated.	Alfalfa system released GHG emissions and had lower emissions than spring wheat rotations	Spring wheat/ pea-barley rotation had the greatest SOC sequestration rate; alfalfa systems had lowest GHG emissions.
6. Ingram et al. 2015	Smith Ranch; Lodgepole, SD		380	Hay cropping	Sandy loam	2003	Alfalfa and native grassland were unfertilized. A single application of N fertilizer was applied to a separated native grassland plot.	Chamber CO_2 and N_2O emissions. SOC.	0-15 cm	SOC stocks were greater in alfalfa plots than fertilized and unfertilized native grassland	Alfalfa had higher biomass than native grassland.	CO_2 was sequestered but N_2O was released in both alfalfa and native grassland systems	GHG emissions were comparable between alfalfa, native grassland, and fertilized grassland. SOC stocks were greatest in alfalfa system.
7. Andrews et al., 2022	University of CA Desert Research and Extension Center; Holtville, CA	18		Hay cropping	Fine silty	2019-2020	Alfalfa and Sudan grass systems. N fertilized applied to Sudan grass. No fertilizer applied to alfalfa plots.	Chamber CO_2 and N_2O emissions.	NA	NA	Sudan grass system had higher biomass than alfalfa system.	CO_2 and N_2O were released in both Sudan grass and alfalfa systems.	Emissions were higher in Sudan grass than alfalfa and positive for both GHGs.

8. Collier et al., 2016	Wisconsin Integrated Cropping System Trial; Madison, WI	6.9	869	Hay cropping	Silt loam	2011	Alfalfa and switchgrass systems. N fertilizer applied to switchgrass; manure applied to alfalfa	Chamber N_2O emissions.	NA	NA	NA	N_2O was released in both systems, slightly lower in switchgrass system. Much greater in fertilized switchgrass plots.	Alfalfa and unfertilized switchgrass released much lower N_2O emission compared to fertilized switchgrass systems.
9. Abagandura et al. 2020	Felt Research Farm; Brookings, SD	5.9	466	Biofuel cropping ; legume intercropping		2014-2018	Prairie cordgrass, clover. Different N fertilizer rates applied to plots growing only prairie cordgrass; no fertilizer applied to plot growing cordgrass and clover	Chamber CH_4, CO_2 and N_2O emissions. SOC.	0-15 cm	SOC content was comparable across treatments.	Clover crop produced higher biomass than all prairie cordgrass treatments, for highest fertilized plots.	Clover system had highest CO_2 and lowest N_2O compared to unfertilized and fertilized prairie cordgrass systems. No trend in CH_4 fluxes.	Clover cropping decreased N_2O emission but increased CO_2, while emission of all three GHGs generally increased with fertilizer application.
10. Skinner, 2013	Haller Research Farm; State College, PA			Grazed pasture	Silt loam	2003-2011	Perennial forage species. Low and high rates of N fertilizer applied	Eddy covariance CO_2 flux	NA	NA	Aboveground biomass was higher in treatment receiving high-N inputs.	CO_2 was sequestered in plots receiving low N fertilizer, and released in pastures with high N	Low N inputs caused CO_2 emissions to decrease, leading to a net benefit.

												fertilizer inputs	
11. Skinner, 2007	Haller Research Farm; State College, PA			Hay croppin g, grazed grassla nd	Silt loam	2003-2005	Alfalfa, grassland. No fertilizer applied	Eddy covariance CO_2 flux	NA	NA	NA	CO_2 was released in both alfalfa and grasslan d plots	Alfalfa plot released less CO_2 than grass plot.

Table 8 U.S. research studies identified by report authors that examine impacts of rangeland plantings on ecosystem carbon storage and GHG exchange.

Study Description										Carbon & Greenhouse Gas Impact			
Study Number; Reference	**Site Name; State or Provence**	**Mean Annual Temp. (MAT; ˚C)**	**Mean Annual Precip. (MAP; mm)**	**Plant Type**	**Soil Texture**	**Years**	**Treatment Type**	**What Was Measured?**	**Soil Sampling Depth (cm)**	**Belowground Carbon Pool**	**Aboveground Carbon Pool (Biomass)**	**GHG Flux**	**Study Conclusion**
1. Hunt et al. (2004)	Chugwater, WY; Shirley Basin, WY	390, 240		Mixed-grass prairie, sagebrush steppe		1999	NA	Chamber CO_2 emissions	NA	NA	NA	Positive NEE in sagebrush; neutral NEE in mixed-grass prairie	Net release of CO2 in both sagebrush systems examined.
2. Yao et al. (2022)	AmeriFlux Experimental Site; Hanford, WA	200	11.8	Sagebrush steppe	Sandy	2019-2020	NA	Eddy covariance CO_2 emissions	NA	NA	NA	Negative NEE during first year; positive NEE during second year	Sagebrush system was a net CO_2 sink.
3. Gilmanov et al. (2006)	Northern Great Basin Experimental Range, Burns, OR; US Sheep Experiment Station, ID	380 (OR); 700 (ID)	7.6 (OR); 6.2 (ID)	Sagebrush steppe	Sandy loam (OR); Loam (ID)	1995-2001	NA	Chamber CO_2 emissions	NA	NA	NA	Positive NEE at both sites	Sagebrush systems were a net CO_2 source
4. Beltz et al. (2019)	Pinedale, WY	290	2.6	Sagebrush steppe	Loam	2012-2015	NA	Chamber CH_4, CO_2, and N_2O emissions.	NA	NA	NA	CO_2 was released regardless of N addition. CH_4 and N_2O emissions were close to zero.	Net source of CO_2, neutral CH_4 and N_2O.
5. Norton et al. (2008)	WY	7.8	340	Sagebrush steppe		2003	NA	Chamber CO_2, and N_2O emissions.	NA	NA	NA	CO_2 and N_2O emissions were both released	Net source of CO_2 and N_2O emissions.

Table 9 North American (U.S. and Canada) research studies identified by report authors that examine impacts of herbaceous weed treatment on ecosystem carbon storage and GHG exchange.

Study Description							Carbon and Greenhouse Gas Impact			
Study	**Study type**	**Study Site Name**	**State**	**Ecotype**	**Mean Annual Precipitation (MAP) (mm)**	**Treatment Type**	**Belowground Carbon Pool**	**Aboveground Carbon Pool**	**Carbon & GHG Flux**	**Ecosystem Carbon**
Bradley et al 2006	cheatgrass Invasion	Ry Patch Res., Button Point, Jungo	Great Basin	Salt Desert Shrub to Sagebrush	190-210	No Management, just impacts of invasion	Inconsistent	Lower in cheatgrass plots due to loss of shrubs (22-94 g c m2 compared to 160-670)	na	
Caspi et al 2018	Annual Grass invasion	Santa Monica Mountains, Bernard Field station, Crafton Hills	California	California sage scrub	324-461	No Management, just impacts of invasion	Only A horizon, decrease of ~20% concentration,	na	na	
Mahood et al 2021	cheatgrass Invasion	20 sites	Nevada	Northern and Central B Asin and Range Ecoregion	254-304	No Management, just impacts of invasion	Generally decreases, but climate and invasion stage dependent	Decreases w/Invasion	na	
Nagy et al 2020	lit review, cheatgrass invasion			Great Basin		No Management,	Increase in SOC in top 10cm with cheatgrass, SOC below 20cm related	Decreases w/Invasion		"We estimate that conversion from native sagebrush to

						just impacts of invasion	to time since last fire. (losses observed >5 years after fire)			cheatgrass leads to a net reduction of C storage in biomass and litter of 76 c C/m2, or 16 Tg C across the Great Basin without management practices like native sagebrush restoration or cheatgrass removal"
Koteen et al 2011	non-native grass invasion	Tennessee Valley and Bolinas Laggon	California	Native Perennial grassland	900	No Management, just impacts of invasion	Average drop in soil carbon storage of 40 Mg/ha in top meter of soil	Decreased with invasion		
Wood and Mealor, 2022	Chemical control of cheatgrass	Pinedale and Saratoga	Wyoming			Liquid or granular herbicide application across range of invasion	na	na	na	na
Endress et al 2012	Chemical control	Wenaha State Wildlife Area	Oregon	Blue Mountain Ecoregion	430	Various herbicide treatments w/ and w/out reseeding	na	na	na	
Katz et al 2025	Site comparison of annual grass invasion	Colorado Plateau and Wyoming Basin	Colorado and Wyoming	Colorado Plateau and Wyoming Basin	259-262	No Management, just impacts of invasion	Variable by depth, type of C, and location. In CO, MAOC stocks were 36.1% less on 0-10 cm depth and 46.1% less in 10-	na	na	na

				Ecoregions			20cm depth, but not in Wyoming (cooler, more carbon rich ecoregion). Increase in SOC in 0-10 cm depth in WY, but via POC (unstable pool)			
Kluender et al 2025	Chemical control of cheatgrass (indaziflam and imazapic)	Minidoka National Wildlife Refuge	Idaho	Snake River Plain/Wyonig big sagebrush	267	Indaziflam w/ w/out imazapic, aerial application	na	na	na	na
Masters et al 1996	Leafy-spurge control	Ainsworth and Ansley	Nebraska	Sub-irrigated meadow in Sand Hills		Herbicide application, burning, reseeding	na	na	na	na`
Maxwell et al 2024	Annual grass invasion (mostly cheatgrass, but one site with Medusahead)	Comparisons across multiple sites		Snake river Plain, Nothern Basin and Range, and Idaho Batholith	253-377	No Management, just impacts of invasion	Carbon stocks in 60-100cm depths were 735 or 43% less in unburned or burned landscapes compared to uninvaded landscapes.	Invaded or burned areas had 55% and 93% less aboveground biomass than not burned and not invaded sites.		
Rau et al 2011	cheatgrass Invasion	Onaqui, Owyhee, Roberts, Grey Butte, Moses Coulee, Rock Creek, Saddle	Washington, Oregon, Idaho, Nevada, Utah	Sage brush steppe	215-307	No Management, just impacts of invasion	Decreases as invasion increases	na	na	na

		Mountain								
Foster et al 2025	Old World Bluestem Control	Bee, Leberg and San Patricio Counties	Texas			Plowing, herbicides, fire, mowing, fertilizing, seeding	na	na	na	na
Kray et al, 2025	Targeted grazing of annual bromes	NE and WY	Nebraska and Wyoming	mixed-grass rangeland	403	Targeted grazing	na	na	na	na

Table 10 U.S. research studies identified by report authors that examine impacts of the hydrological restoration component of wetland restoration on ecosystem carbon storage and GHG exchange, including paired field site, before-after, timeline, and inundation studies.

Study Description					Carbon and Greenhouse Gas Impact						
Study	**Study type**	**Location**	**Soil type**	**Time since restoration**	**Climate Mitigation Benefits? Yes/No, Mixed, or Uncertain**	**CH4**	**N2O**	**CO2**	**SOC**	**DOC**	**Biomass**
1. Gleason et al. 2009	Paired field site: Cropland vs restored	North Dakota	silty	na	Yes, slight	Did not vary by treatment			Slightly more SOC in restored wetlands (30 cm)	na	
						Contributed to 9% net GHG flux; Inundation greatly reduced flux	Contributed to 1% net GHG flux	Contributed to 90% net GHG flux			
2. Daniel et al. 2019	Paired field site: Cropland vs restored playas vs reference playas	Nebraska (mixed & long grass prairie)	silty	na	Yes	Did not vary by treatment		CO_2 was 28% higher in cropland than restored wetlands; neither were different from reference playas	na	na	na
3. Daniel	Paired field site:	Nebraska (shortgra	silty	na	Yes	Cropland and grassland	Restored playas emitted 43%		na	na	na

et al. 2019	Cropland vs restored playas vs native grassland playas	ss prairie)				playas emitted 46 x & 23x more CH4, respectivel y than restored wetlands	less N2O than cropland playas				
4. Tangen et al. 2015	Paired field site: Cropland nondrained, cropland drained, restored, native prairie	North Dakota, Minneso ta, Iowa	silty	na	Mixed. Restor ing drained cropland catchments does not have climate benefits, while restoring non-drained cropland does.	Restoring drained cropland catchment s results in increased CH4 fluxes, while restoring non-drained cropland does not	Restoring any cropland catchment results in decreased N2O fluxes		Variable among treatments with no clear pattern	na	na
5. Euliss et al. 2006	Paired field site: Reference, agricultural nondrained, agricultural drained, restored	Montana , North Dakota, South Dakota, Minneso ta, Iowa	silty	<5 yrs & >5 yrs	Yes. Restored wetlands have the potential to have climate mitigation benefits, due to the significantly greater amount of SOC in references wetlands. Examination of time since restoration (next line)				More SOC in reference wetland than all other categories; restoration did not affect SOC	na	na

					further supports this						
6. Euliss et al. 2006	Space for time field study: Years since restoration for semipermanent (SP) and Seasonal (SA) wetlands	Montana, North Dakota, South Dakota, Minnesota, Iowa	silty	0-14 yrs	Mixed: Yes: Restored SP wetlands ; Uncertain for SA				SOC in the top 15 cm increased across 14 years at a rate of 3.05 Mg OC/ha/y in SP; no measured accrual in SA	na	na
7. Gleason et al. 2008	Paired field site: Cropland vs restored catchments vs native prairie catchments	Montana, North Dakota, South Dakota, Minnesota, Iowa	silty	na	Uncertain				More SOC in top 15 cm in native prairie than in cropland; No difference in SOC stocks in restored catchments relative to cropland.	na	na
8. Gleason et al. 2008	Space for time field study: Years since restoration for glaciated plains and Missouri Coteau	Montana, North Dakota, South Dakota, Minnesota, Iowa	silty	0-20 yrs	Uncertain				No detection of a relationship between time since restoration and SOC levels.	na	na
9. Sigua et al. 2009	Before-After restoration field comparison plus comparison	Florida	sandy	1 yr	Yes				SOC in top 20 cm was greatest in intact wetlands, followed by restored wetlands, then unrestored areas	na	na

	to intact wetland										
10. Reed et al. 2022	Space for time field study: Years since restoration for glaciated plains and Missouri Coteau	Californi a	silty	0-22 yrs	Yes				Soil C stocks (0-15 cm) increased by 232.9 g C m-2 y-1	na	Root biomass (0-15 cm) increased at a rate of 270.3 g m-2 y-1
11. Tagen et al. 2020	Paired field site: Reference, agricultural nondrained, agricultural drained, restored	Montana , North Dakota, South Dakota, Minneso ta, Iowa	silty	1 - 35 yrs, 60% rest ored ≤10 yrs ago, 83% restored ≤15 years ago	Uncertain study examined sequestration on a plot scale rather than a landscape scale				Proportional loss of SOC not affected by landscape position; Decreasing SOC mass per unit area from the inner catchment to the upland. Storage was greatest at inner catchment and lowest in upland areas.	na	na
12. Tagen et al. 2020	Space for time field study: Years since restoration for glaciated plains and Missouri Coteau	Montana , North Dakota, South Dakota, Minneso ta, Iowa	silty	1 - 35 yrs, 60% rest ored ≤10 yrs ago, 83% restored ≤15 years ago	Yes				Results suggest that it could take as few as 20 years to as many as 64 years to replenish SOC stocks depending on landscape position and depth	na	na
13. Marton et al. 2014	Paired field site: Intact vs restored	Indiana	sandy	10 yrs	NA: Comparison doesn't include non-restored wetland				SOC in the top 5 cm was greater in intact wetlands relative to restored wetlands	na	na

14. Richards & Craft 2015	Paired field site: Intact vs restored	Indiana	sandy	10 yrs	NA: Comparison doesn't include non-restored wetland	Did not vary by treatment			4x more SOC in intact than restored wetlands (10 cm)	nab	na
						Smallest contributor of global warming potential, 0.1% & 0% in restored & intact wetlands, resp.	Contributed 2.0% & 1.7% of global warming potential in restored in intact wetlands, resp.				
15. Mushet et al. 2022	Adjacent wet-dry zones	North Dakota	clayey	NA: Intact wetland	Mixed: Wet areas encourage increased release of CH_4, N_2O; mixed results for CO_2 depending on type of wetland.	Greater in wet zone than dry zone for temporarily/seasonally (TS) and semi-permanently ponded (SP)		Less in wet zone than dry zone for TS, opposite for SP	Not greater in wet zone for TS or SP (%)	na	na
16. Mushet et al. 2022	Temporarily / seasonally (TS) hydrated vs. Semi-permanently ponded (SP) wetlands				Mixed: TS release more emissions than SP wetland; However they also sequester more SOC	2x greater in TS wetland	6.5x greater in TS wetland	3x greater in TS wetland	2x greater in TS wetland	na	na
17. Hondula et al. 2021	Adjacent wet-dry zones	Maryland	sandy	NA: Intact wetland	Mixed or Uncertain: Inundation extent and duration, but not frequency and depth are	Greater in inundated areas compared to non-inundated areas; but	na	na	na	na	na

					major driver of CH4 emissions; emissions greater when water levels falling; inundated areas switched from sources to sinks	areas could switch from sources to sinks					
18. Hernadez & Mitsch 2006	Defore, during, after hydraulic pulse	Ohio	silty	9 years	No or Uncertain: Fluctuating water resulted in greater N2O emissions		Greater plots experiencing fluctuating water levels; When plants were present in standing water, emissions increased	na	na	na	na
19. Song et al. 2010	Before, during, after hydraulic pulse	Ohio	silty	14-15 years	No or Uncertain: Fluctuating water resulted in greater N2O emissions		Greater in plots experiencing reflooding compared to those experiencing dry down		na	Did not vary by treatment	na
20. Zou et al. 2022	Synthesis	variable	variable	NA	Uncertain: Level of water compared to water table, along with air temperature can drive GHG flux balance	Decreases as water level drops below surface level	Increases as water level drops below surface level		na	na	na

Table 11 U.S. research studies identified by report authors that examine impacts of the micro-topography restoration component of wetland restoration on ecosystem carbon storage and GHG exchange.

Study Description					Carbon and Greenhouse Gas Impact						
Study	**Study type**	**Location**	**Soil type**	**Time since restor-ation**	**Climate Mitigation Benefits? Yes/No, Mixed, or Uncertain**	**Carbon storage: Soils**		**Carbon storage: Plants**			
						TOC	SOC	Cover	Diversity	Richness	Composition
1. Moser et al. 2009	Intact vs restored (disked & undisked)	Virginia	silty	1 & 5 yrs	No: No evidence that disking affects TOC	Lower in restored than intact wetlands; No difference between disked and undisked	na	na	na		na
2. Moser et al. 2007	Intact vs restored (disked & undisked)	Virginia	silty	1 & 5 yrs	No direct evidence: Plant studies focused on links between microtopography & plant metrics, but did not explicitly link these to carbon sequestration	na		Restored wetlands (undisked & disked) = more cover than intact wetlands	Higher in disked than non-disked or intact		Undisked & disked sites had same assemblages; Disking prevents dominance of generalist species
3. Ahn & Dee 2011	Disked vs undisked	Virginia	na	2 yrs				Higher in disked plots	No treatment effect		No treatment effect
4. Rossell et al. 2009	Ridge vs. Depression in restored wetland	North Carolina	silty	6 yrs	Yes: Soil results indicate relationship between spatial variability of soil	na	Significantly more in lower areas	na	na	Greatest in ridge plots; increased by MT	na

5. Sleeper & Ficklin 2016	Hummock vs. Swale vs. Flat in restored wetland	Arkansas	clayey	11-12 yrs	carbon & intact topography in wetlands	Greater in flat & hummock than swale	na	na	na	na	na

Appendix B: Canadian Prairie Pothole Region References

Badiou, Pascal, Rhonda McDougal, Dan Pennock, and Bob Clark. 2011. "Greenhouse gas emissions and carbon sequestration potential in restored wetlands of the Canadian prairie pothole region." *Wetlands Ecology and Management* 19:237-256.

Badiou, Pascal, Bryan Page, and Lisette Ross. 2019. "A comparison of water quality and greenhouse gas emissions in constructed wetlands and conventional retention basins with and without submerged macrophyte management for storm water regulation." *Ecological Engineering* 127:292-301.

Bedard-Haughn, A, F Jongbloed, J Akkerman, A Uijl, E De Jong, T Yates, and D Pennock. 2006. "The effects of erosional and management history on soil organic carbon stores in ephemeral wetlands of hummocky agricultural landscapes." *Geoderma* 135:296-306.

Bortolotti, Lauren E, Vincent L St. Louis, Rolf D Vinebrooke, and Alexander P Wolfe. 2016. "Net ecosystem production and carbon greenhouse gas fluxes in three prairie wetlands." *Ecosystems* 19:411-425.

Creed, Irena F, Johnston Miller, David Aldred, Jennifer K Adams, Salvatore Spitale, and Rick A Bourbonniere. 2013. "Hydrologic profiling for greenhouse gas effluxes from natural grasslands in the prairie pothole region of Canada." *Journal of Geophysical Research: Biogeosciences* 118 (2):680-697.

Appendix C: Additional studies likely examining the relationships between wetland restoration and gas fluxes but were not reviewed

Altor, Anne E, and William J Mitsch. 2006. "Methane flux from created riparian marshes: relationship to intermittent versus continuous inundation and emergent macrophytes." *Ecological Engineering* 28 (3):224-234.

Andrews, Sarah E, R Schultz, Serita D Frey, V Bouchard, R Varner, and Mark J Ducey. 2013. "Plant community structure mediates potential methane production and potential iron reduction in wetland mesocosms." *Ecosphere* 4 (4):1-17.

Chamberlain, Samuel D, Peter M Groffman, Elizabeth H Boughton, Nuria Gomez-Casanovas, Evan H DeLucia, Carl J Bernacchi, and Jed P Sparks. 2017. "The impact of water management practices on subtropical pasture methane emissions and ecosystem service payments." *Ecological Applications* 27 (4):1199-1209.

Doherty, Kevin E, David W Howerter, James H Devries, and Johann Walker. 2018. "Prairie Pothole Region of North America 53." *The wetland book. Springer, Dordrecht. https://doi. org/10.1007/978-94-007-4001-3_15*.

Flanagan, Lawrence B, Dylan J Nikkel, Lauren M Scherloski, Rachel E Tkach, Kristian M Smits, L Brent Selinger, and Stewart B Rood. 2021. "Multiple processes contribute to methane emission in a riparian cottonwood forest ecosystem." *New Phytologist* 229 (4):1970-1982.

Gilson, Emily Rose. "Controls on Diffusive Fluxes of Methane and Carbon Dioxide in a Wetland in the California Delta."

Jensen, Sydney A, Jackie R Webb, Gavin L Simpson, Helen M Baulch, Peter R Leavitt, and Kerri Finlay. 2023. "Differential controls of greenhouse gas (CO2, CH4, and N2O) concentrations in natural and constructed agricultural waterbodies on the Northern Great Plains." *Journal of Geophysical Research: Biogeosciences* 128 (4):e2022JG007261.

Peacock, Mike, Clare Lawson, David Gowing, and Vincent Gauci. 2024. "Water table depth and plant species determine the direction and magnitude of methane fluxes in floodplain meadow soils." *Ecology and Evolution* 14 (3):e11147.

Sha, Chenyan, William J Mitsch, Ülo Mander, Jianjian Lu, Jackie Batson, Li Zhang, and Wenshan He. 2011. "Methane emissions from freshwater riverine wetlands." *Ecological Engineering* 37 (1):16-24.

Shindell, Drew, Pankaj Sadavarte, Ilse Aben, Tomás de Oliveira Bredariol, Gabrielle Dreyfus, Lena Höglund-Isaksson, Benjamin Poulter, Marielle Saunois, Gavin A Schmidt, and Sophie Szopa. 2024. "The methane imperative." *Frontiers in Science* 2:1349770.

van Kruistum, Henri, Paul LE Bodelier, Adrian Ho, Marion Meima-Franke, and Annelies J Veraart. 2018. "Resistance and recovery of methane-oxidizing communities depends on stress regime and history; a microcosm study." *Frontiers in Microbiology* 9:1714.