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Vegetation, Water Infiltration, and Soil Carbon Responses to Adaptive Multi-Paddock and Conventional Grazing in Northern Great Plains, USA, Ranches

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ABSTRACT

Adaptive Multi-Paddock (AMP) grazing involves alternating short-duration, high-intensity grazing with extended recovery periods. It offers the potential to regenerate degraded rangelands while enhancing productivity. We compared vegetation, soil carbon, and water infiltration responses between nine matched pairs of ranches in North and South Dakota, applying either conventional continuous grazing (CG) practiced for > 30 years or AMP grazing for 10–30 years at higher stocking rates. AMP grazing enhanced several ecosystem functions relative to paired CG ranches, including plant cover (35.7% vs. 13.9%), plant height (264% taller), and standing crop biomass (by 31% within plots and 64% at the ranch scale). Water infiltration rates and soils vary geographically, while the effects of grazing depend on location. Controlling for these interactions, AMP grazing reduced soil N by 45.4% and soil organic carbon by ~10% but increased total soil carbon by 19.7%. Vegetation and soil characteristics fully distinguish CG from AMP-grazed ranches, demonstrating consistent ecosystem-level changes. Both grazing systems retain many non-native cool-season grasses, suggesting that full recovery will take decades. We conclude that AMP grazing enhances ecological functions and ranch productivity, though fuller recovery of native species will require more time and perhaps active restoration efforts. AMP grazing provides a viable strategy to intensify rangeland management while addressing concerns regarding plant cover, diversity, food productivity, and possibly climate change.

1 | Introduction

Ruminant grazers have coevolved with grasses and grazing ecosystems as grasslands expanded over the last 40 million years to encompass ~40% of global land area (Frank et al. 1998). These ecosystems are linked to parallel global increases in carbon-rich soils in these grassland regions (Retallack 2013). However, in most rangelands, free-ranging wild herbivores have been replaced

by fenced-in livestock, often degrading vegetation and soils (Milchunas and Lauenroth 1993; Teague et al. 2011) and reducing productivity, biodiversity, and ecosystem resilience (Archer and Smeins 1991; West 1993; Knopf 1994; Frank et al. 1998). Overgrazing thus represents a major impediment to sustaining viable and functional grassland ecosystems and agricultural productivity (Wuerthner and Matteson 2002).

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Rangelands comprise the largest proportion of lands able to produce food, fibre, and fuel (Daily 1997; Kimble et al. 2007). However, our ability to sustain the productivity of these ecosystems hinges on balancing the needs of humans and our domestic livestock (95% of terrestrial mammal biomass, Ritchie and Spooner 2025) against the needs of wild grazing herbivores and other biota sustaining grassland ecosystems. Ecologists measure many variables linked to ecosystem services, including plant community composition, structure, diversity, and productivity, as well as soil health variables such as carbon, nutrients, physical stability, and water infiltration. To further characterize the biological integrity of rangelands, some also measure water and nutrient cycling, animal habitats and biodiversity, and air quality.

In the Northern Great Plains (NGP) of North America, remaining grasslands are recurrently tilled (Wilcove et al. 1986) and overgrazed, facilitating invasions of non-native and woody plant species (Apfelbaum and Haney 2010; Ryan 1986). Grassland declines from conversions to row crops, including growing corn for ethanol, amount to ~40,000 ha/year annually since 2010 (Lark et al. 2015). Conversions for ethanol production, initially justified to reduce greenhouse emissions, actually increase emissions and consequent impacts (Searchinger et al. 2008; Grunwald 2008). Wildfire suppression, overgrazing and consequent runoff and soil erosion augment invasions of non-native plant and animal species, combining to seriously threaten western grassland native biodiversity (Apfelbaum and Haney 2010; Saab and Powell 2005; Samson et al. 2004; Waters 2019; Chessier et al. 2019; Rosenberg et al. 2019), including grassland birds (Wuerthner and Matteson 2002).

Livestock grazing involves livestock, vegetation, soil, water, and meteorology dynamics driving human decisions acting to affect ecosystem structure and functions (Conant et al. 2017; Godde et al. 2020; Hewins et al. 2018). These effects vary according to how, and how frequently, livestock are grazed, that is, their seasonality, duration, stocking density, the animal mass of grazers, and vegetation utilization rates. Grazing impacts also depend on paddock conditions, the type of grazing implemented, the season of grazing, and how long each paddock is allowed to recover after grazing. Grazing types are often categorized as continuous grazing (CG), rotational grazing (RG), or adaptive multi-paddock (AMP) grazing (Becker et al. 2017; Fynn et al. 2017; Teague et al. 2013; Teague and Barnes 2017). CG and RG have well documented impacts on vegetation composition and structure (Souther et al. 2019; Su et al. 2017), plant productivity and biomass (Biondini et al. 1998; Hillenbrand et al. 2019; Su et al. 2017), root productivity (Hao and He 2019), decomposition and soil microbial community (Kooch et al. 2020; Wang et al. 2018, 2020; Xun et al. 2018), hydrological response and soil carbon and mineral cycles (Abdalla et al. 2018; Conant et al. 2017; Godde et al. 2020; Hao and He 2019; Hewins et al. 2018; Lu et al. 2017; Ritchie 2020; Wagle and Gowda 2018). Here, however, we compare CG with AMP grazing on ranches in the NGP of the United States.

Constant grazing pressure of livestock on preferred areas and plants, with little time to recover after grazing, steadily degrades soils and vegetation, especially under heavy continuous grazing (Fuls 1992; Müller et al. 2014). Bare ground and often less desirable short-grass, cool-season C₃ grass, and annual forbs

increase while standing biomass and nutritious and desirable grass and forb species decline. In contrast, AMP grazing shows promise for maintaining or improving soil conditions, vegetation cover and diversity, and economic viability (Müller et al. 2014).

Here, we assess the effects of AMP and CG grazing management on rangeland vegetation (plant species richness, diversity, dominance, and cover) using paired ‘across the fence’ comparisons between conditions on adjacent ranches employing either AMP or CG grazing. We assess the effects of these grazing practices on native and non-native plant cover, soil conditions (bulk density, water infiltration, etc.) and plot productivity (above-ground biomass). Forthcoming papers will further assess the effects of CG and AMP grazing on soil microbiology and genomics, soil carbon dynamics/health, hydrology (water infiltration), breeding birds, and greenhouse gas emissions. Our focus here is on vegetation and hydrology.

2 | Methods

2.1 | Experimental Design

To avoid the challenges of conducting controlled, replicated evaluations of many environmental variables reflecting variability in landscape, land-use history, and soil conditions, we used a paired design to control for many of these variables by comparing outcomes on adjacent or nearby ranches that employed either CG or AMP grazing practices. This avoids difficulties involved in comparing outcomes on commercial-scale ranches where efforts to control for these factors and year-to-year differences in management have been misleading (Teague and Barnes 2017). We biophysically stratified these ranches to include pairs reflecting regional variation in soil texture, slope, aspect, slope position, depth to bedrock, depth to groundwater table, land use, and grazing practice history while minimizing these differences within pairs using ‘paired soil catenae’ on AMP and neighbouring CG ranches. We generally follow methods in a prior grazing study in the southeastern U.S. (Apfelbaum et al. 2022). Both studies sought to generate findings relevant to ranchers operating working ranches in heterogeneous landscapes that experience variable weather conditions and commodity markets. This research was designed to infer the practical outcomes of actual grazing operations and support later re-measurements for longer-term studies (Hargrove and Pickering 1992; Teague et al. 2011). Studies that capitalize on adaptive grazing treatments are especially germane for improving ranching outcomes in the NGP.

We sought to identify adjacent or nearby pairs of ranches employing either AMP or continuous grazing to ensure that we could: (i) address questions at commercial ranch scales; (ii) use whole-system framing to integrate component science elements; (iii) incorporate pro-active management to achieve desired rancher goals under changing circumstances; (iv) identify emergent ranching issues to inform us of any unintended consequences; and (v) provide useful information for ranchers, land managers and scientists (Teague et al. 2013; Van der Ploeg et al. 2006; Hargrove and Pickering 1992; Teague et al. 2011).

We succeeded in securing across-the-fence paired ranches with eight of the nine ranch pairs sharing a common fenceline, and the treatment differences being separated by < 0.3 km. Ranch

Pair 6 was separated by ~6 km. This success minimized effects of potential confounding variables—including land use history, grazing regime history, and the socioeconomic drivers underlying management decisions—while simultaneously constraining biophysical variability (soils, topography, microclimate) to the smallest practical spatial scale. The ranches we used were carefully vetted and chosen using the following multi-stage process.

2.2 | Ranch Screening and Selection Process

We applied an ‘across-the-fence’ framework (Teague et al. 2011; Apfelbaum et al. 2022) to compare outcomes following ‘best in class’ applications of AMP grazing and best in class conventional CG on adjacent pairs of ranches.

We obtained a list of potential AMP-managed ranches in North and South Dakota from regional Natural Resource Conservation Service (USDA-NRCS) agency technical staff, grazing consultants, and ranching organizations (e.g., Grassfed Exchange, SD Grazing Association, etc.). For background, we then held meetings and presented information to grazing associations and individual ranch families that included two videos (‘Soil Carbon Cowboys’ video series by film producer P. Byck). To solicit AMP ranchers, we then followed up with phone calls and an online survey, beginning a screening process to document details of each ranch’s management history over the previous 10+ years. We collected information on cattle stocking rates; animal mass at each stocking density; their history of planting, fertilization, liming, and mowing; their use of herbicides and worming biocides; their cropping system including the size and number of summer- and winter-grazed paddocks, the number and composition of herds, the frequency at which animals were moved (e.g., multiple times per day, once a day, once a week, once a month, etc.), and how long paddocks were allowed to recover. We also queried local experts, grazing associations, and each AMP rancher to identify a neighbouring well-managed CG ranch. In choosing these pairs, we sought to match soils, slope, aspect, and land use history, allowing us to isolate AMP grazing (on a former CG managed ranch) as the treatment of interest (Table 1). Following telephone interviews of ranchers by R. Teague and A. Williams, we met with the AMP ranchers to confirm that ranches met our AMP criteria and to identify and screen potential CG neighbour ranchers. Field visits in Spring 2018 allowed ecologists R. Teague, the authors, and others (e.g., South Dakota Grazing Assn leader and educators and D. Oswald, NRCS grazing specialist, Bismarck, ND) to ask follow-up questions to confirm we understood grazing operational details, land-use history, and rancher willingness to share socioeconomic, production, and land management details and data. During the ranch visit, the ecological team affirmed comparable biophysical accuracy of soil maps, the presence of shared primary soil catenas between AMP and CG ranches, and the ranch scale was also evaluated to ensure that randomized measurements of all variables to be measured could be spatially accommodated. Matching soil descriptions and type mapping were confirmed by field soil sampling.

2.3 | Study Ranches

After a year of screening and three separate site visits, nine pairs of ranches using AMP and neighbouring CG grazing were

selected in three regions of North and South Dakota (Figure 1, Table 1) to evaluate grazing effects. These fell into three groups of paired ranches across the region with comparable biophysical conditions and known land use histories.

2.4 | Study Layout

We used detailed GIS biophysical mapping over the study area to generate our replicated, nested design. The pairing evaluated each ranch pair to ensure our statistical design matched for soil conditions, vegetation conditions, and land use histories prior to the AMP conversions to be met.

2.5 | Vegetation Sampling

We selected two shared dominant soil catenae (a sequence of soil down a slope created by the same meteorological process and geological parent material) on each of the AMP and conventional grazing (CG) ranches for vegetation and soil sampling. Within each catena, we randomly established thirty-six 1 m² quadrats across the representative paired slope positions and aspects. We excluded areas with bedrock ridge tops, riparian corridors, water bodies, wetlands, cliffs (often fenced and ungrazed), and depositional zones at slope toes to avoid recent erosion influence.

We sampled herbaceous vegetation and biomass between 28 May and 10 June 2023, when most species were actively growing and easily identified. Each quadrat was geolocated using sub-metre GPS and sampled for vegetation composition, soil carbon, and soil genomics (not reported here). We recorded: (a) the presence, % cover, and frequency of all plant species present; (b) foliage height (excluding inflorescences) at the centre of each 1 m² plot; (c) the % cover of bare soil, fine litter (dead stems), coarse litter (> 4 cm in any dimension), rock, and cow pats; and (d) the cover of all shrubs < 1 m tall and < 4 cm DBH. We also grouped plant species by functional type: annual forbs, perennial forbs, annual and perennial C₃ and C₄ grasses.

We estimated abundance for each herbaceous species as the percentage of quadrats in which it occurred (frequency) and by the percentage coverage (cover) of each quadrat. We averaged cover and frequency values across all quadrats for each ranch. We estimated species richness at two scales: total per ranch and the mean number of species per quadrat. To assess species dominance, we summed relative frequency and relative cover to compute an Importance Value (IV) scaled to 200% per ranch (Apfelbaum and Haney 2010). Plant taxonomy follows Gleason and Cronquist (1991).

2.6 | Vegetation Mapping and Biomass Estimation

We mapped community composition and standing crop biomass using remotely sensed data, specifically high-resolution, cloud-free multispectral and multi-temporal aerial imagery and Sentinel-2 satellite data. We used these data to classify vegetation types following Hillenbrand et al. (2019), then used ground sampling to document how these link to species composition, litter cover, bare soil, alkali crusts, bedrock

TABLE 1 | Characteristics of the final selected nine paired ranches.

| Ranch number | Ranch pair | Farm location | Grazing practice | Average precipitation (in) | Prairie sod intact | Previous dryland farmed | Live stock | Average # of sub-paddock | Avg sub-paddock size (acres) | Total paddock area (acres) | Stocking mass (lbs/acre) | Grazing period (days) | Recovery period (days) | Years of current management | Native non-native | Land use history |
|--------------|------------|---------------|------------------|----------------------------|--------------------|-------------------------|------------|--------------------------|------------------------------|----------------------------|--------------------------|-----------------------|------------------------|-----------------------------|-------------------|--|
| 1 | 1 | Pukwana, SO | AMP | 21 | N | Yes | Cattle | 30 | 10 | 295 | 30 | 5 | 300 | 15 | NN | woolyana Tarmea, replanted, (DLF, R) DLF, R |
| 2 | 1 | Pukwana, SD | CG | 21 | N | y | Cattle | 3 | 40 | 120 | 10 | 50 | 40 | 20 | NN | DLF, R |
| 3 | 2 | Highmore, SO | AMP | 21.21 | N | y | Cattle | 100 | 80 | 8000 | 30 | 5 | 300 | 38 | NN | DLF, R |
| 4 | 2 | Highmore, SO | CG | 21.21 | y | No | Cattle | 30 | 221 | 6643 | 10 | 30 | 300 | 30 | N | (NP) |
| 5 | 3 | Ashley, ND | AMP | 20.74 | y | N | Cattle | 81 | 40 | 3220 | 5 | 4 | 300 | 13 | N | NP |
| 6 | 3 | Ashley, NO | CG | 20.74 | Mixed | Mixed | Cattle | 4 | 333 | 1330 | 15 | 30 | 90 | 30 | N | NP + DLF, R to Brome NP + OLF, R Warm season |
| 7 | 4 | Napoleon, ND | AMP | 20 | M | M | Cattle | 55 | 60 | 3300 | 20 | 2 | 300 | 35 | N | NP + OLF, R Warm season |
| 8 | 4 | Napoleon, ND | CG | 20 | y | N | Cattle | 2 | 148 | 296 | 10 | 40 | 300 | 40 | N | NP |
| 9 | 5 | Wing, NO | AMP | 17 | y | N | Cattle | 35 | 40 | 1408 | 8 | 7 | 365 | 15 | N | NP + DLF cover crops |
| 10 | 5 | Wing, NO | CG | 17 | y | N | Cattle | 1 | 638 | 638 | 5 | 60 | 300 | 35 | N | NP |
| 11 | 6 | Bismarck, ND | AMP | 16.8 | M | M | Cattle | 200 | 23 | 4500 | 36 | 2 | 300 | 16 | NN | DLF, R |

(Continues)

TABLE 1 | (Continued)

| Ranch number | Ranch pair | Farm location | Grazing practice | Average precipitation (in) | Prairie sod intact | Previous dryland farmed | Live stock | Average # of sub-paddock size (acres) | Total paddock area (acres) | Stocking mass (lbs/a-c•1000) | Grazing period (days) | Recovery period (days) | Years of current management | Native or non-native history | Land use |
|--------------|------------|----------------|------------------|----------------------------|--------------------|-------------------------|------------|---------------------------------------|----------------------------|------------------------------|-----------------------|------------------------|-----------------------------|------------------------------|------------------|
| 12 | 6 | Bismarck, NO | CG | 16.8 | N | y | Cattle | 7 | 176 | 10 | 90 | 300 | 30 | NN | DLF, R |
| 13 | 7 | Glen Ullin, NO | AMP | 14.88 | y | N | Cattle | 80 | 4795 | 30 | 2 | 300 | 15 | NN | NP |
| 14 | 7 | Glen Ullin, NO | CG | 14.88 | y | N | Cattle | 9 | 242 | 10 | 30 | 300 | 30 | NN | NP |
| 15 | 8 | Faith, SO | AMP | 16.4 | M | M | Cattle | 27 | 142 | 35 | 4 | 300 | 20 | NN | DLF, R Cwheat |
| 16 | 8 | Faith, SO | CG | 16.4 | N | y | Cattle | 9 | 121 | 15 | 20 | 300 | 15 | NN | DLF, R Cwheat |
| 17 | 9 | Quinn, SO | AMP | 17.01 | N | y | Cattle | 60 | 32 | 25 | 1 | 300 | 15 | NN | DLF, R Cwheat |
| 18 | 9 | Quinn, SO | CG | 17.01 | N | y | Cattle | 23 | 173 | 5 | 3 | 300 | 15 | NN | DLF, R Cwheat |

Note: Years in current CG management reflect the existing operator tenure. However, CG grazing was confirmed to have been continued for decades prior. The documented acreages where dryland farming occurred were avoided by this research.

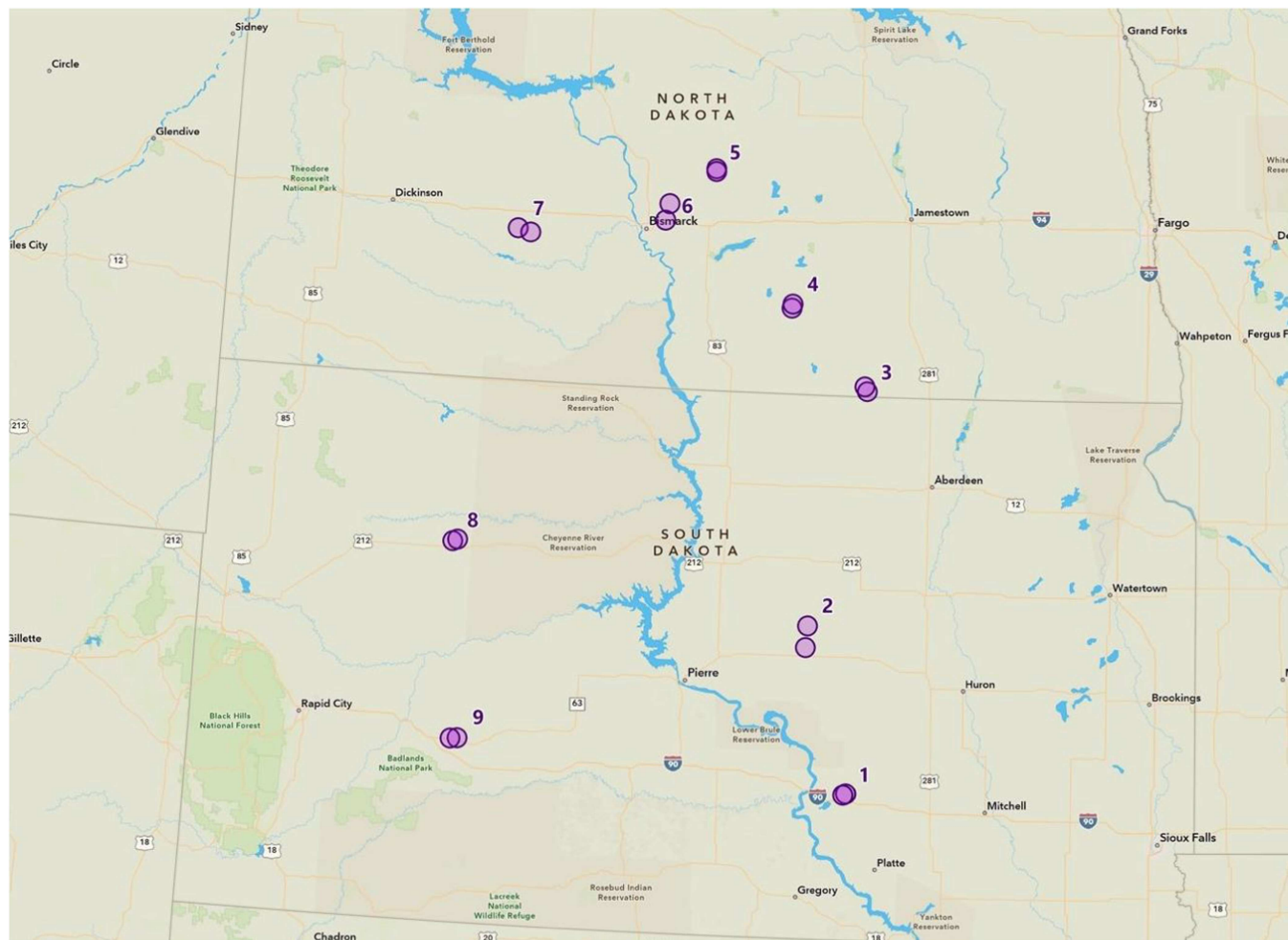


FIGURE 1 | Map showing locations of the nine pairs of Adaptive Multi-Paddock (AMP) and Continuous Grazed (CG) Ranches. These were selected to represent three regions across North and South Dakota, USA, that differ in soils, meteorology, grazing history and existing and historic ecological conditions.

exposures, and soil saturation. We calibrated spectral signatures using GPS-referenced 1 m^2 quadrats, visually confirming stands of dominant native, planted forage, and invasive species.

We collected biomass samples in eight $3 \times 3\text{ m}$ plots, with each located near a randomly selected 1 m^2 vegetation quadrat on each ranch. These provided an area adequate for harvesting plant material and, once georeferenced, large enough to match with remotely sensed data (Wang et al. 2021). After staking a GPS-located wire frame to the ground, we measured foliage height at all four corners and the plot centre. We clipped all aboveground biomass to 2.5 cm (using a battery-powered hedge trimmer), double-bagged these samples, and air-dried them at $\sim 35^\circ\text{C}$ to a consistent dry weight.

We used these foliage height and biomass data to extrapolate local biomass estimates across the entirety of each ranch. We used these inferences to compare estimated biomass between CG and AMP ranches at both the plot and whole-ranch scale. The ranch-scale comparisons rely on mapped biomass classes reflecting the proportions found within each biomass zone, normalized to an area of 405 ha (1000 acres).

2.7 | Water Infiltration

We assessed grazing treatment effects on soil water infiltration using SATURO dual-head infiltrometers (<https://www.metergroup.com/environment/products/saturo/>) between mid-June and mid-July 2023. These measure field-saturated hydraulic conductivity (Kfs). At each sampled area, we randomly selected eight of the 1 m^2 vegetation quadrats as plots to test infiltration. Tests followed the manufacturer's protocol, adjusting pre-soak time based on antecedent moisture and soil texture. Once calibrated, the infiltrometer maintained a constant water level and head pressure. Test duration ranged from 90 min to over 3 h depending on site characteristics. We used paired Student's *t*-tests to compare infiltration rates between ranch pairs.

2.8 | Soil Carbon Measurements

Soil sampling followed The Regenerative Standard's 'Soil Carbon Quantification Method' (Applied Ecological Institute 2025), a method designed and approved for the voluntary carbon marketplace. At the two selected soil catenae in each ranch, the 36 randomly located 1 m^2 vegetation quadrats were the same sampled points for soil carbon, and a subset was also sampled for infiltration as described above. At each georeferenced

random point, a 1-m depth soil core was extracted with a Giddings¹ hydraulic sampler mounted on a Polaris Ranger 6 × 6. Once extracted, each core sample was logged; length was measured, GPS location and cardinal direction photographs were taken. The 2" (~5 cm) inside diameter soil cores were extracted in plastic sleeves, which were capped at both ends, labelled with a bar code, stored in a large heavy-duty insulated (cooled) wood crate, and shipped to Cquester Laboratory for all soil carbon and soil health (Fort Collins, CO) analyses. At the laboratory, the samples were divided into depth increments (e.g., topsoil – 0 cm to the bottom of the 'A' horizon, bottom of the topsoil to 30, 30–50, 50–100 cm) which were separated, homogenized, and sieved to remove, dried and used to quantify materials such as roots, rocks, and litter. An intact subsampled standard length (and soil volume) of each core depth increment was dried and measured for gravimetric water, and bulk density was determined by weighing the standard volume, subtracting the mass of the removed materials for each depth increment, after oven drying to bone dryness at 60°C. Soils from each depth increment were analyzed for total soil carbon, and organic and inorganic carbon levels using Lecor combustion analyzers (Mosier et al. 2021).

2.9 | Statistical Analyses

We first examined distributions for continuously distributed environmental and vegetation predictor and response variables to identify conspicuous outliers and decide whether transformations were needed to meet model assumptions (e.g., homogeneous variance and normal residuals). The few outliers detected were checked against the original data sheets and corrected for data-entry errors. We also examined bivariate relationships between relevant pairs of variables to check linearity of effects and homoscedasticity. Most variables met the assumptions of linear models, but we log-transformed cover values because they were log-normally distributed.

We applied simple parametric and multivariate statistical methods to analyze variation among plots and between CG and AMP grazed ranches at both the 1 m² quadrat and whole-ranch scales. We analyzed fine-scale data (e.g., species frequency and cover) at the 1 m² quadrat scale and broader differences in environmental conditions and community composition at the whole-ranch scale. These analyses employed linear and non-linear regression and multivariate general linear models (GLM) or general linear mixed models (GLMM), incorporating interactions between interacting variables when these were significant. Regressions allowed us to evaluate and calibrate spatial trends measured from aerial photography and satellite imagery with on-ground measures of soil, vegetation, biomass, infiltration, land cover, and plant species present. Most models evaluated how plant species, vegetative cover, water infiltration, soil, and other measured variables differed among plots and between the pairs of CG- and AMP-grazed ranches. Crossing ranch grazing types (CG vs. AMP, the fixed treatment) with ranch pairs (reflecting location, a random variable) added power for evaluating effects of the grazing treatment under regional variation in soil, climate, and vegetation conditions. Treatment effects were tested using the $p < 0.05$ criterion for significance, except for species comparisons across grazing

treatments, which used the $p < 0.10$ criterion at the ranch scale (36 quadrats).

We built a Discriminant Function Analysis (DFA) model to distinguish CG from AMP-grazed ranches. DFA computes the best linear combination of predictor variables to predict which grazing method was applied. The 13 variables we used for the DFA represent all (the complete set of) ecological response variables that were measured in this study, as described in the Methods. No variable selection or screening procedure was applied; all measured variables were included to allow the DFA to identify which combination of ecological responses most effectively discriminated between AMP and CG pastures. No potentially informative variables were excluded from the analysis.

We constructed a causal path (structural equation) model (SEM) to distinguish direct from indirect effects of precipitation. This model was used to further evaluate the full GLM model, analyzing native plant cover within 1 m² quadrats and tests of non-plant native cover, ranch pair, and the pair × grazing type interactions between non-native and native plant percent cover associated with AMP grazing. The SEM was constrained by sample size ($n = 9$ pairs) and the available measured variables. We included relevant potentially confounding variables but avoided introducing additional predictors without a strong a priori justification to minimize overfitting risk. Alternative model structures were evaluated, but complexity was limited by sample size and data availability. Despite these constraints, strong statistical signals from the SEM, aligned with ecological theory and field data, provide a foundation for future evaluation.

We then related among-ranch and between-grazing treatment variation to differences in vegetative composition and the environmental variables using two multivariate ordination methods (Principal Component Analysis, or PCA, and Multidimensional Scaling, or MDS). We focus on MDS as this method provides simpler and more interpretable results. All analyses were conducted in JMP Pro, Version 18 (JMP Statistical Discovery LLC, Cary, NC).

3 | Results

We address these study questions:

1. How has AMP grazing affected plant cover, biomass, height, physical conditions, and community composition relative to conventional grazing?
2. How has AMP grazing affected the abundance of non-native species and plant diversity?
3. How do the effects of AMP grazing scale up to whole ranches?
4. How consistent are the changes linked to AMP grazing among ranch pairs?
5. How do the effects of AMP grazing vary with respect to land use history, precipitation, and other physical and biotic characteristics?

3.1 | Effects of AMP Grazing

We compared the effects of AMP grazing relative to conditions on the nearby matched conventionally grazed ranch using GLM

¹Giddings Machine Company, Windsor, CO (www.soilssample.com).

models that tested the main effects of grazing type, ranch pair, and their interaction. Grazing type had simple additive effects on plant biomass and plant height as assessed at three locations within each ranch (first two lines, Table 2). AMP grazing increased plant biomass 3.94× over CG ranches (334.0 vs. 84.7 g/m²) and plant height 2.64-fold (34.0 vs. 12.9 cm). Neither the ranch pair nor the grazing × ranch pair interaction showed significant effects for these two key variables. In contrast, plant height in quadrats showed highly significant effects from ranch pair and the grazing × ranch pair interaction (line 3, Table 2). This high significance reflects the large sample size for quadrat-level plant heights, showing that ranch conditions affect both overall plant growth (the main effect) and how plant heights respond to AMP grazing (the interaction). Total soil carbon averaged about 20% higher on AMP-grazed ranches, a difference that was almost significant, but variation among ranch pairs (locations) and the ranch pair × grazing type interaction had greater effects.

Many variables responded to AMP grazing via both a main effect for ranch pair (geography) and the grazing × ranch pair interaction (Table 2). Strong grazing type × ranch pair effects indicate that the effects of AMP grazing depend on geographic location. The large sample sizes for ground cover variables allowed most of these predictors to be statistically significant even when the AMP grazing effect was modest. AMP grazing appeared to reduce exposed rock cover in quadrats by 55%, a big effect. Ranch pair effects outweigh these interaction effects, indicating that ranch locations generate more variation than the effects of AMP grazing. Locations differ in climate, soils, and other factors, with consistent patterns in some variables. For example, although water infiltration rates did not differ between grazing types overall (AMP grazing 3.7 cm/h vs. CG 3.8 cm/h), both the ranch pair and its interaction with grazing treatment were significant (Table 2). AMP grazing tended to reduce infiltration at the drier end of the spectrum (ranches

with high pair numbers on the right, Figure 2a). Infiltration rates tended to increase as standing plant biomass and grass cover increased (Figure 2b,c). Rates of infiltration showed no relationship to variation among ranches in the number or percentage of native species, native plant cover, total plant cover, or species richness.

3.2 | How Are Grazing, Non-Native Species Abundance, and Plant Diversity Related?

Plant species richness per 1 m² quadrat ranged from 1 to 11 on CG ranchlands (mean: 4.53) and from 1 to 14 on AMP grazed lands (mean: 4.43), with no main effect of grazing. However, the ranch pair effect and grazing × ranch pair interaction effects were both significant ($F = 22.04^{***}$ and $F = 15.14^{***}$, respectively). This indicates that AMP grazing had different effects on plant species richness in different locations. Distributions of species richness per quadrat were normal and positively related to quadrat plant cover, as expected (Figure S1). Grazing treatment and ranch pair did not affect either plant species richness or the $\exp(H')$ estimate of overall plant diversity at the ranch level ($F = 0.01$ NS, $N = 18$). Given the large differences in plant biomass, height, and cover between AMP and CG ranches and the fact that species richness increases with the cover of native species but declines with non-native cover (Figure 3), we were surprised not to find any significant difference in richness or diversity between grazing types. Small sample size (18 ranches) may have limited our power for finding significant effects of AMP grazing and ranch pair on native and non-native plant cover and diversity (middle lines, Table 2).

Plant growth form (% cover of grass or forb) had no overall effect on species richness or $\exp(H')$ diversity (Likelihood ratio Chi-square tests, $N = 446$). However, ranch pair and grazing type interacted to significantly affect grasses versus forbs ($F = 9.37^{**}$,

TABLE 2 | Effects of AMP grazing relative to CG grazing on abiotic and biotic conditions on nine pairs of ranches.

| Variable | Sample size | AMP relative to CG | Grazing effect F | Ranch pair F | Grazing × pair F |
|---------------------------------------|-------------|--------------------|--------------------|----------------|--------------------|
| Log (Biomass) | 54 | +394% | 8.93** | NS | NS |
| Mean veg height | 54 | +264% | 5.2* | NS | NS |
| Quadrat height | 644 | +21.5% | 51.9*** | 10.96*** | 33.6*** |
| Soil infiltration | 122 | −23% | NS | 3.88** | 3.10** |
| Log (Soil total carbon) | 2572 | +19.7% | 3.10 AS | 76.5*** | 3.56*** |
| Log (Non-native plant cover) | 54 | −19% | NS | NS | NS |
| Log (Native plant cover) | 54 | −23% | NS | 2.18 AS | NS |
| Plant species richness | 18 | +9.5% | NS | — | — |
| Plant Species diversity ($\exp H'$) | 18 | +8.5% | NS | — | — |
| Ground cover | | | | | |
| Fine litter | 647 | +4.8% | 14.2*** | 41.1*** | 8.27*** |
| Log (Bare soil) | 648 | −18.1% | 4.36* | 62.1*** | 7.61*** |
| Log (Cow pies) | 648 | −2.3% | 1.91 | 3.53*** | 2.83** |
| Log (Rock) | 648 | −55% | 4.87* | 2.81** | 1.73 AS |

Note: The table shows the contributions of the two grazing treatments, the ranch pairs, and their interaction to variation in each variable across the nine matched pairs of ranches. Results compare the relative least-square means of AMP-grazed fields relative to CG-grazed fields (always on a linear scale for AMP relative to the CG column). F -values and significance levels from GLM analyses of the variables treating the ranch pair as a random variable yielded the same qualitative results. Significance levels: *** $p < 0.001$; ** $p < 0.01$; * $p < 0.05$; AS $0.05 < p < 0.10$.

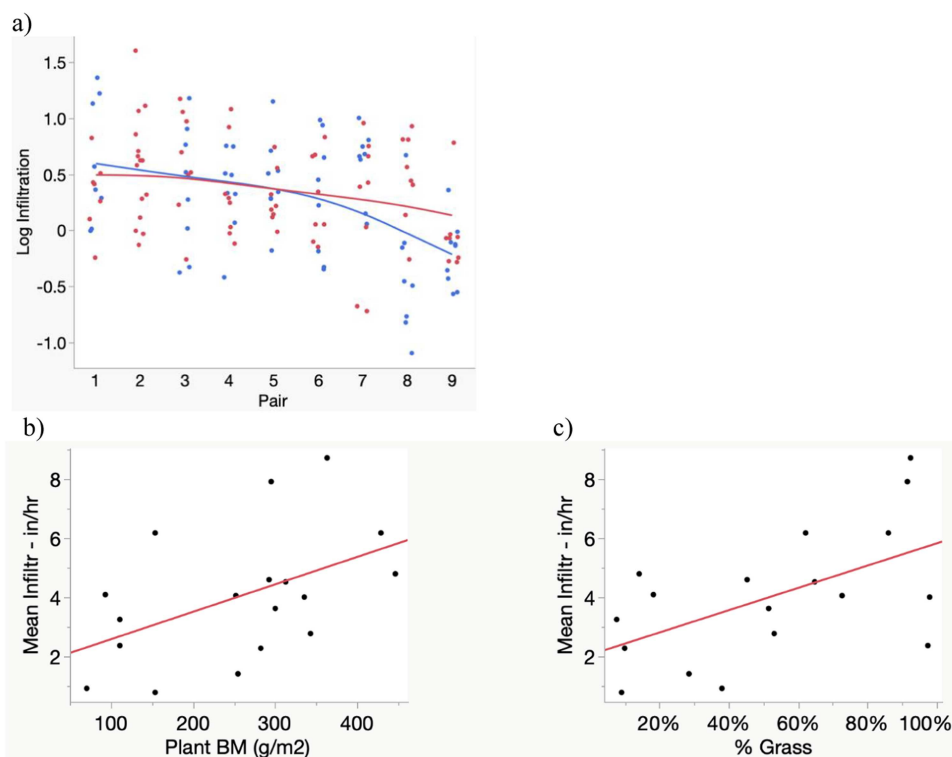


FIGURE 2 | Rates of soil infiltration vary across ranch pairs and in response to plant cover. (a) Rates of infiltration over ranch pairs. Although grazing type had no significant main effect, the ranch pair and the grazing \times ranch pair interaction were significant (Table 2). Red points and lines indicate CG grazing, blue AMP grazing. (b) Rates of water infiltration increase as plant biomass increases ($r = 0.48$). (c) Infiltration also increases with grass cover ($r = 0.56$, both $p < 0.05$).

$r^2 = 0.46$). Grass cover was higher in ranches showing no significant grazing effect, leading to some differences between means for AMP and CG ranches (Table 2, Figure 3).

Grazing type had little or no effect on C3 versus C4 plant species richness and cover, or the cover of annual, biannual, or perennial species. These variables also showed no effect of the ranch pair or its interaction with grazing. Grazing effects and grazing-ranch pair interaction effects as measured by differences in mean quadrat plant species richness did not differ between grazing treatments ($F = 0.48$, $p > 0.49$), but there were significant differences among grazing pairs ($F = 15.1$, $p > 0.0001$, $r^2 = 32\%$, $n = 647$). These location differences account for 23% of the variance in 1 m² plant species richness across the ranches.

3.3 | Effects of AMP Grazing on Whole Ranches

Plant heights accurately predict plot dry weight biomass ($r = 0.755$, Figure S2). This relationship was consistent across grazing types ($F = 0.30$, NS) and ranch pairs ($F = 0.47$, NS) with no significant interactions. We used this simple relationship to estimate above-ground biomass both within the 36 individual 1 m² quadrats and for whole-ranches (standardizing these to 1000 ac = 405 ha). We did this by estimating biomass within zones using the interpolated isopleth maps derived from remotely sensed height data (see Section 2). These clipped plots' predictions for 'landscape biomass' averaged 31% higher on the AMP grazed ranches (2.97 ± 0.59 T/ha vs. 2.27 ± 0.69 T/ha for CG ranches) and 164% higher when extrapolated to whole ranches (3533 ± 2032 tons vs. 2148 ± 2071 tons, both $p < 0.05$,

Figure 4). AMP ranches consistently had higher mean dry weight total biomass, and one had the highest per-hectare dry weight yield measured (3.54 T/ha). CG ranches averaged less biomass, including the lowest recorded yield (1.05 T/ha).

3.4 | What Features Distinguish AMP- From CG-Grazed Ranches?

For variables with only one observation per ranch, we could not perform conventional GLM analyses and so applied DFA to these 13 physical and biotic variables to distinguish AMP- from CG-grazed ranches. This analysis cleanly discriminated between the two grazing types, correctly assigning all 18 ranches to their correct group ($r = 0.99$, $F = -14.7$, $p < 0.001$). The canonical plot (Figure 5) shows clear separation between the CG ranches (left, blue circles) and AMP-grazed ranches (right, red circles). CG ranches had lower plant biomass and infiltration rates and more non-native species, while AMP grazed ranches had more native species and higher diversity (Exp H'). This clear separation reflects distinct outcomes emerging over the term of AMP grazing. The DFA thus shows how a simple set of biophysical measurements suffices to categorize ranches even without more detailed measurements of grazing outcomes.

3.5 | Interactions Among Grazing, Plant Cover, and Precipitation

AMP grazing increased total plant vegetation cover (summed over 1 m² quadrats) from 51.4% (± 57.0) on CG ranches to 70.5%

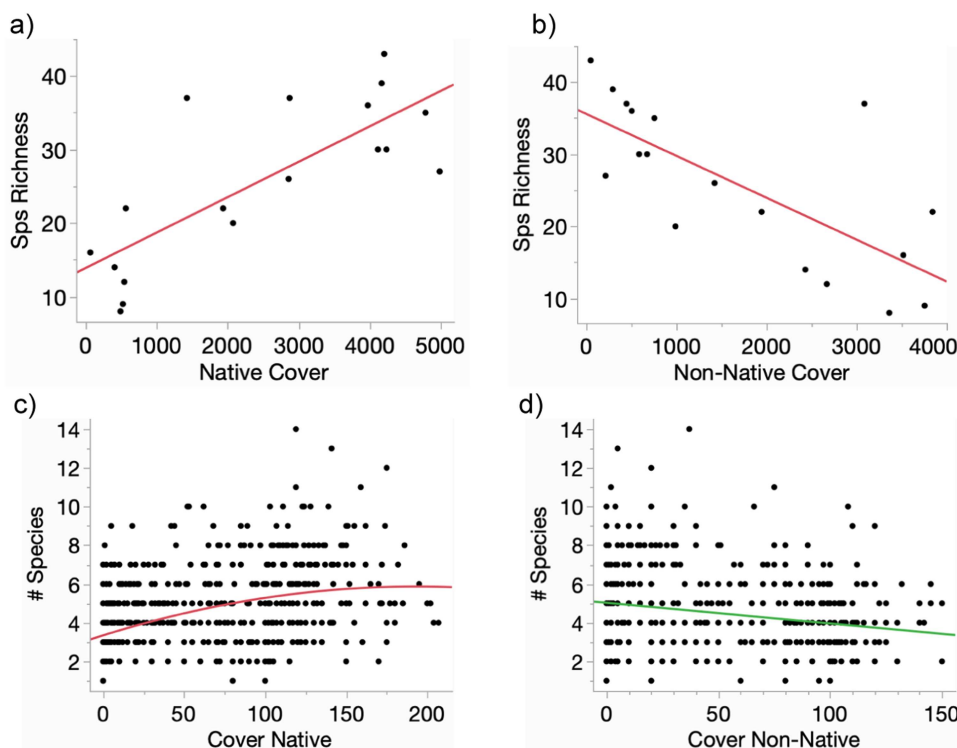


FIGURE 3 | Effect of native and non-native plant cover on species richness at two scales. (a) Whole-ranch richness increases in response to higher cover of native species, while (b) shows declines in richness as non-native cover increases ($N = 18$). Both relationships are highly significant and predict species richness better than their sum (total cover), with coefficients of determination, r^2 , of 0.60 and 0.53 compared to 0.23 for total cover. Lower panels show corresponding relationships at the 1 m^2 quadrat level ($N = 648$) where the number of species observed increases as a quadratic function of native species cover ($r^2 = 0.19$, cover $t = 11.82^{***}$ and cover $^2 t = 2.33^*$, c) and decreases as non-native cover increases ($r^2 = 0.05$, $t = -5.82^{***}$, d).

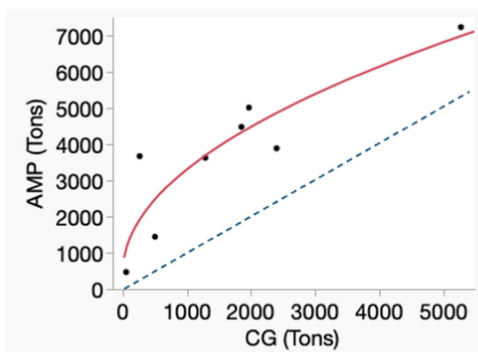


FIGURE 4 | Total above ground biomass on AMP-grazed ranches exceeds that found on 8 of the 9 nearby paired CG ranches (Faustich-Schuette pair excluded – see text). The dotted line shows a 1:1 relationship. Fitted equation: AMP Tons = $462.095 + 89.827 \cdot \text{Sqrt}(\text{CG Tons})$.

(± 51.9) and native plant cover (from $13.9 \pm 31.9\%$ to $35.7 \pm 44.8\%$) while reducing non-native cover (from $62.4 \pm 42.0\%$ to $40.2 \pm 43.7\%$). The nature of these effects, however, is complicated by how both native and non-native cover respond to precipitation (Figure 6). Ranches experiencing higher precipitation show declines in native cover (Figure 6a) and increases in non-native cover (Figure 6b). However, non-native species also compete strongly with native species, reducing their abundance (Figures 6c and S3), making it difficult to disentangle these effects (i.e., whether native species

declines on some ranches reflect precipitation effects vs. effects of high non-native cover). To address this uncertainty, we constructed a causal path SEM to distinguish direct from indirect effects of precipitation (Figure 6d). This model demonstrates that the declines in native species observed on wetter ranches do not reflect increased precipitation (as suggested by Figure 6a) but rather the indirect effect of precipitation on non-native cover, which then reduces native cover. Reinforcing this conclusion, a full GLM model analyzing native plant cover within 1 m^2 quadrats found that non-native cover, ranch pair, and the pair \times grazing type interaction all exerted very strong effects, with non-native cover reducing native cover (overall model $F = 139.4$, $p < 0.0001$, $r^2 = 0.75$). Including these effects allowed a sensitive, adjusted comparison showing that AMP grazing enhanced native cover by 55.4% (41.9 vs. 27.0%, $F = 4.92$, $p = 0.027$). Thus, AMP grazing increases native plant cover dramatically and in large part by reducing the cover of non-native species.

Despite being limited by sample size and data availability, the SEM model provided strong statistical signals from the SEM, aligned with ecological theory and field data and provided a foundation for future evaluation.

3.5.1 | Plant Species Richness and Compositional Response to Grazing Treatments

We used ANOVA to test whether intact prairie sod, a history of dry land farming, or a pasture being dominated by native or

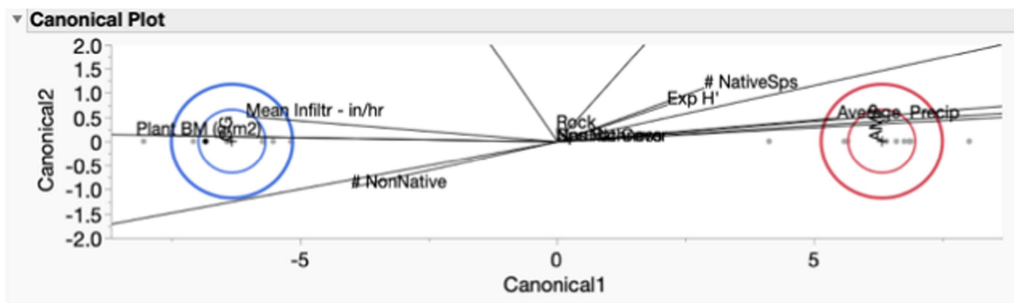


FIGURE 5 | Discriminant function analysis correctly predicts grazing treatments from biotic and abiotic characteristics across 18 ranches. The CG ranch centroid is at left (blue circles), and the AMP-grazed ranch centroid is at right (red circles). The influence of the predictor variables is shown as vectors.

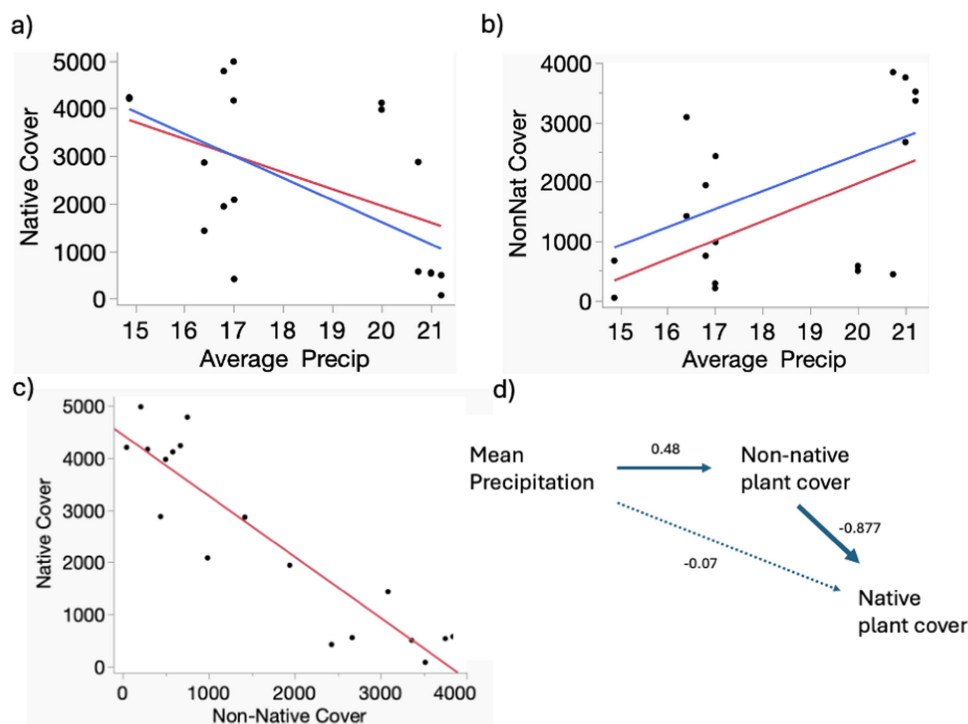


FIGURE 6 | Responses of native and non-native cover to regional increases in precipitation. Precipitation appears to reduce native cover (a) while increasing the cover of non-native plants (b). However, native and non-native species also appear to compete strongly with one another (c). A causal path analysis (d) makes clear that declines in native species abundance at ranches with more precipitation are the indirect effect of precipitation enhancing non-native cover (especially rhizomaceous cool-season grasses), which then reduces native plant cover (d). Path coefficients (standardized partial regression coefficients, or beta values) show direct effects of each variable. The indirect path causes most of the decline in native cover in response to precipitation ($0.48 \times -0.877 = -0.42$ vs. -0.07 via the direct path). Effects of precipitation on native and non-native cover are unrelated to the grazing treatment (red lines: AMP grazing, blue lines: CG grazing, interaction effects NS).

planted non-native plant species affected plant species richness. Total plant species richness reflected ranch location ($r^2 = 6.1\%$, $F = 3.6$; $p > 0.04$) with ranches in the Glen Ulin and Napoleon, ND area having higher plant species richness than ranches near Highmoor, Pukwana and Quinn, SD. This analysis shows that grazing and the presence of an intact prairie sod did not have a significant effect on species richness, yet when intact sods were present, species richness averaged 10 species higher than in ranches lacking intact prairie sod. This lack of difference in species richness between AMP and CG ranches held despite some differences in summed % total cover for shared species between ranch pairs (Table S1).

About 111 plant species occurred within the 1 m^2 quadrats ($n = 682$). Native and non-native species richness per square metre ranged from 3–36 and 3–10, respectively. Among the non-native species, 19 taxa dominated, including Kentucky blue grass (*Poa pratensis*), European brome grass (*Bromus inermis*), crested wheat grass (*Agropyron cristatum*), and intermediate wheat grass (*Agropyron intermedium*). These account for most of the non-native plant cover. Among the 89 native species recorded, wheat grass (*Agropyron dasystachum*), western wheat grass (*A. smithii*), little bluestem (*Andropogon scoparius*), moustache grass (*Bouteloua hirsuta*), and buffalo grass (*Buchloa dactyloides*) were dominant. Co-dominants included June grass

(*Koeleria cristata*), annual trefoil (*Lotus purshianus*), switch grass (*Panicum virgatum*, found only on AMP ranches), green needle grass (*Stipa viridula*), and Snowberry shrub (*Symphoricarpos occidentalis*).

4 | Discussion

Our comparison of outcomes on ranches using AMP versus Conventional Grazing (CG) practices in the NGP reveals both simple and complex responses. Ten plus years of AMP grazing (short-duration, high-intensity grazing followed by a long recovery period) generated markedly different outcomes in vegetation dynamics, soil carbon, and ecosystem functions compared to CG. These results align with and extend findings from other studies, as summarized in meta-analyses by Teague et al. (2011) and Byrnes et al. (2018). In general, these differences reflect basic concepts in disturbance and grazing ecology across diverse global rangelands (Diaz et al. 2007; Porensky et al. 2021). In short, it appears that AMP grazing that mimics historical patterns of herbivory by native herbivores can enhance multiple ecosystem functions (Fuhlendorf et al. 2017; Augustine et al. 2020).

This study makes several novel contributions to the AMP grazing literature. First, our across-fence paired design—comparing AMP and CG ranches under matched edaphic and climatic conditions—controls for confounding environmental variation more rigorously than prior observational studies. Second, our sites span the NGP, extending the continental-scale evidence base that now includes the Southern Great Plains (Teague et al. 2011; Mosier et al. 2021) and Canadian provinces (Wang et al. 2016, 2020). Third, and most critically, our results identify management behaviour—specifically, the animal mass applied per unit area—as the primary constraint on ecological recovery, rather than geography or climate per se. In the sections that follow, we integrate our findings by first linking observed ecological responses to their mechanistic drivers, then synthesizing regional variability across this expanding body of continental AMP research, and finally articulating actionable management implications for rangeland practitioners.

4.1 | Mechanistic Drivers: From Animal Impact to Ecosystem Response

A central finding of this study is that the ecological responses observed under AMP grazing can be understood through a linked chain of mechanistic drivers: animal mass intensity → hoof impact and defoliation pressure → changes in soil physical properties → altered infiltration → plant productivity feedbacks. This mechanistic framework organizes the discussion of our results below and helps explain why management intensity—not geography—appears to be the primary determinant of ecological outcomes.

Our most important finding is that AMP grazing dramatically increased the biomass (by 31%) and height (by 264%) of vegetation relative to conventional grazing, in line with results from other regions. Teague et al. (2011) reported 25%–35% higher herbaceous standing crop on AMP ranches in Texas, while Wang et al. (2016) documented 40% increases in Alberta, Canada. Our results (2.97 ± 0.59 vs. 2.27 ± 0.69 T/ha) match results by

Hillenbrand et al. (2019) for South Dakota shortgrass prairie, who observed increases of 2.5–3.5 T/ha under AMP grazing. What mechanisms drive these biomass differences? Brief but intense grazing events in AMP systems stimulate compensatory growth, as documented by McNaughton (1979, 1983) in African savannas and confirmed in North American grasslands by Williamson et al. (1989) and Frank (2005). This response is mediated by enhanced tillering following defoliation (Briske and Richards 1995), increased photosynthetic rates in remaining tissue (Nowak and Caldwell 1984), and mobilization of stored reserves (Richards and Caldwell 1985).

The extended recovery periods used in AMP systems (see Table 1) allow for deeper root development, as documented by Weaver (1968) and more recently by Wilson et al. (2018). Our findings align with Gill et al. (2017), who showed that shorter grazes increased root biomass by 30%–40% compared to CG in mixed-grass prairie. Additionally, the concentrated animal impact in AMP systems accelerates nutrient cycling as dung and urine deposition increase locally (Augustine and McNaughton 2006). These likely enhance microbial activity and nitrogen mineralization within fertility ‘hotspots’ (Schönbach et al. 2011; Frank et al. 2000). While we did not measure decomposition rates, the greater living green biomass on AMP ranches suggests higher utilization rates, and enhanced microbial activity would be expected to align with faster decomposition rates on AMP ranches. This would support findings by Liu et al. (2015), who documented 55% slower litter decomposition under CG in Inner Mongolia.

The ability of AMP ranches to maintain higher productivity despite higher stocking rates challenges conventional range management paradigms. This phenomenon, termed the ‘grazing optimization hypothesis’ by McNaughton (1979), has been demonstrated across diverse ecosystems. Savory and Parsons (1980) first proposed that properly managed high-density grazing could increase carrying capacity, while Teague et al. (2013) documented 50%–100% higher stocking rates on AMP ranches without degradation. Park et al. (2017) used modelling to demonstrate how AMP grazing can sustainably support 25%–40% more livestock. Our results support these findings and the ‘ecological intensification’ concept proposed by Garnett et al. (2013), where improved management increases production while maintaining or enhancing ecosystem services.

What mechanisms affect how carbon accumulates in soils? Reeder and Schuman (2002) documented 40% greater root carbon inputs under RG, while LeCain et al. (2002) found increased root:shoot ratios. We found increased plant biomass, height, and cover on AMP-grazed ranches. Such enhanced biomass is likely to enhance soil structure according to Six et al. (2004). Wilson et al. (2009) showed 25% better soil aggregation under RG, and Franzluebbers and Stuedemann (2010) showed that such aggregates protect soil carbon. Microbial mechanisms also play roles, with AMP grazing increasing microbial biomass carbon (Northup et al. 1999), carbon use efficiency (Kallenbach et al. 2016), and fungal:bacterial ratios (Bailey et al. 2002). The much higher levels of carbon at shallow depths mirror patterns described by Jobbágy and Jackson (2000). Follett et al. (2001) found deeper carbon storage, while Mosier et al. (2021) found a significant increase in soil organic carbon at ~0.8 m depth under AMP grazing, respectively, leading Sanderman et al. (2017) to stress how important it is to understand and measure deep soil carbon.

TABLE 3 | Results of GLMM analyses of variation in soil characteristics based on $N = 2572$ samples across all 18 ranches.

| Variable | R^2 | Effects of: | | | % of Variance due to the ranch | |
|---------------------------|-------|-------------|----------------|------------|--------------------------------|------|
| | | Grazing | AMP rel. to CG | Soil depth | Trend | Pair |
| Bulk density | 0.733 | 0.58 NS | — | 1696*** | Incr | 56.0 |
| Log total carbon | 0.643 | 2.99 AS | +19.7% | 737.3*** | Decr | 51.0 |
| Log organic carbon stocks | 0.397 | 5.19* | -10.7% | 189.2*** | Var | 40.1 |
| Log nitrogen stocks | 0.585 | 34.12*** | -45.4% | 751.4*** | Var | 46.3 |

Note: Soil samples were taken from four depths: 'A' Horizon, to 30, 30–50, and 50–100 cm. 'Trend' refers to how the values of each variable respond to increasing soil depth. Effects of AMP relative to CG grazing are shown based on least square means from the analysis back-transformed from logged values. Values shown in the table for Grazing and Soil Depth are F -values and associated significance levels: *** $p < 0.001$; * $0.01 < p < 0.05$; AS $0.05 < p < 0.1$.

We found no significant overall difference in soil infiltration rates between grazing treatments, but infiltration varied among ranch locations, and this variable interacted with the grazing treatment. Thurow (1991) also found spatial variability in infiltration, and Park et al. (2017) modelled variable infiltration responses. Teague et al. (2011) found 50%–100% higher infiltration under AMP grazing in some southeastern soils, reflecting the higher plant cover and consequent buildup of litter and organic matter in these soils. This result aligns with Blackburn's (1984) classic work on plant cover–infiltration relationships. Castellano and Valone (2007) found similar vegetation structure effects, as did Ludwig et al. (2005), within grass patches. Like Savadogo et al. (2007), who found 50%–70% more bare soil under CG grazing, we found lower bare ground and higher fine litter accumulation under AMP grazing. Waters et al. (2017) also documented litter accumulation benefits, as did Byrnes et al.'s (2018) meta-analysis of ground cover improvements. Naeth et al. (1991) and Daniel et al. (2002) proposed that litter serves to protect soil surfaces and promote infiltration, suggesting that the higher plant cover and litter we observed may promote improved soil structure and resilience.

4.2 | Plant Community Composition and Diversity Dynamics

Native and non-native plant cover were negatively correlated ($r = -0.76$), matching patterns found across rangelands, for example, DiTomaso (2000) in California annual grasslands, Prober et al. (2005) in Australian temperate grasslands, and Bakker and Wilson (2004) in northern mixed-grass prairie. Our SEM showed how non-native species directly suppress native cover, even after controlling for the effects of precipitation (Wald $Z = -7.36$, $p < 0.0001$). Even with a low ($n = 9$) sample size and limited variables that were measured, strong statistical signals from the SEM aligned with ecological theory and field data from Seabloom et al. (2003), MacDougall and Turkington (2005), and Grace et al. (2016) who demonstrated persistent competitive exclusion by exotic species even under improved management.

The persistence of invasive cool-season grasses, particularly Hungarian brome (*B. inermis* L.), and Kentucky bluegrass (*P. pratensis* L.), across both grazing systems reflects century-long selection pressures (Coupland 1961; DeKeyser et al. 2015). These species dominate our study sites, with *P. pratensis* present in 100% of quadrats and showing the highest IV (IV = 66.3) under CG management. Our hypothesis that recovery may require decades aligns with Bakker et al. (2003), who estimate 20–30 year recovery times for native plant communities and

Török et al. (2011), whose meta-analysis suggests recovery times of 10–25 years. Martin and Wilsey (2006) documented similarly slow rates of native species reestablishment. The seasonal growth advantages exploited by these invasive species, particularly early spring green-up, mirror patterns described by Schacht et al. (1996) and Hendrickson and Lund (2010). The low stocking rates in the NGP (see Table 1) in AMP and CG ranches retards the re-establishment of native species and favour persistent cool-season native grasses. The small range of differences across all study sites in livestock stocking rate, density, and/or animal mass/acre (see Table 1) and normalized for time on a pasture was not sufficient to test such fundamental relationships.

Differential C3/C4 responses between AMP and CG systems have been found in several temperate grasslands. Derner and Schuman (2007) showed similar C3 promotion under RG, while Epstein et al. (1997) documented C3/C4 shifts along grazing gradients, and Morgan et al. (2011) demonstrated how grazing timing affects functional group composition. Deliberate C3 planting and promotion of C3 grasses by CG ranchers to extend grazing seasons align with management strategies described by Roche et al. (2015) to shift from emphasizing peak season production to favouring year-round forage availability. Such efforts could yield cascading benefits for ecosystem services, but may further delay and erode the recovery of native species as documented in AMP paddocks with Hungarian brome and crested wheat grass domination.

4.3 | Soil Carbon and Nitrogen

AMP grazing reduced overall soil nitrogen stocks by 45% and soil organic carbon stocks by 11% while increasing inorganic soil C and total soil C slightly (Table 3). Soil N and C levels varied much more, however, in response to soil depth and location. These findings contrast with our previous report that AMP grazing increases organic C by 13% in soils in the Southeast (Mosier et al. 2021). Conant et al.'s (2017) meta-analysis found 0%–19% increases in soil C under improved grazing, matching McSherry and Ritchie's (2013) finding of mean increases of 14% in soil C globally. Abdalla et al. (2018) documented increases of 8%–19% in soil C in temperate grasslands. All these studies show much site-specific variation, supporting Henderson et al.'s (2015) point that grazing–carbon relationships vary with context.

The apparent contradiction: AMP grazing increased total soil carbon by ~20%, yet organic carbon and nitrogen both declined, has resulted because of an increase in total carbon, because

measured soil inorganic carbon (SIC) increased, not organic carbon. In several other studies (Apfelbaum 2020), this increase in SIC appeared to be driven mainly by high urine loads of excreted calcium oxalate from cattle and possibly by wind-deposited carbonates. In this study, we note and believe that organic carbon gains were not statistically significant at most sites due to low stocking densities/animal masses on the studied ranches in the NGP. The authors acknowledge that SIC dynamics deserve further dedicated study.

4.4 | Regional Synthesis: Continental-Scale Patterns and the Role of Management Intensity

Our study is part of an expanding body of continental-scale AMP research spanning the Southern Great Plains (Teague et al. 2011; Mosier et al. 2021), Canadian provinces (Wang et al. 2016), and—here—the NGP. Rather than merely describing geographic variation, synthesizing across these regions reveals a critical pattern: the magnitude of ecological response scales with management intensity, not latitude or biome. In the Southern Great Plains and Southeast, where AMP ranchers apply substantially higher animal mass per acre (250,000–1,000,000 lbs./acre; Mosier et al. 2021), AMP grazing consistently improved water infiltration, increased soil organic carbon by 13%, and drove measurable vegetation recovery. In contrast, the weaker and more inconsistent ecological responses we observed in the NGP appear attributable not to environmental limitations but to the far lower stocking rates employed here (< 25,000–50,000 lbs./acre; Table 1). This synthesis clarifies the role of the present study within the broader continental framework: our results help identify the management-intensity thresholds below which ecological responses become attenuated, and above which recovery is accelerated.

Livestock mass stocking rates in the NGP (Table 1) are far lower than in southeastern US AMP ranches. These lower stocking rates reflect the far more arid and variable climates encountered in the Great Plains (Smart et al. 2007). Historical droughts have impacted Great Plains ranchers (Riebsame 1990), making ranchers averse to risking their livelihoods by overstocking (Kachergis et al. 2014). Such conservative stocking practices could be limiting the potential benefits of AMP grazing (Müller et al. 2007). Torell et al. (2010) and Ritten et al. (2010) also discuss managing rangelands under drought and these economic-ecological tradeoffs. AMP grazing, however, appears to enhance drought resilience by increasing above- and below-ground plant biomass (Weaver 1968), increasing infiltration, total soil organic carbon, and water holding capacity (Hudson 1994). In grasslands, greater plant diversity enhances stability (Tilman and Downing 1994) and buffers droughts (Jackson et al. 1996). If such results are cumulative and widespread, ranchers could use AMP grazing to gain resilience to drought effects, which has been documented to allow an increase in livestock carrying capacity and stocking (Teague and Apfelbaum 2023).

The ordinations suggest three themes. First, biophysical differences exist across the three geographic subregions reflecting gradients in moisture and soil texture (Table 1). The sandy, coarse-textured soil near Rapid City had the least annual precipitation, resulting in the lowest standing crop biomass. Ranches near Chamberlin, SD and Bismarck, ND had soil with

higher clay fractions that held moisture more effectively (Table 1). During the course of this study, these biophysical differences across the region that alter drought risks have emerged in conversations with the ranchers (Apfelbaum pers. commun.). In deciding herd sizes and winter ‘forage stockpiles’ (unharvested grass left for winter foraging), ranchers in the NGP rely on a dynamic and flexible grazing programme that can benefit from AMP grazing and its flexible, adaptive approach (Teague and Apfelbaum 2023). This thinking is rooted in the work by Toombs et al. (2010) on outcome-based grazing, Fuhlendorf et al. (2009) on heterogeneity-based management, Bestelmeyer et al. (2017) on state transition models, Wilmer et al. (2018) on collaborative adaptive rangeland management, and by di Virgilio et al. (2019) on precision grazing management.

Our DFA revealed that grazing management could be perfectly predicted based on extant vegetation and the presence of legacy ‘prairie sods’ (even in the absence of native plant species persistence). This suggests that AMP grazing creates consistent detectable changes in ecosystem structure and function despite complex, site-specific variation. Local environmental conditions will always remain key for making grazing management decisions, aligning with calls for place-based management (Sayre 2017) and precision grazing (di Virgilio et al. 2019).

4.5 | Actionable Management Implications and Restoration Challenges

Our results, combined with the broader continental AMP evidence base, point to several actionable management implications. First, stocking intensity matters: ecological benefits appear to require animal mass levels substantially above those currently employed in the NGP. Second, transition timelines are long—our data and the literature (Bakker et al. 2003; Török et al. 2011) suggest 10–30 years for meaningful native plant community recovery, with soil carbon and infiltration gains emerging on variable timescales depending on management intensity. Third, complementary restoration practices—including targeted grazing for invasive species control, native seed additions, and fire-grazing interactions—will likely be needed alongside AMP grazing to achieve full ecological recovery in heavily invaded rangelands.

These grazing systems face several restoration challenges. These include the persistence of non-native species (‘invasion resistance’, Bakker and Wilson 2004) and the parallel idea of ‘restoration thresholds’ (Davies et al. 2011). Such challenges imply the need for concerted integrated restoration approaches (Sheley and James 2010) that recognize the importance of restoring rangeland resilience (Walker et al. 2002).

A century-long CG has favoured a grazing-tolerant flora dominated by invasive species, as Collins et al. (1998) noted for the Konza Prairie, where fire and grazing now have opposing effects on diversity. This ‘Konza effect’ is manifest in non-native plant species dominance and invasion resistance (Bakker and Wilson 2004), restoration thresholds (Davies et al. 2011), and the need for integrated restoration approaches (Sheley and James 2010). To address these restoration challenges, restoration strategies will need to consider targeting grazing for invasive species control (Frost and Launchbaugh 2003), native seed

additions (Bakker et al. 2003), and fire-grazing interactions (Fuhlendorf and Engle 2004). However, the invasion problem, particularly with rhizomatous species like Kentucky bluegrass (*P. pratensis*) and Hungarian brome grass (*B. inermis*), suggests that restoration will require long-term commitment and perhaps more intervention. Previous heavy grazing pressure implies a consequent need for protracted recovery periods to allow deep-rooted native plant species to re-establish and replace non-native planted and invasive forage grasses.

Our study identifies several critical research needs. These include long-term studies to understand ecological recovery trajectories (Fuhlendorf and Engle 2001), the importance of legacy effects (Foster et al. 2003), and how climates interact with soils, invasive species, and grazing affects to grazing lands (Polley et al. 2013). To improve our mechanistic understanding, it will be necessary to understand soil microbiome responses (Bardgett and van der Putten 2014), plant-soil feedbacks (Bever et al. 2010), and carbon stabilization mechanisms (Cotrufo et al. 2013). Research on adaptive optimization, so that strategies typically focused on determining optimal stocking rates/densities/animal herd mass (Hart et al. 1988), recovery periods (Briske et al. 2008; Bestelmeyer et al. 2003, 2017), and effects of alternative practices (Ryschawy et al. 2012) can help ranchers understand the increased resilience of AMP grazing. Although beyond the scope of this study, social components, including new ranch business and economic models, are needed to capitalize on the benefits of AMP grazing and help ranchers anticipate and address the increasing risks from climate change.

This study has been designed to extend the current observation window (10–30 years), sufficient to assess the long-term ecological effects of AMP grazing. No specific extended studies have been formulated, but we explicitly clarify our intent for longer-term monitoring and analyses to understand ecosystem recovery timelines.

5 | Conclusions

AMP grazing in the NGP can deliver multiple ecosystem benefits while supporting higher stocking rates compared with CG. By mimicking natural disturbance–recovery cycles, AMP shifts grassland management to enhance productivity, resilience, and ecosystem services. The fact that AMP ranches exhibit 31% higher aboveground biomass after about 10 years in this arid region suggests that broader ecological gains, such as increased root biomass, greater infiltration, and higher soil organic carbon, may follow. We see biomass as an early indicator of other co-benefits, which require more time to develop in this drought-prone region. The higher above-ground plant biomass and structure on AMP ranches already show signs of benefitting wildlife, including grassland birds in the NGP (McGraw et al. Forthcoming), as documented in the Southern Great Plains (McGraw et al. 2023). About half of the AMP ranches also show increases in infiltration and higher soil organic carbon. These effects of many other variables interacted strongly with ranch location, indicating that management outcomes depend on historical conditions and/or climate. Non-native species also persist at many sites, strongly suppressing the growth of native species. This means active restoration will be needed to restore a more complete set of native species and

accompanying ecosystem services. The benefit of AMP grazing to native species cover would seem like a glimmer of opportunity to evaluate at higher animal-mass stocking rates, which may accelerate native grassland restoration. This might be especially the case if higher pressure AMP grazing pressure could be timed to reduce root reserve needs for overwintering by non-native plants.

Although AMP grazing shows promise for generating more sustainable, productive rangelands in the NGP, its benefits vary by location, and the performance indicators of success require more time to develop. Given that outcomes are context-dependent, it will be important to monitor AMP grazing outcomes across multiple regions and adjust management accordingly. Such adaptive management will require close collaboration among ranchers, researchers, and policymakers to evaluate and realize the full potential of AMP grazing. This might be approached using a phased approach to increase AMP grazing while addressing rancher concerns on stocking levels and the risks imposed by droughts in these landscapes. Realistic expectations, longer time horizons, and adaptive management are all needed for these efforts to succeed. In historically overgrazed grasslands, we expect AMP grazing to enhance soil carbon levels sufficiently to make material contributions to carbon storage, but these effects remain modest and inconsistent.

The ability of AMP to support higher livestock stocking rates and animal mass while improving some ecological metrics aligns with sustainable-intensification goals, yet human risk-avoidance—particularly concerns about drought—may limit adoption and thereby constrain ecological restoration (Griscom et al. 2017; Pretty et al. 2018). Greater adoption may therefore require financial incentives, technical assistance, and risk-reduction tools such as drought contingency supports.

This study highlights that management intensity (animal mass per unit area) appears to matter more than geography or climate. Under high management-intensity AMP grazing improved water infiltration and soil organic carbon across all study regions — from southern U.S. to Canadian sites. The weaker or inconsistent ecological responses observed in the NGP we attribute not to environmental limitations but to conservative stocking rates, as management intensity is linked to measurable ecological gains.

In summary, drought-risk aversion is likely the single biggest barrier to AMP grazing adoption in the NGP, as ranchers maintain stocking rates far below ecologically beneficial levels and destock at the first sign of drought. An encouraging trend among progressive AMP operations may be a shift from cow-calf to stocker/feeder ranching models, which allow flexible, seasonal stocking without overwintering livestock and may better align annual forage conditions and drought risk as a practical pathway to the higher animal densities needed for ecological recovery. A socio-economic analysis is beyond this study's scope, but the authors acknowledge that policy tools like drought insurance and cost-share programmes could help mitigate risk. Collaborators at ASU are examining these dimensions separately.

Author Contributions

Steven I. Apfelbaum, Susan M. Lehnhardt, and Jason Carlson collected vegetation data and contributed to the drafted paper and with the help

of Donald Waller, who conducted the final statistical analyses for this paper. P. Daniels (RES) collected infiltration data and conducted the QA/QC on sample runs. Advisor Dr. Fugui Wang contributed to framing the remote sensing biomass sampling protocol, and Jason Carlson conducted the remote sensing quality assurance, biomass estimates, and Steven I. Apfelbaum conducted the statistical analyses. Isaac Welsh and Ry Thompson (RES) collected the soil samples. Steven I. Apfelbaum (with early guidance by R. Teague, A. Williams, D. Oswald, R. Conser, and others on ranch selection) did the study design and layout. Steven I. Apfelbaum, Isaac Welsh, and Ry Thompson coordinated with rancher visits. Steven I. Apfelbaum oversaw field data collection, data analyses, statistical analyses, and drafting of the paper.

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Ethics Statement

External funders and their funding supported this project but had no sway, review or influence over the analyses, findings, and no involvement in this manuscript's preparation or representations. The authors are solely responsible for the programme design, deployment, analyses and all representations and findings herein.

Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

The data that support the findings of this study are available on request from the corresponding author. The data are not publicly available due to privacy or ethical restrictions.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section.

Supplementary Figure S1: The number of plant species per quadrat increases with total plant cover. **Supplementary Figure S2:** The mean height of vegetation on ranchlands accurately predicts above-ground dry biomass (g/m²). **Supplementary Figure 3:** Covariation between the cover of non-native and native plant species within the 648 small (1 m²) quadrats across all ranches. **Supplementary Table S1:** Plant species summed % cover from the 36 1m² sampled quadrats in each ranch for Native (N) and Non-Native (NN) dominant species across the ranch pairs.