THESIS

CONSERVATION MANAGEMENT PRACTICE IMPACTS ON RANGELANDS IN CALIFORNIA

Submitted by

Ashley Banuelos

Department of Soil and Crop Sciences

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Master's Committee:

Advisor: Keith Paustian

Francesca Cotrufo
Caroline Havrilla

ABSTRACT

CONSERVATION MANAGEMENT PRACTICE IMPACTS ON RANGELANDS IN CALIFORNIA

Rangelands hold potential for mitigating climate change through soil organic carbon (SOC) storage. SOC plays a critical role in plant growth, soil structure and water retention, yet significant degradation of the world's soils poses major risks to forage production and water quality. To address this, California has promoted the adoption of conservation practices to restore SOC storage. Given California's diverse climatic zones, climate-specific conservation strategies are necessary, as climate influences the effectiveness of different practices. These practices not only affect overall SOC stocks but also influence how SOC is stabilized in the soil, particularly through the formation of SOC fractions - particulate organic carbon (POC) and mineral-associated organic carbon (MAOC). POC generally contributes to the short-term carbon pool due to its rapid turnover however, when microbial activity is limited, its decomposition slows, allowing it to persist in the long-term carbon pool. In contrast, MAOC is more inherently stable and primarily associated with long-term carbon storage.

This thesis investigated the effects of three conservation management practices - riparian restoration, tree plantings, and perennial seeding - on SOC storage in California rangelands. We used a retrospective paired-site analysis, comparing 'restored' (i.e., locations where a conservation practice was adopted) and 'unrestored' sites (i.e., a nearby similar location but lacking adoption of a conservation practice). Restored sites varied by the time since conservation practices were adopted, providing a chronosequence approach to estimate SOC and SOC fractions (POC and MAOC) change over time. While overall SOC differences between restored and unrestored sites

were inconclusive, clear trends between practice types emerged within the restored sites. In drier regions, perennial seeding had higher POC stock compared to riparian restoration and tree plantings. Climate significantly influenced apparent SOC accrual in tree plantings, with a rate of 3.1 Mg C ha^{-1} yr^{-1} observed in moist climates, while in drier climates, SOC stocks were lower in tree planting sites compared to the unrestored sites. However, soil under tree canopies had 9% higher SOC content compared with soil sampled between trees, outside the tree canopy. Canopy cover appeared to promote proportional contributions to both POC and MAOC, highlighting the potential of tree plantings to increase SOC stocks, in relatively cooler, wetter regions. These findings underscore the importance of climate-specific conservation strategies for maximizing carbon storage in rangelands, particularly given the challenges inherent in managing these dynamic ecosystems.

The variability in the apparent response to conservation practice adoption from the retrospective paired-site analysis raised questions about potential confounding factors. While this approach offers an alternative to long-term experiments by leveraging existing conservation practices, it introduces inherent uncertainties, particularly concerning prior disturbances that may influence SOC storage. A key assumption of the paired analysis is that vegetation and soils were approximately the same on both sites within a pair before the adoption of conservation practice. However, even when controlling for factors such as soil type, topography, and current vegetation, differences in past land use - such as disturbance events occurring at one site but not the other - could have led to SOC stock differences prior to when conservation practices were implemented. These historical land-use differences may obscure or exaggerate the measurement inferred impacts of conservation practices, highlighting the need to account for site history when interpreting SOC dynamics in retrospective studies. To address this, we analyzed remote sensing imagery to evaluate site conditions, prior to conservation practice adoption, identifying disturbance events and assessing vegetation cover and soil exposure from historical observations dating back to 1984. Our analysis revealed that 12 out of 36 paired sites experienced a disturbance event, on only one of the sites within a pair, including mastication, tillage, and burn events, potentially confounding the assumption of similar SOC stocks prior to the time of conservation practice adoption. Additionally, restored sites with significant pre-treatment differences in vegetation cover and bare soil exposure often originated from more degraded conditions compared to the unrestored site in the pair. This suggests a potential selection bias toward implementing conservation practices on more degraded lands, emphasizing the need to account for pre-existing site conditions in retrospective studies. Integrating remote sensing into paired-site analyses enhances the accuracy of assessments of conservation practice effectiveness assessments on SOC dynamics. This study underscores the importance of both climate considerations in conservation management and the value of remote sensing tools for improving SOC research methodologies.

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CHAPTER 1: Introduction

Rangelands encompass diverse ecosystems including grasslands, oak savannas, and cold desert steppes (Cameron et al. 2014). Accounting for 31% of the U.S. total land cover, these landscapes provide essential ecosystem services, including livestock grazing, biodiversity conservation, and soil organic carbon (SOC) storage (Chaplin-Kramer et al., 2011; Christensen et al., 2008; Silver et al., 2010). However, these landscapes face significant threats from climate change, including altered precipitation patterns, rising temperatures, and more frequent droughts (Berg & Hall, 2015). Changes in precipitation can delay germination and shorten growing seasons, reducing forage quality and availability, leading to declines in vegetation productivity, disrupted plant communities, and accelerated soil erosion (Chaplin-Kramer & George 2013). These climate-driven effects directly influence SOC storage, as climate and vegetation production play critical roles in SOC accumulation (Carey et al., 2020). In California, an estimated 3 million metric tonnes of SOC is lost annually (CDFA 2018), threatening not only rangeland ecosystem services, but also the capacity to act as carbon sinks. As SOC plays a role in climate regulation and soil health, addressing its loss is crucial. SOC improves soil structure, infiltration, and water-holding capacity (Hudson, 1994), making rangelands more resilient to drought and floods. By implementing effective conservation practices that enhance SOC storage, we can not only adapt to the impacts of climate change but also strengthen the long-term sustainability and ecological function of these vital ecosystems.

While rangeland research has extensively investigated the influence of grazing management on SOC, the role of conservation planting practices remains less understood. These practices, including establishing perennial grasses and planting woody vegetation, are promoted through USDA's Natural Resource Conservation Service (NRCS) incentives to enhance carbon storage. However, California's complex geography, characterized by diverse climates and soil types, presents significant challenges in developing standardized conservation guidelines. Factors such as soil texture, climate, and plant litter composition strongly influence the effectiveness of these conservation efforts (Bradford et al., 2019; Ledo et al., 2020). Therefore, comprehensive studies evaluating the impact of conservation practices on SOC dynamics are crucial for guiding effective, context-specific land management strategies in California rangelands.

To further refine the understanding of how these practices influence SOC storage, it is essential to distinguish between its two key fractions: particulate organic carbon (POC) and mineral-associated organic carbon (MAOC), each with distinct physical and functional attributes (Lavallee et al., 2019). Differentiating between these fractions allows researchers to determine where carbon is stored in response to conservation practices and how it's stability changes over time. POC is primarily derived from structural litter inputs such as from perennial roots and woody debris, and it persists when microbial activity is limited, such as in dry, cold, or water-saturated environments. However, it represents a more readily available carbon pool that is susceptible to decomposition under favorable microbial conditions. In contrast, MAOC forms in soils with sufficient clay-sized mineral content, where organic inputs can associate with mineral surfaces, making them less accessible to microbes. Unlike POC, MAOC can persist even in microbially active soils due to its stabilization on mineral surfaces, contributing to longer-term carbon storage (Cotrufo & Lavallee, 2022). Understanding the balance between POC and MAOC can provide insight into how management practices influence soil's resilience to climate change. However, because these fractions are shaped by factors such as soil type, climate, and land use history (Cotrufo & Lavallee, 2022; Hansen et al., 2023; Fu et al., 2024), it is essential to analyze management impacts across a range of soil types, and ecosystems to develop effective conservation strategies. Since management practices are not a one-size-fits-all solution, their effectiveness may vary depending on environmental conditions.

The second chapter of this thesis evaluates the impact of three widely adopted conservation practices - perennial seeding, riparian restoration, and tree planting - on SOC, POC, and MAOC across California's rangelands within diverse climates and soil types (Figure 1). Over the past few decades, ranchers have increasingly implemented these practices, creating an opportunity to apply a retrospective study using paired-site comparisons between restored and unrestored sites. Additionally, by using a chronosquence approach, this study aimed to determine the timeframe required to detect significant changes in SOC by selecting sites along a gradient of time since practice implementation (Figure 1). Understanding how quickly SOC responds to different conservation practices can help inform policymakers and land managers, setting realistic expectations for soil carbon storage benefits.

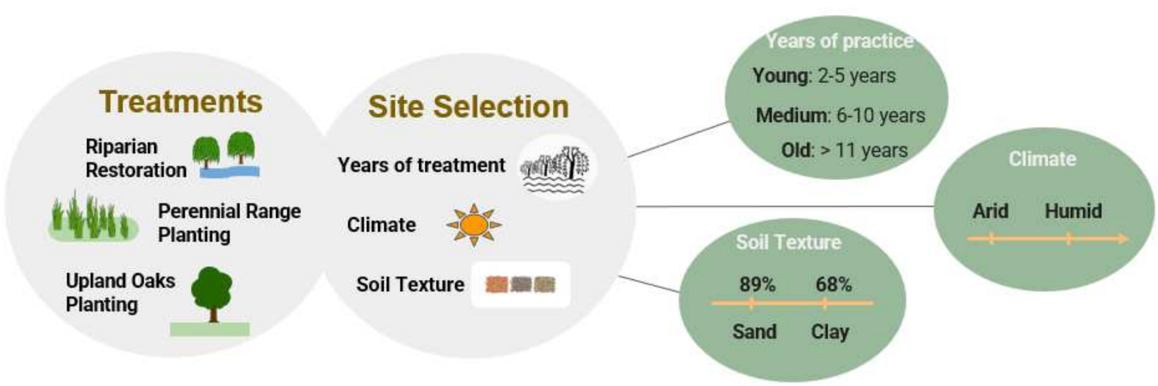


Figure 1. Factors considered within our site selection, demonstrating the range of soil texture, climates, and duration of practices across three conservation practices- riparian restoration, perennial planting, and tree planting.

Retrospective paired-site designs provide a valuable perspective by capturing the variability inherent in working landscapes. Controlled long-term experiments, with statistically replicated treatments and time series measurements, provide the most reliable way to measure the effects of conservation practice implementation on ecosystem attributes such as soil carbon stocks. However, long-term field experiments are in general few in number and in the case of rangelands in California are almost entirely lacking, and in no way can encompass the diversity of climate regimes, soil types and management systems on these rangelands. Hence on-farm studies provide the only feasible alternative for field measurements that can account for the real-world variability in agricultural settings. By carefully selecting paired sites that are similar in terms of soil type, topography, and historical land use, researchers can minimize the influence of confounding factors and more accurately assess the impact of the conservation practice on SOC (Poeplau & Don, 2013, Laganiere et al., 2009). This approach enables the evaluation of conservation effectiveness across diverse climates without the substantial investment required for long-term, controlled trials.

However, retrospective studies have inherent limitations. Confounding factors from unaccounted for historical management practices, such as tillage, mastication, or burn events, can leave legacy effects that alter soil properties, including SOC (Wyngaard et al., 2016; Collins et al., 2023; Phukubye et al., 2022). These events have a particularly strong influence on the perceived effects of conservation practices when they occur at only one of the two paired sites. The absence of detailed historical management records can confound the selection of comparable sites and hence the interpretation of SOC changes and the true effectiveness of conservation practices (Stanley et al., 2023). Therefore, careful consideration of rangeland forage production and prior land management and disturbance history is crucial when selecting sites to better interpret SOC data and evaluate the success of conservation intervention.

Remote sensing has become an increasingly valuable tool for land management analyses (Beeson et al., 2020; Washington-Allen et al., 2006), yet its application in site selection for retrospective studies remains largely unexplored. It has the potential to improve site selection by identifying historical land uses and assessing forage health and abundance prior to conservation practice implementation. This provides a powerful means to evaluate historical landscape conditions, which can reduce uncertainties in site selection and enhance the robustness of retrospective studies. It is particularly important in cases where control site differences may confound results (Kucharik et al., 2003; Karlen et al., 1997), limiting the ability to draw meaningful conclusions about the effectiveness of conservation practices. In a time of rapid technological advancement, integrating remote sensing into rangeland research can improve both the accuracy and efficiency of data analysis. Many conservation research projects require significant time and financial investment, yet without careful site selection, results may remain inconclusive.

Chapter 3 of this thesis explores the use of remote sensing to evaluate the comparability of paired sites used in Chapter 2 before practice implementation. This analysis aimed to provide a deeper understanding of the soil dynamics that were inferred from the field measurement reported in Chapter 2 and assessed how historical land use management may have influenced our results. By leveraging remote sensing tools, we sought to analyze initial conditions between our paired-sites and highlight the challenges associated with paired-site study designs. By analyzing vegetation productivity and soil exposure, we aimed to quantify the degree of degradation between sites prior to practice implementation. The remote sensing analysis may also reveal distinct soil and vegetation disturbance events that could significantly impact SOC stocks. This research underscores the potential of remote sensing as a tool for improving site selection in retrospective studies, ultimately enhancing conservation practice research.

By bridging soil carbon research with advanced remote sensing applications, this thesis underscores the importance of integrating a range of technologies into rangeland conservation strategies. The findings from this thesis not only inform ranchers and land managers on the effectiveness of conservation practices but also have broader implications for climate adaption policies. As California's rangelands continue to face environmental challenges, leveraging scientific research to support evidence-based management will be essential in promoting sustainable land stewardship and long-term ecosystem resilience.

CHAPTER 2: Conservation management practice impacts on rangelands in California: Field measurements and analysis

Introduction

Rangelands account for 30% of California's land cover and provide multiple ecosystem services (ES), including livestock production, biodiversity, and carbon storage (Havstad et al., 2009). However, climate change has exerted pressure on Californian rangelands, through prolonged droughts, and increased wildfires (Asner et al., 2016, Stewart et al., 2020). To ensure the long-term sustainability of rangeland ecosystems, conservation management strategies are essential for fostering ecosystem services such as carbon storage. Increasing soil carbon stocks improves many soil attributes such as soil structure, water holding capacity and nutrient availability, which promotes increased forage production, greater biodiversity, and mitigates air and water pollution (Alloway, 1995; Kalbitz et al., 2000). Despite the recognized importance of soil organic carbon (SOC) in rangeland ecosystems, a comprehensive understanding of its storage potential across different conservation practices and climatic contexts remains limited. Climate exerts a significant influence on soil carbon pools, and the effectiveness of various conservation practices in enhancing SOC storage likely varies across different climatic regions (Carey et al., 2020b). Further research is needed to inform effective management decisions and help increase carbon storage to achieve multiple benefits.

California has implemented incentive programs to encourage the adoption of conservation management practices aiming to achieve climate mitigation, and also contribute to climate adaptation. For example, the California Department of Food and Agriculture's (CDFA) Healthy

Soils Program (CDFA 2018) is focused on enhancing soil quality. This program seeks to improve resilience of California's livestock industry, which generates approximately $2.6 billion in annual revenue.

Various estimates of soil carbon storage potential from improved management practices on rangeland have been made, with research yielding average per area estimates ranging from 0.1 to 1.12 Mg C ha^{-1} $year^{-1}$ (Conant et al., 2017, Matzek et al., 2020). In California annual-grass dominated systems, improved management practices, including seeding perennial grass species, riparian restoration (including shrub and tree planting), and tree planting – have been implemented to promote rangeland resilience. Previous studies have demonstrated that the transition from perennial to annual grasses can result in significant SOC losses, with estimates of losses up to 40 Mg C ha^{-1} in the top half meter of soil (Koteen er al., 2011). Conversely, tree plantings have shown potential to enhance soil carbon storage within California (Carey et al., 2020a). However, there is a lack of long-term field studies assessing the impacts of these conservation practices across the diverse climatic zones found within California, especially considering the significant variability in SOC stocks across California's rangelands (Carey et al., 2020b).

To better understand soil carbon differences, we measured soil organic carbon stocks (SOC) as well as functional soil carbon pools, specifically mineral-associated organic carbon (MAOC) and particulate organic carbon (POC). Measuring these fractions can provide a clearer picture of carbon stock differences across varying management regimes and climates, as these fractions differ in their formation pathways and relative stability, being impacted by litter quality, climate, and soil properties (Cotrufo and Lavalle, 2022). Aboveground biomass (AGB) was also measured, as it provides a direct link to subsequent contribution of organic matter to soil carbon pools (Arije et al., 2024). Differences in AGB between paired sites may help explain observed variations in SOC, POC, and MAOC.

Our observational study aimed to fill this gap of understanding conservation practices impact on soil carbon stocks across diverse California rangeland and climates. To assess the potential for rangeland conservation practices to increase soil carbon storage, we used a retrospective study of 36 pairs of rangeland sites, consisting of a 'restored' site, where rangeland conservation practices were adopted in the past and maintained to the present, and an 'unrestored' site, where the specific conservation practices were not adopted. Our research incorporated a chronosequence, or 'space-for-time' substitution, approach by including sites with varying durations of conservation management, effectively treating time as a factor in assessing carbon accrual. This allowed us to assess the temporal dynamics of soil carbon accumulation, including: 1) At what time-point do measurable differences in carbon storage become apparent following the implementation of conservation practice? and 2) Which climatic conditions are most conductive to carbon accumulation under different conservation practices? This retrospective approach aimed to assess the short-term and long-term impact of conservation practices on SOC storage across California rangelands.

Materials and Methods

Site and Study Description

Our field sites were distributed across three distinct regions in California: The San Joaquin Valley, the Sacramento Valley, and the Central Coast (Figure 2.1), selected to represent the heterogeneity of California's rangelands. These regions encompass a diverse range of Mediterranean climates, with a gradient extending from cooler summers in the coastal areas to drier summers in the interior Valleys. Mean annual precipitation across these regions ranged from 378-1071 mm, mean annual temperature from 13-17 °C, and elevation varied from sea level to 601 meters. Soil characteristics also exhibited significant variability, with percent clay ranging from 6% to 56%, and pH values ranging from 4.2 (acidic) to 8.2 (slightly alkaline).

Our research focused on three conservation practices implemented across a range of durations, providing a temporal perspective on their effectiveness within restored sites. The conservation practices standards are from the USDA Natural Resources Conservation Services (NRCS): 1) Range Planting (550) referred to as perennial seeding, i.e., seeking to transition from annual-dominant to perennial-dominant grasses; 2) Riparian Forest Buffer (391) referred to as riparian restoration, i.e., plantings native grasses, forbs, shrubs and trees and fencing out livestock along streambanks; and 3) Oak Forest Regeneration (E666J) referred to as tree plantings, specifically planting oak trees within upland areas to establish an oak-grass savanna system. Ranchers have implemented these practices over several decades across California following NRCS practice standards. Therefore, our retrospective study selected ranches that have undergone at least one of the three conservation practices. Among the 36 pairs of sites in our study, 12 pairs each had implemented one of these three conservation practices with varying durations since practice implementation, ranging from 2 to 27 years. In addition to using age as a continuous variable, to facilitate other analyses, we categorized the ages of restored sites into three distinct groups: "young" (2-5 years), "medium" (6-10 years), and "old" (> 11 years), while unrestored sites were grouped as “zero” (0 years).

Individual conservation practices had their own specific requirements. Notably, riparian restored sites required fencing out cattle from the restored sites. Additionally, treated perennial seeded sites needed to have roughly 30% successful establishment of sown perennials. All sites were subject to cattle grazing, and three sites were also accessible to grazing by large wild ungulates (e.g. deer and elk). Regardless of these specific requirements, all selected sites were accessible for sampling within the timeframe of the project and fulfilled the overall criteria of our observational design.

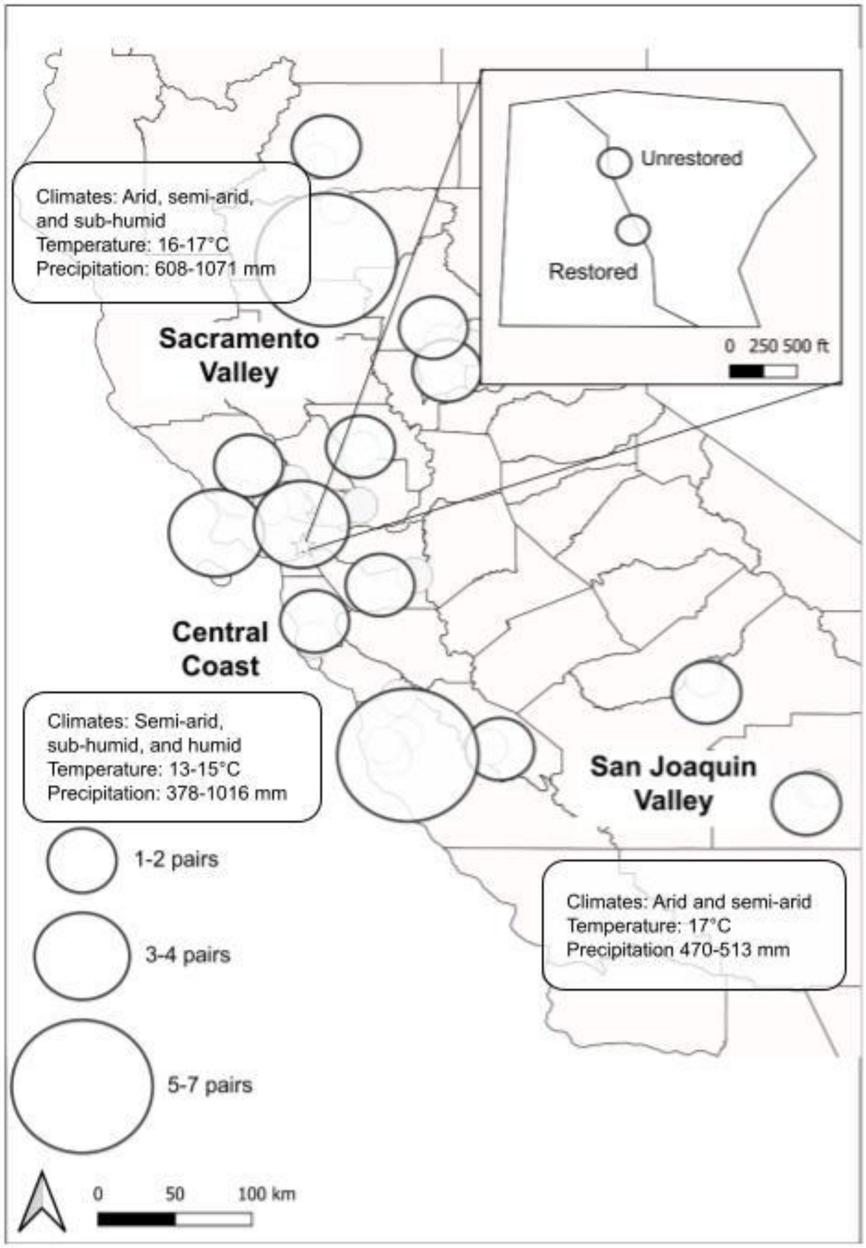


Figure 2.1. Map of study sites across three regions in California by counties. Inset map displays typical proximity between restored and unrestored sites. Sites were distributed across three distinct regions, Central Coast, Sacramento Valley, and San Joaquin Valley, spanning arid to humid climates (PRISM Climate Group, 2020). Average annual temperature and precipitation values are for designated sites within the respective regions.

Each paired site consisted of a restored site, where a conservation practice had been implemented within the past 30 years, and an unrestored site maintained under previous management.

Unrestored sites were carefully selected to have, similar topography, grazing management practices, and soil characteristics to their paired restored sites. In many cases, paired sites were located on adjacent properties within 500 meters of each other, potentially minimizing the impact of micro-environmental variations. Thus, retrospective approach relies on the assumption that vegetation and soil properties (including SOC stocks) on restored sites were comparable to those in the unrestored sites, prior to the implementation of conservation practices.

To characterize climate conditions at each site, we used the Global Aridity Index Database (Trabucco and Zomer, 2018). This index (0 to 1) is based on the ratio of precipitation to potential evapotranspiration, and the aridity of our specific sites exhibited a range from 0.19 (indicating arid conditions) to 0.67 (reflecting more humid regions). There was an unequal distribution of sites among arid and humid regions (Figure 2.2); therefore, for some analyses, we combined arid sites with semi-arid sites to create a dry climates category and humid sites were merged with sub-humid sites to create a moist climate category for conservation practice analyses.

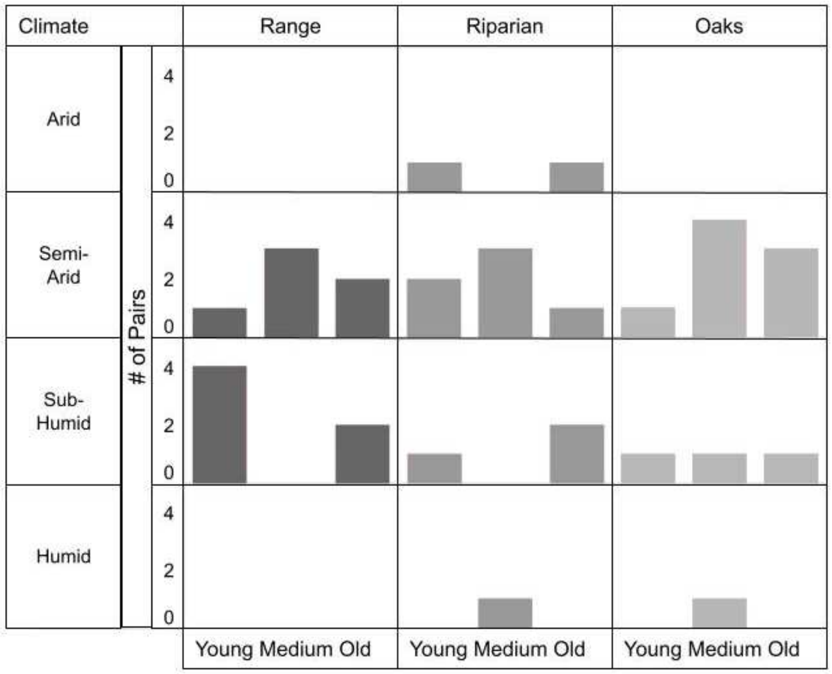


Figure 2.2: This graph displays pair distribution by conservation practice, climate and age groups; "young" (2-5 years), "medium" (6-10 years), and "old" (> 11 years). This highlights the unbalanced distribution of paired-sites across the three conservation practices. Sites were also selected based on an array of soil types (Table 2.2).

The vegetation composition across these regions varied but primarily consisted of annual grasses and graminoids (e.g. *Avena sativa, Elymus caput-medusae, and Aegilops cylindrica*) and various forbs, along with scattered relic oak trees (*Quercus* ssp.). Notably, the Sacramento Valley region featured a higher prevalence of invasive grass species compared to the other sites.

Soil and Biomass Sampling

Field data collection encompassed both above and below-ground measurements. Aboveground sampling involved several steps, with details provided below. Briefly, we gathered peak herbaceous biomass measurements and clippings from 1.5 m diameter grazing exclosures. Woody plant biomass measurements were breast height diameter and plant height via field tapes and smartphone clinometer (Foster et al., 2023). We quantified perennial grass cover using a line-point intercept along an 80-meter perimeter transect, which traced the perimeter of a 20m x

20m square established at each sampling location, with the presence or absence of perennial grasses evaluated at 5-meter intervals.

We collected 11 soil samples (0-30cm depth) from each restored and unrestored sites at 36 sites pairs (n=792). Soil samples were randomly placed within the boundary of each site, which was determined based on the extent of the restored area. Site sizes varied from less than half an acre to 30 acres. Sample locations were selected using a random number generator to determine paces along two perpendicular axes.

Where trees were present, sample locations were noted by their proximity to canopy by using a quadrant. A 2.4-meter diameter circular site was established at each sample location. Within each site, the percentage of canopy cover was estimated visually by counting presence or absence of tree canopy within four equal sections of a 2.4-meter diameter circle around the point, in which areas with no canopy overhead were rated 0, areas with canopy cover were marked as 1, adding up to a maximum of 4 if each section contained some canopy cover. For below-ground measurements, soil samples from each site using a 5.08-centimeter bucket auger, following the removal of aboveground vegetation using clippers and brushing away surface litter. Bulk density was computed using the millet method (Foster et al., 2023) to determine the volume of soil (including coarse fragments) removed by auger. There were a few missing bulk density measurements due to challenging field conditions and missing measurements were replaced with SSURGO bulk density values. The fine fraction mass was measured separately, and bulk density was corrected by accounting for the volume of coarse fragments (rocks).

Soil Analyses

Soil samples were initially sieved to an 8mm size fraction, with separation of coarse litter, roots and rocks and then air dried. An air-dry subsample was then passed through a 2 mm sieve for subsequent soil analyses of the fine fraction. The remaining roots and rocks were weighed and discarded. All soil samples were analyzed for pH in a 2:1 slurry of deionized water to soil (Thomas, 1996). The sand fraction (>53µm) was quantified by mass after wet sieving, while the percent clay content (<2µm) was assessed by hydrometer measurements after a settling period of six hours (Gavlak et al., 2005). Silt content was calculated as the difference between the percent sand and percent clay.

To analyze carbon within our SOC fractions we first segregated soil into distinct size fractions, i.e., particulate organic matter (POM+sand) and mineral-associated organic matter (MAOM+silt +clay. We followed the protocol to fractionate by size as outlined by Cotrufo et al. (2019). Most soil samples were dried at 60 ° C, with a small subset of quarantine samples that were heat-treated (110 ° C). Soil samples were mixed with 30mL of a 0.5% sodium hexametaphosphate solution and shaken for 18 hours with glass beads. POM+sand was separated using a 53µm sieve, where MAOM+silt+clay passed through the sieve (<53µm), leaving POM on the surface of the sieve. Subsequently, the soil fractions were dried at 60°C and ground using a mortar and pestle prior to analysis of total SOC and POC using Tru-SPEC elemental analyzer (Leco Corp., St. Joseph, MI). To account for sand mixed in with POC, we adjusted POC based on the proportion of POM in the total sample. We did this by multiplying the POC concentration (g POC/g POM) by the fraction of POM in the total soil mass (g POM/g soil), resulting in POC per unit soil mass (g POC/g soil).

$$f(POM) \times POC\ (g)$$

Subsequently, MAOC was calculated by subtracting the sand-corrected POC from the SOC content:

$$SOC\ (g) - POC(g)$$

Lastly, soil samples were tested for carbonates on samples with a pH > 7 using a fizz test. We identified 22 samples that needed to be measured for inorganic carbon; we used the pressure transducer method (Sherrod et al., 2002). Following this analysis total soil carbon values were adjusted for carbonates to get total SOC.

Statistical analyses

Recognizing the influence of climate on SOC distribution across California rangelands (Carey et al., 2020), first we determined climatic-related (i.e., aridity index) differences in SOC, POC, and MAOC stocks using a one-way analysis of variance (ANOVA). Post-hoc Tukey HSD tests were performed to determine differences between climate and carbon stocks after testing for assumptions of homogeneity of variance and linearity in R Studio version 4.4.0 (2024). We also performed a t-test on bulk density between paired-sites to determine the best way to proceed with further analysis between paired-sites (e.g., by %SOC or SOC stock).

We used a hierarchical mixed-effects model to examine the impact of conservation practices on SOC, in aggregate and by specific practices. This was done as a function of time since adoption and by climatic factors, controlling for site location and pair as random effects, to control for potential correlations with sites and among soil samples. Several iterations of linear mixed effects models were performed using R package *lme4,* after testing assumptions and log-transforming %SOC, %POC, and %MAOC. The analyses first used aggregated data from all conservation practices to test for any overall effects and then used stratified data for each individual practice to examine practice-specific impacts on SOC unique impacts. The models were structured with an interaction between treatment (unrestored/restored) and age group, as well as the interaction between treatment and climate. Percent clay was used as a covariate to account for its potential influence on carbon storage.

We repeated a similar analysis to compare the impact among conservation practice on SOC, POC, and MAOC stocks. We excluded unrestored sites from this analysis as they did not undergo any conservation practice. We tested the interaction between practices and climate type (dry and moist), as well as testing the interaction between practices and age group. Finally, to test for significant differences between specific conservation practices, we conducted pairwise comparisons of each age group and climate, using *emmeans* package in R.

To delve deeper into the effects of tree plantings, we tested the impact of canopy cover on SOC and SOC fractions. This was done by converting canopy cover values (0-1) to either "under canopy" (1-4) or "outside" (0). We used a linear mixed effect model with canopy cover and percent clay as independent variables and ranch as random effect. A similar analysis was performed with climate as a covariant.

A multiple linear regression model was used to identify soil carbon changes by climate over time. By leveraging data from all conservation practices (without unrestored sites), this model incorporated a three-way interaction encompassing conservation practices, climate type, and years of practice. Further, all tests were evaluated at the $\alpha = 0.1$ in consideration of the high variability of soil variables within rangelands.

Lastly, to quantify the effect size of our SOC data, we calculated Cohen's d using the mean and pooled standard deviation of our paired-sites (restored and unrestored). Using Cohen's d we calculated power of our analysis using the *pwr* package in R.

Results

Variations in soil properties across diverse California climates

On average SOC stocks had a negative relationship with aridity, such that drier climates had lower carbon stock compared to wetter regions. However, sub-humid sites exhibited the highest SOC stocks with an average of 85 Mg C ha^{-1} and humid sites followed with an average of 75 Mg C ha^{-1} (Table 2.1). Similarly, drier regions (arid/semi-arid) had significantly lower MAOC stocks compared to more mesic regions (sub-humid/humid) ($P<0.001$). POC stocks were highest for the sub-humid climate, and consistently lower for humid, arid and semi-arid sites ($P<0.05$) (Figure 2.3). As a fraction of SOC, MAOC accounted for a substantial portion, ranging from 70-78% (Table 1), with relatively minor differences across climate types. We identified one pair that had inorganic carbon with values ranging from 0.07 g to 0.8 g within a perennial, semi-arid region.

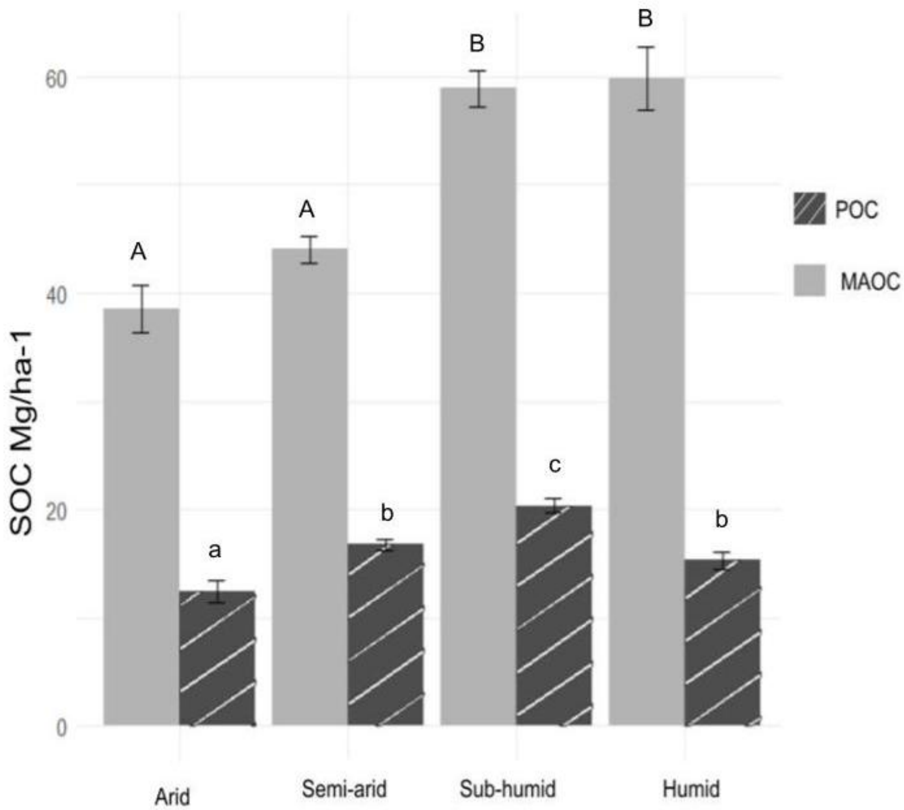


Figure 2.3 *Distribution of mean and standard errors of SOC fraction stocks (Mg C ha^{-1}) across climates in California rangelands. Letters above bars denote significant differences in SOC fraction stock (POC and MAOC) between climates based on ANOVA results. Sites in drier climates (arid and semi-arid) had significantly lower SOC stock compared to wetter climates (sub-humid and humid).*

Climate also influenced other soil and biomass properties within our sites (Table 2.1). As anticipated, soil pH exhibited an increasing trend from wettest to driest sites. Similarly, herbaceous plant biomass also increased along the gradient from arid to humid climate types. However, no discernible patterns were observed for bulk density or soil textures as a function of climate type upon a qualitative assessment of data in Table 2.1.

Table 2.1 Rangeland soil and aboveground herbaceous biomass properties distribution across climates

Climate	SOC Stock (Mg ha^{-1})	POC Stock (Mg ha^{-1})	MAOC Stock (Mg ha^{-1})	*f*(maoc)	Bulk Density (g cm^{3})	pH	Sand (%)	Clay (%)	C:N	Biomass (g m-^{2})
Arid	51 ±2.84	12.4 ±1.02	38.5 ±2.16	0.75 ±0.2	1.2 ±0.05	7 ±0.08	73.3 ±1.22	11.4 ±0.5	10.1 ±0.45	137 ±11.4
Semi-Arid	60 ±1.24	16.1 ±0.45	42.7 ±1.09	0.71 ±0.08	1.4 ±0.01	6.5 ±0.03	46.4 ±0.9	23.7 ±0.5	8.64 ±0.13	280 ±7.67
Sub-Humid	85 ±2.07	20.4 ±0.61	64.4 ±1.97	0.72 ±0.09	1.3 ±0.02	5.6 ±0.03	43.5 ±1.2	27 ±0.85	9.61 ±0.17	353 ±10.9
Humid	75 ±3.08	15.3 ±0.83	59.8 ±2.92	0.79 ±0.1	1.1 ±0.04	5.3 ±0.4	67.3 ±1.23	13 ±0.8	9.45 ±0.21	444 ±15.9

Values are mean ± standard error. Bulk density is expressed as g cm^{-3}, and SOC stocks are expressed as Mg ha^{-1} from 0 to 30 cm depth. Biomass is defined as standing herbaceous biomass in g m^{-2}. f(maoc) is defined as the fraction of MAOC relative to SOC.

Management Driven Impacts

In aggregate, the sites with conservation practices showed no differences in carbon content (SOC, MAOC, POC) compared to the unrestored group, with average SOC at 1.7% for both restored and unrestored sites (Supplemental Table 2.1). Similar results were found by conservation practice where no differences between practice types were found by climate (Figure 2.4) or age group. We found a small effect size (Cohen's d) of 0.03 and our paired study had a low power of 0.16, meaning there was only a 16% chance of detecting a true effect if one existed.

Although our rangelands had generally similar soil carbon values, one medium-aged, sub-humid tree-planted pair that was located on a poorly drained soil had an SOC content 3X greater than average SOC content. To prevent this outlier from influencing our results, it was excluded from any analyses.

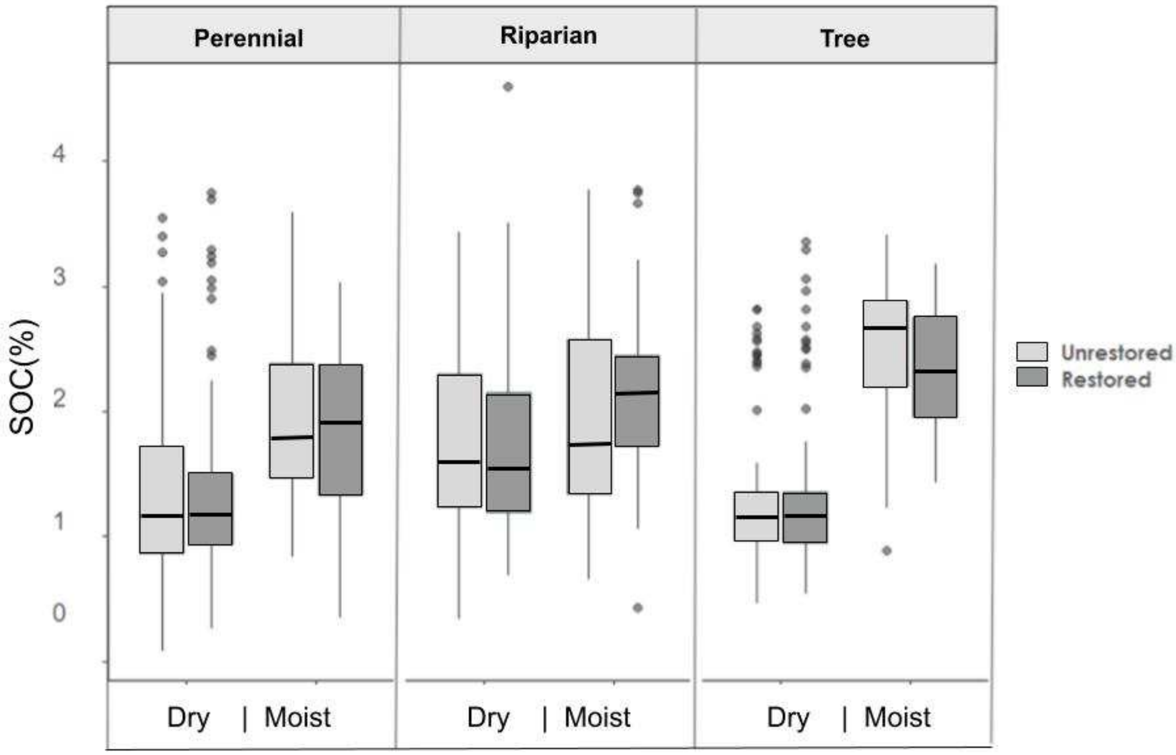


Figure 2.4 This boxplot displays the mean, standard errors, and first and third quartiles of SOC content for unrestored and restored sites across three conservation practices (perennial, riparian, and tree planting) under dry and moist climates. Aggregated data did not reveal significant differences between restored and unrestored sites, similarly when analyzed separately by climate conditions and age groups (P>0.1). However, tree-planted sites in moist climates exhibited slightly higher mean SOC values compared to unrestored sites. Additional information on the distribution of soil properties (MAOC, POC, texture, pH, bulk density) are listed in Table 2.2.

Table 2.2 Distribution of soil properties and herbaceous biomass (oven-dry) across conservation practice of restored versus unrestored sites across moist and dry regions

Practice	Climate	Unrestored/ Restored	SOC (%)	SOC Stock Mg ha^{-1}	MAOC (%)	POC (%)	*f*(maoc)	Biomass (g m^{-2})	Sand (%)	Clay (%)	pH	BD (g cm^{-3})
Perennial Planting	Dry	Unrestored	1.37 ±0.1	56.8 ±2.96	0.99 ±0.08	0.44 ±0.03	0.65 ±0.3	328 ±27.5	43.6 ±2.39	24.8 ±1.03	6.2	1.5 ±0.03
		Restored	1.40± 0.1	58.0 ±2.74	0.96 ±0.08	0.48 ±0.03	0.64 ±0.2	280 ±9.23	46 ±2.01	23.4 ±0.79	6.1	1.5 ±0.03
	Moist	Unrestored	1.96 ±0.08	78.5 ±2.56	1.34 ±0.07	0.63 ±0.04	0.67 ±0.2	433 ±17.7	45.1 ±2.35	27.5 ±1.8	5.6	1.4 ±0.02
		Restored	1.82 ±0.09	70.5 ±3.0	1.32 ±0.08	0.51 ±0.03	0.68 ±0.2	551 ±19.6	42.3 ±2.72	28.5 ±1.69	5.8	1.3 ±0.03
Riparian Restoration	Dry	Unrestored	1.76± 0.08	68.1 ±2.8	1.36 ±0.07	0.41 ±0.02	0.75 ±0.1	305 ±20	48.8 ±2.4	22.4 ±1.2	6.5	1.3 ±0.03
		Restored	1.69 ±0.08	58.8 ±2.9	1.22 ±0.06	0.46 ±0.04	0.73 ±0.2	243 ±16	52.8 ±2.19	19.6 ±1.1	6.0	1.2 ±0.03
	Moist	Unrestored	1.91 ±0.1	67.3 ±4.6	1.39 ±0.1	0.52 ±0.04	0.72 ±0.2	268 ±13	53.6 ±2.92	16.3 ±1.47	5.5	1.2 ±0.04
		Restored	2.15 ±0.1	74.5 ±4.1	1.72 ±0.1	0.44 ±0.02	0.79 ±0.1	240 ±15	51.5 ±2.38	20.5 ±1.23	5.8	1.2 ±0.04
Tree Planting	Dry	Unrestored	1.33 ±0.07	55.9 ±2.9	1.00 ±0.06	0.33 ±0.02	0.73 ±0.2	275 ±26	52.7 ±2.03	20.4 ±1.29	6.7	1.4 ±0.03
		Restored	1.35 ±0.07	55.5 ±3.1	1.04 ±0.08	0.31 ±0.01	0.73 ±0.2	214 ±7	52.4 ±2.06	19.8 ±1.21	5.6	1.4 ±0.03

	Moist	Unrestored	2.5 ±0.1	101 ±4.3	2.08 ±0.09	0.41 ±0.02	0.83 ±0.08	469 ±17	44.5 ±2.67	28.9 ±1.95	5.1	1.4 ±0.02
		Restored	2.3 ±0.08	92 ±4.2	1.89 ±0.08	0.45 ±0.03	0.81 ±0.1	389 ±11	49.3 ±3.59	26.2 ±2.7	5.2	1.3 ±0.03

Values are mean ± standard error. Bulk density (BD) is expressed as g cm^{-3}, and SOC stocks are expressed as Mg ha^{-1} from 0 to 30 cm depth. Biomass is defined as standing herbaceous biomass (oven-dried) in g m^{-2}. f(maoc) is defined as the fraction of MAOC relative to total SOC.

Conservation practice impact on soil carbon storage and aboveground biomass carbon

In subsequent analyses, we looked for potential differences between conservation practices and interactions with different climate conditions. All three conservation practices, i.e., riparian restoration, perennial seeding, and tree plantings, and for restored versus unrestored sites, showed similar SOC, even when controlling for years since implementation and climate (Figure 2.4, Table 2.2).

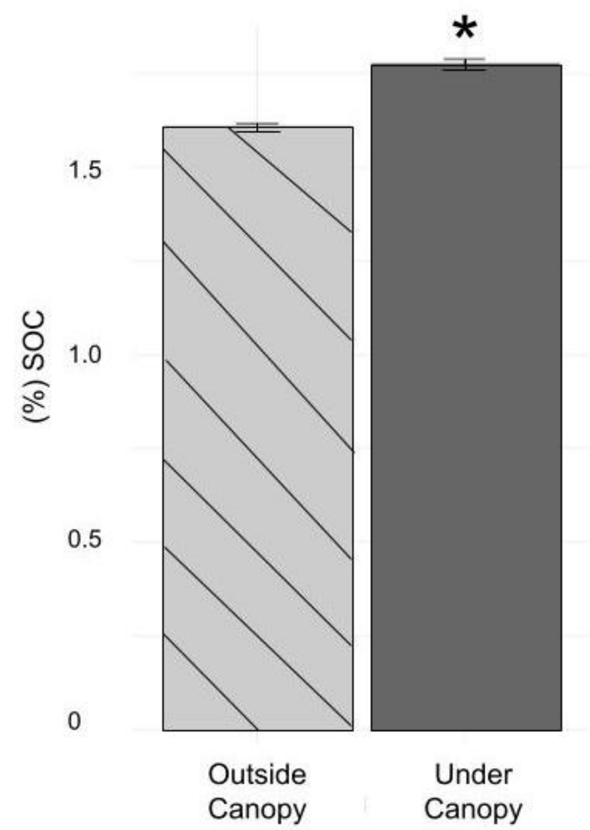


Figure 2.5 Mean and standard errors of %SOC under canopy and between canopies. Higher SOC was found under canopy (P<0.01), noted with an (*). The analysis was used on log-transformed data to meet statistical assumptions; however, results are presented in their original scale for clarity and ease of interpretation.

However, analyzing the impact of canopy cover, across all practices, we found differences in SOC related to canopy cover. Soil samples taken under canopy had 9% higher SOC, 10% higher MAOC, and 9% higher POC content, compared to those taken outside of the canopy (Figure 2.5). The impact of canopy cover was moderated by climate, with a +18% MAOC content in moist climates ($P<0.01$), and +10% POC content in dry climates ($p<0.05$).

As expected, tree biomass proved to have a positive relationship with years since adoption. Yet, the total tree biomass was substantially different between age groups (Supplemental Table 2.2). However, our sample included only one young site, which had the lowest tree biomass (0.2 Mg C ha^{-1}). Tree-planted sites in medium (0.4 Mg C ha^{-1}) and old age classes (17 Mg C ha^{-1}) were evenly distributed (Supplemental Table 2.2).

Across all practices, herbaceous biomass showed a counter-intuitive response to conservation practices. Among all climates, unrestored sites had an average of 122g m^{-2} more herbaceous biomass (dry matter) compared to restored sites (Table 2.2). The only conservation practice that showed a positive influence on herbaceous biomass was within moist climates.

Comparison of conservation practice impact on soil carbon and carbon fractions over time

Comparing the three conservation practices, we identified potential impacts on soil carbon as a function of time since practice adoption. Analyzing only sites with conservation practices, we found significant differences between practices by age group and climate within MAOC and POC stocks, but not for SOC stocks.

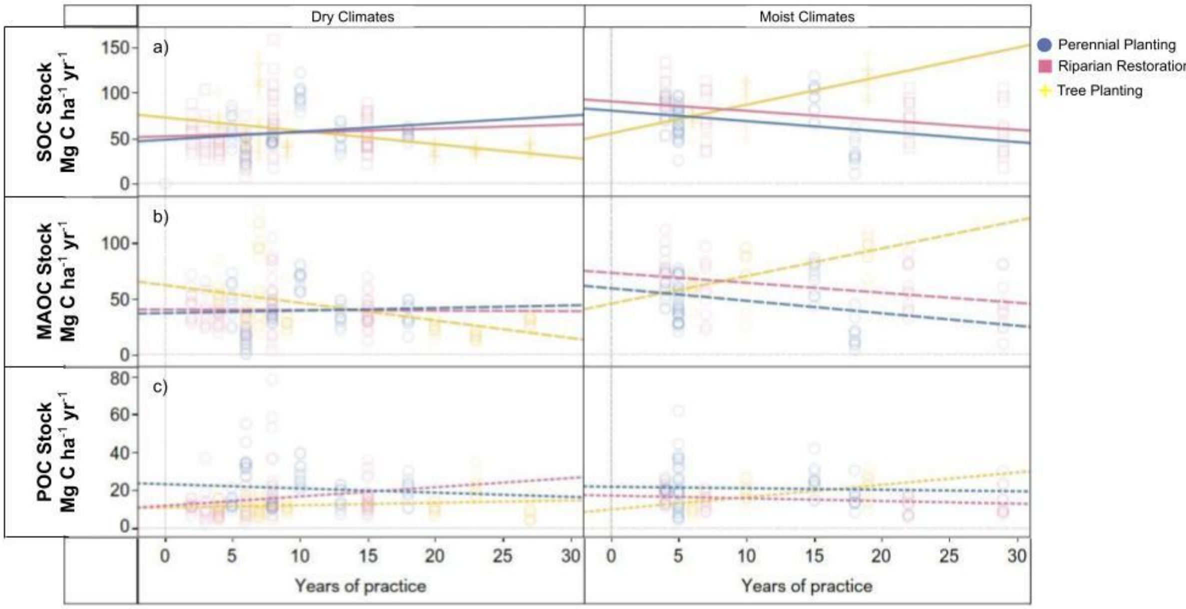


Figure 2.6 Inferred rate of soil carbon change over time since the year of practice implementation by the three conservation practices, riparian restoration, perennial seeding, and tree plantings. Soil carbon distributions in dry and moist climates across three pools a) SOC stock (R^2 =0.27, P<0.0001), b) MAOC stock (R^2 =0.3, P<0.0001), and c) POC stock (R^2 =0.17, P<0.0001) in Mg C ha^{-1}.

Temporal trends for tree plantings and perennial seeding varied by climate. In moist climates, tree plantings had higher MAOC stock (26 Mg C ha^{-1}, SE ± 4.4, P<0.05), compared to perennial seeded sites. This was accompanied by a positive trend in SOC over time, increasing by 3.1 Mg C ha^{-1} yr^{-1} in tree-planted sites. In contrast, tree plantings in dry climates showed a negative trend, decreasing by 1.7 Mg C ha^{-1} yr^{-1}. In dry climates, both tree plantings and restored riparian sites had lower POC stock compared to perennial seeding, -44% (SE ± 18%, P<0.1) and -52% (SE ± 18%, P<0.05), respectively.

By age group, perennial seeded sites had higher POC stock compared to tree-planted sites. Within sites ranging from 6-10 years (medium age group) since conservation practice adoption, perennial seeded sites had 13.1 Mg C ha^{-1} more POC compared to tree-planted sites (P<0.05,

Figure 2.6c). POC within perennial sites stayed consistent over time within both climates, while tree-planted sites tended to be lower during the 6–10-year range, especially within dry regions.

Riparian restoration sites also varied temporally and by climate. In moist climates, riparian restoration, showed a decrease in total soil carbon stocks of 1 Mg C ha^{-1} yr^{-1}. In contrast, for dry regions, SOC and MAOC did not change significantly over time.

Discussion

SOC stocks across the study sites ranged from 50 to 85 Mg C ha^{-1} (0-30cm), where dry regions had the lowest SOC stock and more mesic regions had the highest SOC storage. This broad climate-driven pattern is well understood and has been frequently reported due the greater primary productivity and plant inputs in wetter sites (e.g. Jenny, 1941; Jobbágy and Jackson, 2000, Cotrufo and Lavallee, 2022). Relative to other California rangeland studies, our SOC stock measurements were somewhat greater than the SOC stocks (39-57 Mg C ha^{-1}, 0-40 cm) reported by Carey et al. (2020). However, our findings are closer to values found by Silver et al. (2010) who reported an average of 58 Mg C ha^{-1} to 25cm depth, in a meta-analysis of 17 studies across California. The differences between our findings and previous studies could be attributed to differences in overall soil depth of A horizon, topography, and microclimates at our study locations.

The impact of climate on SOC and MAOC stocks were not directly proportional to increases in moisture levels across the full range of sites. It is somewhat surprising that sub-humid regions had higher SOC stocks compared to humid sites, as higher moisture levels are generally associated with increases in SOC accumulation. Typically, humid climates, with higher moisture availability, support greater organic matter inputs and microbial activity (Bai & Cotrufo, 2022).

Despite our humid sites having an average of 26% more aboveground biomass, which is typically associated with higher SOC due to increased plant inputs and microbial activity (Arije et al., 2024), SOC was somewhat less than the sub-humid group. This suggests that factors other than moisture, such as vegetation type, or soil physical properties, may be influencing SOC dynamics across these regions (see further investigation of soil mineralogy in Stewart et al. 2025 – in review). For example, sub-humid sites had 15% more percent clay than humid regions, which we expected to impact carbon accumulation potential, as SOC and particularly MAOC storage typically increases with clay content (Laganiere et al., 2010, Johannes et al., 2017).

Our findings demonstrate a strong positive correlation between the MAOC fraction and SOC stock, which is consistent with previous research by Lugato et al., (2021). However, Cotrufo et al. (2019) suggest that higher *f(*MAOC) is often associated with lower SOC, indicating that this relationship may depend on environmental factors such as mineral protection capacity and decomposition dynamics. Across our sites, MAOC fraction was approximately 75%, with little variation by climate. These findings are consistent with the distribution reported by Hansen et al., (2024), and Sokol et al. (2022) who found *f*(maoc) to range from 43% to 91%. However, our results are on the upper limit of findings from Bai and Cotrufo (2022) who found the fraction of MAOC to be between 50%-75% within grasslands. Several factors could contribute to the higher proportion of MAOC observed in our study, including microbial composition and abundance, soil texture, and land management history. The relatively high proportion of SOC in the MAOC fraction could also be an indicator of more recent losses of POC due to rangeland degradation and/or increased drought frequency.

Our results confirm that California's diverse environments are important to consider when implementing conservation management practices, especially perennial seeding. While we were unable to detect overall significant differences in SOC between restored and unrestored sites, a comparative analysis of these practices revealed potential benefits within specific climates. POC stocks in sites with perennial seeding were consistently greater than in both tree-planted and riparian restored sites within dry climates. We suspect that the year-round litter production and accumulation within perennial plants, coupled with the drier climate favoring slower decomposition rates, may have contributed to the greater POC fraction pool. These results are consistent Li et al., (2017), who demonstrated that precipitation significantly influences POC dynamics, particularly within long-term conservation practices.

Furthermore, perennial seeded sites with 6-10 years since adoption exhibited significantly higher POC levels than tree-planted sites within the same duration of practice. This isn't surprising since tree biomass within medium-aged sites was significantly lower than sites older than 11 years. We especially found POC levels to decrease over time in dry tree-planted sites (Figure 2.5), which might be due to limiting precipitation, inhibiting tree biomass growth and ultimately impacting SOC storage. However, at higher elevations, increased water availability may alleviate growth limitations while still maintaining seasonal drought conditions that restrict decomposition in Mediterranean climates (Ortiz et al., 2016).

Canopy cover had a positive impact on both SOC and SOC fraction contents. We found canopy cover to lead to equivalent increase of both %MAOC and %POC, regardless of climate. The similar increase of both POC and MAOC is likely due to greater root and leaf litter inputs under tree canopy. Coarse litter, such as woody debris and roots, contributes to POC, while organic inputs, such as root exudates promote MAOC formation through microbial processing and mineral association (Cotrufo & Lavallee, 2022, Lugato et al., 2021). We found greater %MAOC content in moist climates under canopy (Cotrufo & Lavallee, 2022) while the opposite was found within dry climates, where %POC was higher, attributable to lower levels of decomposition with limited water availability in dry sites, reducing microbial activity (Cotrufo & Lavallee, 2022). Additionally, higher SOC storage under canopies has also been linked to the nutrient island effect, where tree roots concentrate soil nutrients closer to the canopy, leading to increased nutrient availability beneath the canopy (Dalgrehn, 1997; Carey et al., 2020b).

Although our paired-sites did not show overall significant differences in SOC for tree-planted sites versus non-tree-planted sites, the effect of canopy cover (where under canopy showed higher SOC) suggests that mature tree plantings may contribute to increases in carbon storage. While canopy cover is expected to increase over time, the positive impact on SOC may take over a decade to become apparent. This is supported by our observation of a significant increase in tree biomass in trees older than 11 years and by the findings of Ortiz et al., (2016), who studied the conversion of grasslands to 40-year-old forests. Their research found minimal differences in carbon stocks, suggesting that significant changes in soil carbon may take longer than four decades. These results emphasize the need for long-term studies and monitoring to accurately evaluate the effectiveness of tree planting initiatives in conservation practices. Baseline soil carbon data and archived soils from our study can support future research with, for example, repeated soil samples at the same geo-referenced locations in the future, to track direct changes over time.

We also suspect that older California riparian restoration sites may provide further insights on carbon dynamics. The inferred decrease in soil carbon over time within moist riparian restored sites is surprising, especially since revegetation efforts in California’s riparian rangelands have generally been successful in building SOC. This decline may simply highlight the limitations of our site selection and the paired-site approach. For instance, Matzek et al. (2020) reported SOC increases of 1.2 Mg C ha^{-1} $year^{-1}$ in mesic riparian sites within California. The discrepancy between their findings and ours may be due to methodological differences, as their study included sites up to 45 years old and used an unbalanced paired-site design. Additionally, several researchers suggest sites younger than 13 years might not yet show significant SOC increases, as riparian restoration typically reach 90% of their peak biomass after 15 years within arid environments (Smith et al., 2004, Raiesi & Gilani 2020, Follet et al., 2001).

As our results suggest, detecting SOC changes by comparing restored to unrestored sites is not always straightforward. While our analyses indicated that tree plantings and perennial seeding had positive impacts on SOC dynamics when conservation practices were compared to each other, we did not find other significant differences using our paired-site design and analysis.

Is it possible that we did not collect enough soil samples to detect differences between paired-sites? The spatial heterogeneity of rangelands requires a large number of soil samples to accurately measure SOC across ranches. Despite collecting 11 soil samples from each site, we observed a small effect size indicating a difference of only 0.03 standard deviations between restored and unrestored sites. This result suggests that differences between paired-sites were minimal. Increasing the sample size could have improved statistical power by reducing variability. To detect an effect size of 0.5, a minimum of 18 samples would have been required.

However, factors beyond soil properties, such as land use history and legacy effects, also influence SOC storage and may influence the comparability between paired-sites (Paul et al., 2002, Stanley et al., 2023). Therefore, while increasing sample size could improve detection, it may not fully address the influence of land use history. By relying on an adjacent site as a "control" with no prior soil carbon measurements, our research design could have included biases due to legacy effects from unknown differences in past management practices between paired-sites. While we attempted to mitigate this by selecting sites with similar soil types, topography, vegetation, and recent land use histories through farmer interviews, subsequent analyses, as shown in Chapter 3 indicate that differences in long-term land use and disturbance events likely played a significant role. Stanley et al. (2023) identified historical land use as the primary factor influencing SOC, with climate and plant species of secondary importance. Past land management practices, such as tillage and crop cultivation, significantly influence soil properties (Reicosky, 2003, Zhang et al., 2004).

There is also the strong possibility of ranchers choosing to implement conservation practices on their less productive lands. As farmers and ranchers receive limited incentives to implement conservation practices, it is logical that they would prioritize applying these practices to more degraded land in an effort to enhance its future productivity. Thus, the sites where conservation practices were implemented might well have had lower carbon stocks than adjacent less degraded lands that were selected as controls. This uncertainty makes it difficult to isolate the true impact of restoration practices on SOC compared to an adjacent reference site. Remote sensing methods could potentially fill gaps in land management history and productivity where management records are unavailable. These tools might also provide insights into the initial conditions of paired-sites prior to practice implementation.

Conclusion

Our research aimed to quantify the impact of conservation management practices on carbon storage across California's diverse climates. We found that perennial seeding was particularly effective in dry regions, outperforming riparian restoration and tree plantings in POC content.

While tree canopy cover showed promise in increasing SOC, our results suggest that long-term monitoring is essential to fully assess the benefits of tree planting initiatives, which tended to accrue only after 11 years and only in wetter regions. Our paired-site design had limitations due to uncertainty about previous land use history, and thus future studies could benefit from larger sample sizes, and/or remote sensing applications for site selection to address potential confounding factors. By addressing these issues, we can gain a more comprehensive understanding of the role of conservation planting practices in carbon storage and inform effective rangeland management.

CHAPTER 3: Conservation management practice impacts on rangelands in California: Remote sensing applications for assessing land use history

Introduction

Soil degradation poses a significant threat to global food security and ecosystem sustainability, highlighting the need for conservation-based land management strategies. An estimated 33% of the world's soils are degraded (FAO & ITPS, 2015), contributing to a cumulative global loss of 133 Pg of soil carbon, since the beginning of agriculture approximately 10,000 years ago (Sanderman et al., 2017). This decline in soil health hinders agricultural productivity, reinforcing the urgency of adopting sustainable land management practices. Climate change further exacerbates these challenges by increasing the frequency and intensity of extreme weather events, leading to increased soil degradation and erosion (Coant et al., 2014). Conservation management practices have the potential to enhance soil organic carbon (SOC) storage and contribute to long-term ecosystem resilience (Carey et al., 2020; Matzek et al., 2020), particularly given the critical role of healthy soils in climate change adaption and sustained agricultural productivity (Dubey et al., 2019).

While long-term experiments have provided valuable insights into the effects of sustainable management practices on soil health and productivity (Rasmussen et al., 1998), they often lack the breadth and depth necessary to capture the full range of climatic and management variability. To address this limitation, researchers have increasingly turned to paired-site designs, which use retrospective on-farm 'experiments' (Guo and Gillford, 2002). This design relies on comparing a treated site, where a specific management practice has been implemented, with a nearby untreated control site that is assumed to be very similar except for the management treatment. This approach allows for the assessment of long-term management impacts without requiring the establishment of new, long-term experiments and aims to capture the variability of working landscapes.

The control site is typically selected to have similar vegetation and prior management practices, soil type, topography, and microclimate as the treated site, serving as a baseline for comparison. However, while many studies consider land use history of the selected sites, this is primarily based on current land use or landowner knowledge (Brown et al., 2018; Keith et al., 2014). Only a few studies explicitly reference the use of written records or imagery to determine longer term historical land use (Sitzia et al., 2012; Kabasinskiene et al., 2021). While soil type (e.g. soil series from existing soil maps) is commonly controlled for in paired-site studies (Liu et al., 2017; Breuer et al., 2006; Gu et al., 2024), fewer studies explicitly control for soil texture (Vanden et al., 1999; Chang et al., 2012). Similarly, while most studies consider topographic factors (slope, aspect, elevation), the specific methods used to quantify these factors are often not explicitly described (Wellock et a., 2011; Ricker et al., 2014; Feyisa et al., 2017). While these designs strive for similarity in site characteristics, uncertainties often remain regarding the historical land use of sites.

Previous land use history and management can include past disturbance events such as fires, tillage, or vegetation removal, that can significantly influence soil properties for decades after the occurrence (Paul et al., 2002; Stanley et al., 2023; Krause et al., 2020). These practices can alter soil properties, structure, and nutrient availability, creating lasting impacts that persist well after the disturbance event. Recognizing and accounting for legacy effects is essential for accurately assessing the impacts of conservation practices on soil health and ecosystems. Although researchers can mitigate these challenges by carefully selecting control sites, accounting for confounding factors, and obtaining management history from land managers, a thorough investigation of historical land use is often lacking. This can potentially introduce a lack of comparability in SOC levels (Oliver et al., 2004). This highlights the critical need for careful historical land-use assessments in paired-site studies.

One promising approach for addressing this challenge is the use of remotely derived satellite imagery in site selection processes. Agricultural research has increasing adopted the use of remote sensing tools to analyze the impact of management practices on vegetation health (Allred et al., 2022; Weiss et al., 2020). Despite the growing application of these tools for monitoring agricultural and rangeland conditions, a significant gap remains in their use for accounting for historical management practices during paired-site selection for ecological studies. Remotely derived data, including topographic variables and vegetation indices, can provide valuable information for making informed decisions in site selection.

This study aims to explore the utility of these tools by evaluating the suitability of paired sites used in Chapter 2, which assessed the impact of three conservation planting practices on SOC. The retrospective study revealed no significant differences in SOC between restored and unrestored sites across 36 paired sites. Several findings suggested that at least some of the paired sites may not have had sufficient similarity in site characteristics prior to the conservation practice adoption. To assess whether paired sites had substantially dissimilar initial vegetation and soil conditions, we analyzed over 30 years of remotely-derived net primary production (NPP) data from the Rangeland Analysis Platform, and other indices to identify past disturbance events and potential differences in degree of land degradation, that could influence SOC stock. NPP is an indicator of biomass production and carbon fixation therefore a major control on the magnitude of soil carbon stocks (Paustian et al., 1997), potentially accounting for differences in SOC between site pairs members prior to practice implementation.

Despite the growing use of remote sensing in ecological studies, few studies have analyzed historical imagery to assess site comparability. This study aims to demonstrate the use of remote sensing tools for quantifying initial site conditions in paired-site studies. By refining methods for assessing paired-site suitability, this research contributes to the development of more robust ecological studies and supports sustainable land management research.

Materials and Methods

Site Selection Design

This study builds on our initial research using a dataset of 36 paired-sites (restored and unrestored) across California rangelands, selected to evaluate the impact of three conservation planting practices – riparian restoration, perennial seeding, and tree planting – on SOC dynamics (Supplemental Table 3.1). These sites encompassed a diverse range of California's climates and soil types, as detailed in Chapter 2 Each pair consisted of a restored and unrestored site (Figure 3.1), with the latter chosen for its similarity in climate, topography, grazing management practices, and soil characteristics to the restored counterpart but without the adoption of a conservation practice. Restored sites were selected across a spectrum of practice durations to allow for a chronosquence analysis, with conservation practices implemented between 2 to 27 years prior. Unrestored sites were used as a baseline for SOC, serving as a proxy for the restored site's initial conditions before practice implementation. This pairing process assumed that both restored and unrestored sites had comparable initial SOC levels and shared similar historical management practices at the time of practice implementation.

Figure 3.1: Google Earth image from May 30th,2024. Example of a typical paired-site design from our study. Paired-sites were typically within 500m from each other and encompassed a 20x20m perimeter transect. This pair is a tree-planted site planted in 2002, with restored and unrestored sites labeled with “R” and “U”, respectively.

Remote Sensing Data Acquisition

To evaluate the similarity of historical management practices and site characteristics between members of each paired-sites, we used several remote sensing datasets, including net primary productivity (NPP), bare soil index (BSI), normalized difference vegetation index (NDVI), potential evapotranspiration (PET), and topographic variables such as slope, aspect, and elevation. The majority of remote sensing data was acquired and processed through Google Earth Engine (Table 1). NPP data was sourced from the Rangeland Analysis Platform (RAP), which was derived from models integrating Landsat 5 TM, Landsat 7 ETM+, and Landsat 8 OLI to estimate annual NPP values at a 30-meter spatial resolution. NPP values were extracted for each site, spanning the years 1984-2022. BSI and NDVI were calculated using spectral bands from Landsat 5 TM and Landsat 7 EMT+ (Level 2, Collection 2, Tier 1) with corrected surface reflectance. Additionally, elevation, slope and aspect were derived from digital elevation models (DEM) at the same 30-meter spatial resolution, providing insights into the topographic variability between sites.

Table 3.1: Remote sensing data collection sources and where applicable Google Earth Engine Image collection ID.

Data Type	Source	GEE Image Collection ID	Reference
NPP (Net Primary Productivity)	RAP (Rangeland Analysis Platform)	projects/rap-data-365417/assets/npp-partitioned-v3	Jones et al., 2021
BSI (Bare Soil Index)	Landsat 5 and Landsat 7	ee.ImageCollection("LANDSAT/LT05/C02/T1_L2") ee.ImageCollection("LANDSAT/LE07/C02/T1_L2")	Rikimaru et al., 2002
NDVI (Normalized Difference Vegetation Index)	Landsat 5 and Landsat 7	ee.ImageCollection("LANDSAT/LT05/C02/T1_L2") ee.ImageCollection("LANDSAT/LE07/C02/T1_L2")	Deering & Eck, 1975

Potential Evapotranspiration (PET)	WorldClim	N/A	Trabucco and Zomer, 2018
Slope	Digital Elevation Model (DEM)	ee.Image("USGS/SRTMGL1_003")	Rabus et al., 2003
Aspect	Digital Elevation Model (DEM)	ee.Image("USGS/SRTMGL1_003")	Rabus et al., 2003
Elevation	Digital Elevation Model (DEM)	ee.Image("USGS/SRTMGL1_003")	Rabus et al., 2003
Image Validation	National Agricultural Imagery Program (NAIP)	ee.ImageCollection('USDA/NAIP/DOQQ')	United States Geological Survey (USGS)
Image Validation	Google Earth	N/A	Map data ©2022 Google

NDVI and BSI calculations

We used two indices, normalized difference vegetation index (NDVI) and bare soil index (BSI) to assess initial site conditions prior to conservation implementation. NDVI, a widely used vegetation index, measures plant greenness and is sensitive to changes in vegetation health and density (Sommer et al., 2002). Values range from -1 to 1 with lower values indicating reduced productivity and higher values representing healthier vegetation (Deering & Eck, 1975). BSI quantifies bare ground cover, providing insights into soil exposure and vegetation absence. Like NDVI, BSI values range from -1 to 1, where lower negative values indicate greater vegetation coverage, while higher positive values signify increased soil exposure (Rikimaru et al., 2002).

To evaluate long-term trends in site conditions, we analyzed NDVI and BSI data over a 36-year period (1984-2022), concluding with soil sample collection. However, to ensure that only pre-restoration conditions were considered, our analyses were restricted to the time period preceding each site's conservation practice implementation. This generally covered the period from 1984-2020, depending on the year of implementation.

Both NDVI and BSI were calculated using spectral bands derived from Landsat 5 TM (1986-1999) and Landsat 7 ETM+ (2000-2022). These indices were calculated in Google Earth Engine using the respective bands for each sensor. To improve accuracy and minimize biases from cloud inference, cloud-masking filters were applied prior to index calculation.

NDVI equation:

$$\frac{NIR - Red}{NIR + Red}$$

BSI equation:

$$\frac{(SWIR + Red) - (NIR + Blue)}{(SWIR + Red) + (NIR + Blue)}$$

Detecting disturbance events through NPP trends and image analysis

To identify potential disturbance events and further investigate NPP variability between our paired-sites, we analyzed a time-series of annual NPP differences (kg C ha^{-1} yr-^{1}) between paired-sites (restored minus unrestored). Significant deviations from long-term NPP trends between paired sites were identified as potential indicators of disturbance events, such as fire, tillage, or other shifts in management practices. These anomalies, characterized by unusually high NPP differences, were interpreted as evidence of vegetation changes within one or both sites, potentially indicating a disturbance event.

To further investigate these NPP anomalies, we visually analyzed high-resolution imagery from Google Earth Pro and the National Agriculture Imagery Program (NAIP). These datasets were chosen for their high spatial resolution of approximately 1 meter, which is essential for identifying fine-scale environmental changes. Given that each study site spans roughly 80 meters, this level of detail was necessary to ensure accurate temporal analysis. These imagery sources also offer historical archives. NAIP imagery date back to 2002, while Google Earth Pro incorporates data from multiple satellite sources, such as Landsat, providing coverage dating back to 1984. However, it’s important to note that imagery from the 1980s and 1990s may exhibit lower spatial resolution in some cases.

Statistics

To examine the relationship between NPP and %SOC, we performed simple linear regression. The analysis included data from all conservation practices, utilizing %SOC measurements (n=792) and average annual NPP values over the past 36 years. Prior to the analysis, data was visually inspected for normality and homogeneity of variance using the *performance* package in R Studio (version 4.4.0), as well as for the remainder of the analyses. Log transformations were applied to %SOC to meet assumptions of linearity and homoscedasticity. This regression aimed to evaluate the suitability of NPP data as a potential indicator of soil carbon disturbance.

Alongside visual inspection, we applied the Interquartile Range (IQR) method, as described by Tukey (1977), to detect outliers in NPP differences. The IQR method involved dividing our dataset into four equal parts, where Q1 represented the 25th percentile and Q3 represented the 75th percentile. To evaluate the sensitivity of the outlier detection, we tested both the default IQR multiplier of 1.5 and a less stringent multiplier of 1. Data points exceeding these thresholds from the first or third quartiles were considered potential disturbance events within one or both sites. The results from this analysis were compared to disturbance events identified by visual assessment.

IQR equation.

$$Q1 - 1.5\ X\ IQR$$

$$Q3 + 1.5\ X\ IQR$$

To assess the influence of environmental variables on the observed variation between paired sites, a Constrained Analysis of Principal Coordinates (CAP) was performed. This analysis incorporated a laboratory measurement of combined percentage of silt and clay, alongside remotely derived variables: elevation, aspect, slope, PET, and NPP measured prior to the implementation of any conservation practice. Variables were chosen based on the most significant model that captured the most variability. The analysis was constrained by treatment (restored vs unrestored) to measure the variability between paired-sites. To quantify the influence of environmental variables, we calculated the position of each site along the constrained axis and determined the absolute difference for each site pair, using these values as a score.

To categorize the scores for each paired-site, values were grouped into three clusters – minimal, moderate, and substantial – using the K-means clustering algorithm from the *cluster* package in R. These clusters indicated the degree of variability between the paired sites. Within the minimal difference cluster, where paired sites showed little variation in the selected variables, we evaluated the presence of pairs with known disturbances. This was done to assess the sensitivity of variables used in the analysis to these events.

Additionally, to assess the initial conditions of paired-sites, we performed several iterations of linear mixed-effects models using up to 37 years of annual NPP, NDVI, and BSI data. The number of years used in these analyses were dependent on the year of implementation for each pair. First, we evaluated suitability by conservation practice types (e.g., riparian restoration, perennial seeding, tree planting) using treatment (restored or unrestored) as a covariant. Year, ranch, and sites were treated as random effects to account for variability within and between paired sites across time. This analysis aimed to provide a general understanding of the similarity of paired sites within each specific conservation practice.

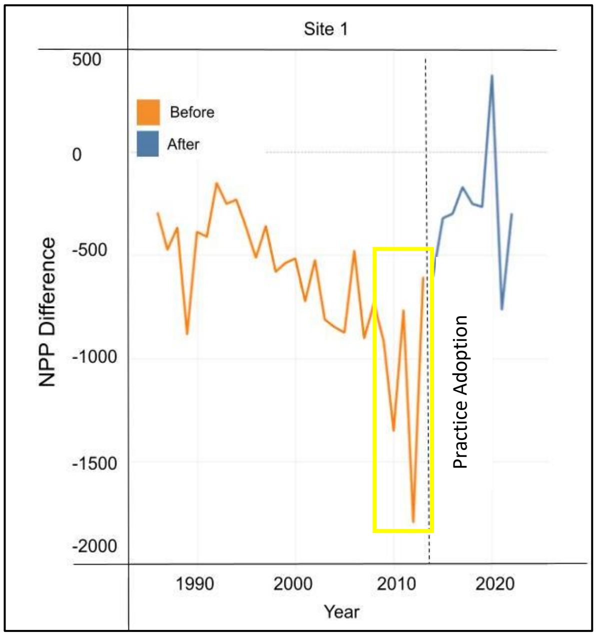
Second, we assessed the suitability of each individual pair by analyzing the initial vegetation health and cover between restored and unrestored sites prior to the implementation of the conservation practice. In this analysis, treatment was included as a covariate with an interaction with ranch to account for potential differences in paired sites across ranches. Year was included as a random effect to account for the variability in NPP, BSI, and NDVI over time due to annual weather fluctuations and drought periods. The results from these analyses were used to further investigate the relationship between land degradation and %SOC changes.

Results

Visual assessments of paired sites

Analyzing fluctuations in NPP and imagery over a 30-year period across the 36 pairs of rangeland sites provided valuable insights into historical management differences between them. For detecting disturbance events, strong temporal fluctuations in NPP between members within paired sites were generally indicative of differential disturbances that were corroborated by imagery analysis. However, three disturbance events (one mastication event and two burn events) were not detected by NPP data alone. Overall, visual inspection identified disturbances in 14 pairs: seven within perennial plantings, two within riparian forest, and five within tree-planted pairs. These disturbances included mastication, tillage, heavy grazing, or burning events, often occurring within only one of the paired-sites members. In some cases, similar disturbances occurred in both sites (Supplemental Table 3.2), but with varying severity. Three of the most notable differences between pairs are described in the next section.

A)



B)

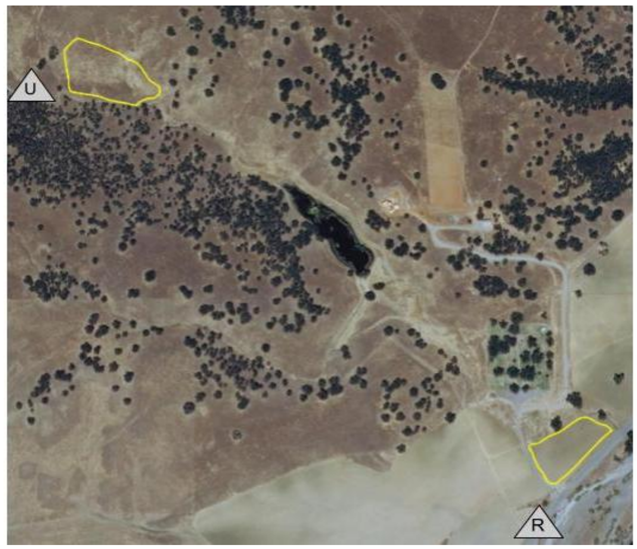
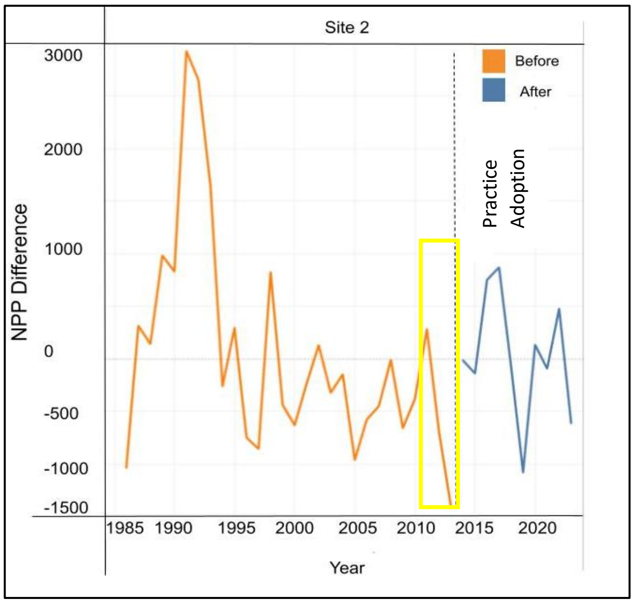


Figure 3.2a: Variability in annual NPP differences (restored-unrestored) in kg C ha^1 yr^1 for a perennial planted site. Prior to practice implementation, the restored site exhibited overall lower productivity, as indicated by NPP differences values below 0. A sharp decline in NPP around 2010 suggested a potential disturbance event.

Figure 3.2b: Google Earth Pro image from August 2nd, 2010. The observation in NPP difference from 2010 was corroborated by site imagery, which revealed evidence of tillage and differences in the restored (R), while the unrestored site (U) showed no signs of disturbance.

A)



B)

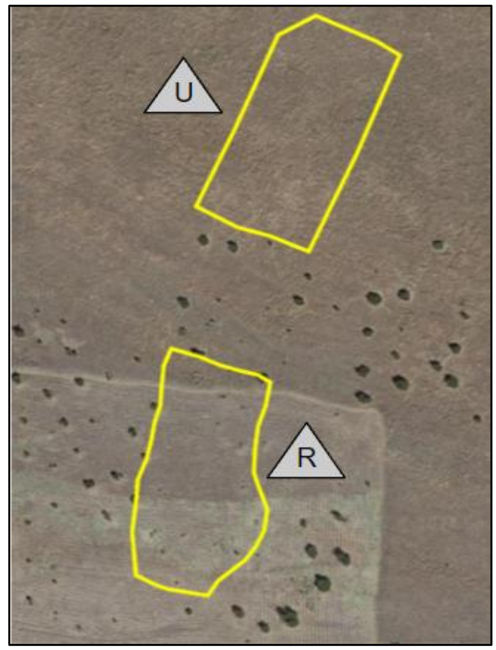


Figure 3.3a: In 1991-92, a much greater NPP difference (~2,900 kg C ha-^{1} yr-^{1}) in the sites which was later selected for tree planting in 2014 (R), suggested a potential disturbance event within the unrestored site (U). However, the absence of high-resolution imagery from 1992 limited the ability to identify the cause of this difference. However, between 1998 and 2014, there was lower NPP in the (restored) site prior to the conservation practice (tree planting) adoption.

Figure 3.3b: Google Earth Pro image from August 30th, 2013. Practices such as mowing, tillage or heavy grazing, are suggested by sparser vegetation and distinct disturbance patterns in the restored site, prior to conservation practice adoption, that were absent in the unrestored site.

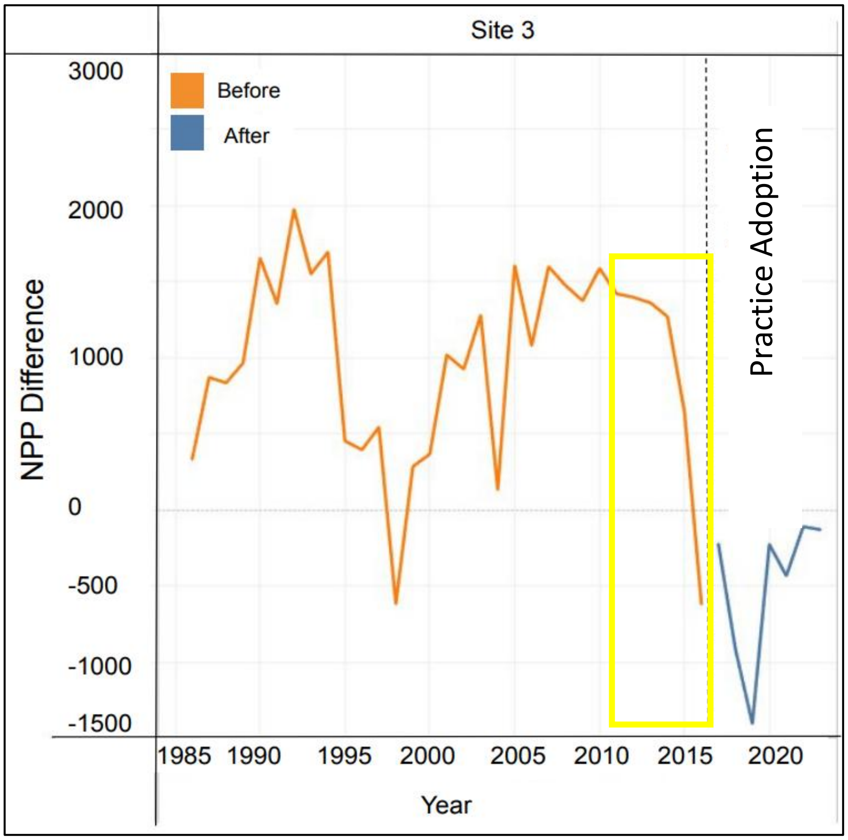


Figure 3.4a: Prior to the adoption of perennial planting in 2017, the restored site (R) already had higher NPP than the unrestored site (U). However, beginning in 2014 a rapid decrease in NPP difference was observed, indicating a possible disturbance event.

Figure 3.4b: Google Earth Pro images from April 30th, 2015 (left) and June 30th, 2017 (right). 2015 image reveals a significant presence of shrubs within the restored area. By June 2017, shrubs were removal within the restored site, while the unrestored site remained undisturbed, showing no evidence of vegetation removal or disturbance. This suggests a mastication event likely occurred in 2015, potentially contributing to the decrease in NPP within the restored site.

Relationship between NPP and Soil carbon

We observed a significant positive relationship between soil organic carbon and NPP across the study sites ($P<0.001$). However, the model explained only 23% of the variability in SOC ($R^2= 0.23$), indicating a relatively weak explanatory power. Nonetheless, examining differences in NPP between restored and unrestored sites helped assess relative degree of range degradation and identify disturbance events that may have influenced initial soil carbon levels between paired sites.

Comparing outlier computed results to visual assessment

The IQR method detected most disturbance events initially identified through visual inspection of NPP dynamics. Lowering the multiplier from 1.5 to 1 more than doubled the number of detected outliers increasing from 59 to 119 across all sites – a 102% increase. Comparing these results to visually identified outliers revealed that the lower multiplier (1) more effectively captured dates previously flagged as outliers. Despite its improved sensitivity, this approach failed to detect certain disturbance events, including some burning events and the mastication event at site 3 (Figure 3.4b). Additionally, it highlighted differences in vegetation cover between sites that were not previously identified by visual NPP inspection.

Soil and remotely derived properties impact on paired sites

The multivariate analysis of soil properties and remotely derived topographic and forage health variables explained 3.6% of the total variance between paired sites, whereas unconstrained variability across all sites accounted for 96%. The CAP ordination plot (Figure 3.5) shows that variability between paired-sites, rather than within-site variation contributed most to the observed spread. This is evident by the spread of paired sites across the y-axis. However, marginal significance (P=0.07) of the PERMANOVA performed on the CAP results between restored and unrestored plots indicates some sites had pre-existing differences in soil, topography, and NPP, evident in the x-axis spread (Figure 3.5). The pre-existing differences violate the assumption of similar conditions before conservation practice, suggesting inadequate pairing in some sites.

Among the environmental variables included in the CAP analysis, NPP emerged as the dominant factor explaining 94% of the observed variance within the constrained model between paired sites, followed by aspect accounting for 3%, while percent clay and silt contributed 1% (Table 3.2a). These findings highlight the strong influence of forage productivity and topographic properties, particularly aspect, on site comparability. Although soil texture played a comparatively smaller role, it still contributed to observed differences. In contrast, variables such as elevation, slope, and PET had minimal influence on paired-site differences, but still contributed to the overall unconstrained variability across all study sites.

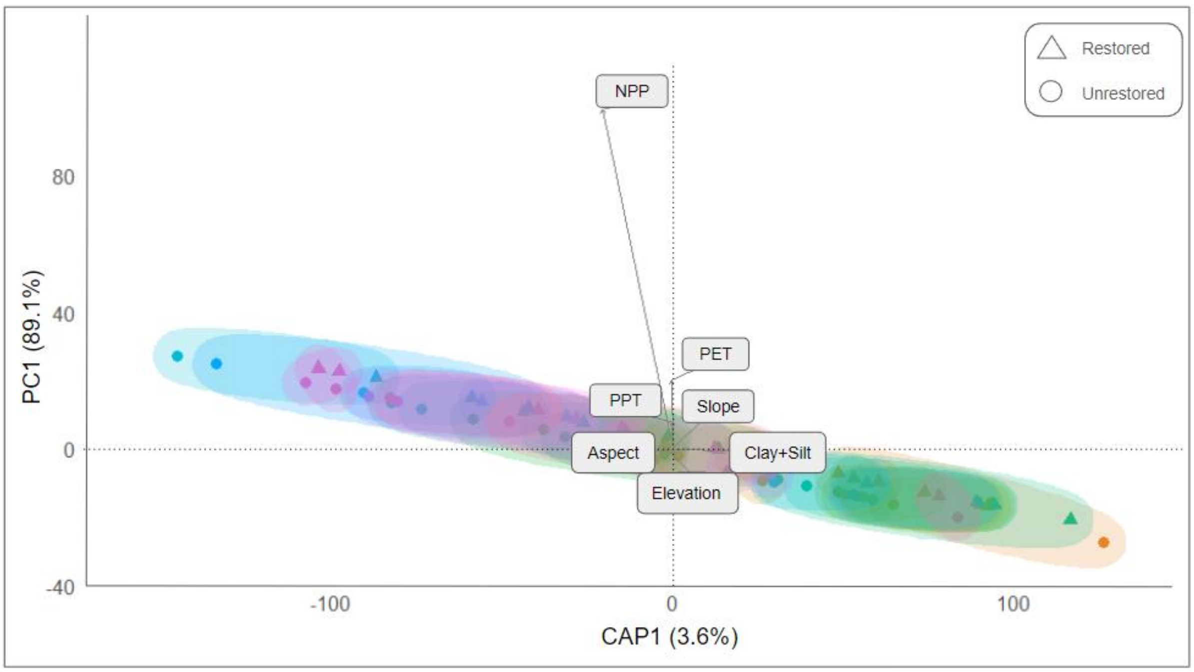


Figure 3.5 A Constrained Analysis of Principal Coordinates (CAP) of soil properties and remotely derived variables of the 36 paired sites, constrained by treatment. Only significant variables are shown in the graph (NPP, aspect, elevation, PET: potential evapotranspiration, PPT: mean annual precipitation, and percent clay plus silt). Variables are displayed using repel function to better visualize the significant factors. Each ellipsis is colored by individual pairs with a restored and unrestored site.

After each paired sites' score were calculated, pairs were categorized into three clusters based on the magnitude of environmental differences between pair members: minimal, moderate, and substantial. Of the 36 paired sites analyzed, 15 pairs were classified as having minimal differences, 13 exhibited moderate differences, and eight displayed substantial variation between restored and unrestored sites (Table 3.2a). Notably, tree-planted sites were most represented in the substantially different cluster, with five pairs presenting considerable variation, followed by riparian sites (Table 3.2b). In contrast, none of the perennial sites revealed substantial differences between paired sites based on our analysis.

Despite the majority of paired sites falling into the minimal difference cluster, eight of the 15 pairs within this cluster experienced disturbances at one or both of the pair member sites. These disturbances, including mastication, fire, and tillage events, were not incorporated into the CAP analysis. Therefore, they were not considered in the cluster categorization. For example, an analysis of a perennial planted pair within the minimal differences cluster revealed disparities in historical management practices dating back to the 1980s and 1990s, as evidenced by historical NPP data and imagery. However, these disturbances were not detected in the CAP analysis. Interestingly, five of the eight pairs with unincorporated disturbance events were located within perennial planted areas, including Site 1, as shown in in Figure 3.3.

Table 3.2a: Constrained Analysis of Principal Coordinates (CAP) of paired sites under different soil and remotely derived topographic and vegetation health properties. **Table 3.2b:** The number of sites across conservation practice and across all paired sites that fall into each category of minimal, moderate, and substantial differences based on CAP results.

a) Variable	**CAP1**	**(%) of variance**
NPP	-20.5	94.4
Aspect	-0.66	3
Clay+silt	0.25	1.1
Elevation	-0.18	0.8
PET	-0.1	0.5
Slope	-0.01	0.05

b)	**Minimal**	**Moderate**	**Substantial**
Riparian	5	4	3
Range	8	4	0
Tree	2	5	5
All sites	15	13	8
% Of sites	42%	36%	22%

Pre-practice variability in ecosystem function across paired sites

The linear mixed-effects model revealed no overall significant differences in NPP, NDVI, or BSI before practice implementation by conservation practice. However, significant inter-site variability was observed within paired sites. Prior to practice implementation, 15 paired sites exhibited significant differences in NPP between restored and unrestored sites ($P<0.05$). In 11 of these pairs, restored sites displayed lower NPP values compared to unrestored sites, with within pair differences ranging from 313 to 1698 kg C ha^{1} yr^{1}. Conversely, four paired sites showed higher NPP values, prior to conservation practice adoption, within restored sites, with within pair differences ranging from 367 to 1724 kg C ha^{1} yr^{1}.

Similar patterns were found analyzing NDVI and BSI observations prior to the date of conservation practice adoption. Significant differences in NDVI between pair members were observed in 16 paired sites ($P<0.05$). Among these, 12 restored sites exhibited lower NDVI values compared to unrestored sites, with lower values ranging from 7% to 18%. In contrast, the remaining four pairs showed higher NDVI for restored sites, ranging from 5% to 25%. Significant differences in BSI were also observed in 15 paired sites ($P<0.05$). Of these, eight restored sites exhibited higher BSI values, indicating greater levels of bare soil compared to their unrestored counterparts. Interestingly, eight paired-sites exhibited significant differences across all three variables (NPP, NDVI, and BSI), while seven pairs differed in two variables, and six pairs showed differences in only one variable.

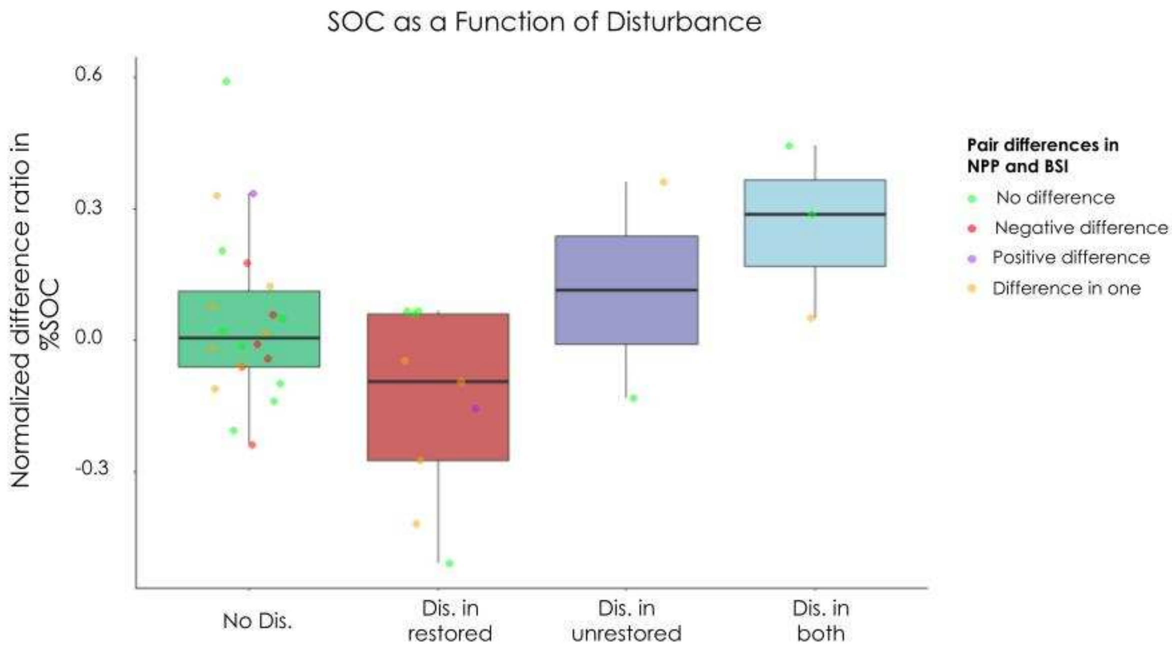


Figure 3.6: Distribution of normalized SOC (%) difference ratio (restored-unrestored/unrestored) by disturbance across paired-sites. Categorized by disturbance history: (1) No disturbance (No dis.), (2) disturbance within restored site (Dis. in restored), (3) disturbance within unrestored site (Dis. in unrestored), and disturbance within both sites (Dis. in both). Each point is colored based on significant differences between pairs by NPP and BSI. "Positive difference" indicates the restored site had higher NPP and ground cover (negative BSI difference). "Negative difference" indicates the restored site had lower NPP and ground cover (positive BSI difference). "Difference in one" reflects that either NPP or BSI was lower or higher or no difference in one of the two.

The distribution of normalized %SOC differences (i.e. equivalent to a response ratio for conservation practice adoption) across disturbance categories (Figure 3.6) reveals expected trends. Sites where disturbances occurred within the restored area only exhibited the lowest %SOC differences with the mean response ratio being negative, while those with disturbances in the unrestored area showed higher %SOC values. Notably, sites with disturbances in both the restored and unrestored sites exhibited the greatest normalized SOC differences with an implied response ratio of 0.3. However, there were only three site pairs with significant disturbance events occurring in both pairs. In contrast, sites without any recorded disturbance showed, on average, no difference in the normalized SOC difference between paired sites.

However, a closer examination of site pairs in which there were no significant disturbance events (since 1984) reveals that all six sites with lower soil cover and vegetation production fell within this category. Contrary to expectations, paired sites where restored sites had lower NPP and higher BSI (negative difference), corresponding with lower vegetation cover, did not fall within any of the identified disturbance groups (colored in red in Figure 3.6). This suggests that long-term data averaging can mask disturbance effects on bare soil and vegetation health, especially as some paired sites had data over 20 years.

Discussion

While the weak correlation between SOC and NPP suggests that additional factors significantly influence soil carbon stocks, the positive relationship between them reinforces the role of NPP as a driver of SOC inputs. Soil carbon dynamics are shaped by a complex interplay of climate, vegetation type, soil texture, and land use history (Cotrufo and Lavallee, 2022; Zhou et al., 2024; Carey et al., 2020). Despite this complexity, Hansen et al. (2024) identified NPP as one of the dominant controls on the mineral-associated carbon fraction globally, a stable form of SOC that plays a critical role in long-term carbon storage (Cotrufo and Lavallee, 2022). This is because an increase in NPP directly translates to increased plant biomass production, leading to greater inputs of organic matter in the form of roots and litter, contributing to SOC storage (Bai and Cotrufo, 2022). This further supports the use of NPP as a valuable indicator for inferring soil carbon dynamics and soil disturbances (Chen et al., 2021; Cotrufo and Lavallee, 2022).

Although NPP data correlated with the majority of disturbance events, it failed to detect some instances of burning and mastication events, which can significantly impact soil carbon dynamics (Turner and Lambert, 2000). Instead, we found that visual inspections of imagery provided a more comprehensive evaluation of site comparability (Supplemental Table 3.2). This may be attributed to a limitation in the RAP model's ability to capture short-term disturbances (Jones et al., 2021) stemming from its focus on assessing long-term productivity. Since the model emphasizes annual NPP, it has reduced sensitivity to finer scale interannual variations indicative of disturbances. For example, a localized fire could have minimal effects on annual site-level NPP, especially if vegetation recovers within the same year. These limitations suggest that remotely derived NPP data alone may not be sufficient for detecting variations in site attributes. We recommend supplementing NPP data with image analysis to enhance the understanding of historical management practices.

Integrating these disturbances into the CAP analysis was limited by the availability of high-resolution imagery, hindering the systematic verification of all suspected disturbance events. This limitation consequently restricted the full intergradation of disturbance events into the analysis. The selected variables (NPP, aspect, elevation, slope, PET, and clay and slit) were not sensitive enough to effectively distinguish between sites with and without disturbance histories. This is evident by the presence of pairs with known disturbances within the minimal difference cluster. This finding suggests that these variables were not adequate to capture the ecological impacts of past disturbances, despite NPP demonstrating potential for detecting disturbance events. To address this, future studies should consider incorporating other remotely derived vegetation health indicators that may be more sensitive to disturbance events, such as burn severity index (BSI), normalized difference tillage index (NDTI), or enhanced vegetation index (EVI).

Although the multivariate analysis did not distinguish disturbance events, we found aspect and soil texture to play significant roles in distinguishing site characteristics, alongside NPP as the dominant variable driving differences between paired-sites. Direct comparisons of SOC between sites with substantially different aspect and/or soil-texture can introduce bias and obscure the true impact of conservation practices, potentially leading to inconclusive results, as demonstrated by our initial research in Chapter 2. This is due to the strong influence of aspect on SOC stocks, as it affects temperature, moisture, and microclimates (Griffiths et al., 2009). In general, north and east facing aspect tend to have higher SOC storage compared to south and west facing aspect, in some cases up to 3.2 times greater (Lenka et al., 2013; Yohannes & Soromessa, 2015; Zhu et al., 2017). Additionally, soil texture, particularly clay and silt content are critical for SOC storage as these fractions promote the mineral-associated fraction of SOC, by providing the necessary surfaces for organo-mineral associations (Cotrufo & Lavallee, 2022; Haddix et al., 2020; Hassnik et al., 1997). Therefore, careful consideration of aspect and soil texture during site selection are essential for accurately assessing SOC dynamics and the effectiveness of conservation practices.

Our findings highlight that confirming soil texture between paired sites is crucial when selecting study sites. The assumption that soil properties (including SOC stocks) of paired plots, based on similar topography and soil type (as determined from soil maps), were very similar before conservation practice implementation is often unrealistic. Breuer et al., (2006) incorporated geographic information systems (GIS) tools and aerial imagery to assess land-use history for site selection. However, despite these efforts, significant differences in soil properties persisted between paired sites, leading them and other researchers to advocate for direct measurements of soil properties prior to site selection (Poeplau & Don, 2013). Many studies have either assumed similar soil characteristics based on proximity or relied on soil maps for texture classification (Brown et al., 2018; Kucharik, 2007; Novara et al., 2012) and in some cases failed to report how soil texture was determined (Alberti et al., 2011; Barnett et al., 2014; Lockwell et al., 2011). Soil properties and management practices have been shown to vary as the distance between paired sites increases (Poeplau & Don, 2013, Breuer et al., 2006). This suggests that integrating prior soil sampling with remote sensing applications would enhance researchers' ability to select truly comparable sites for paired-site studies in soil research. This approach minimizes confounding variables, strengthening the reliability of SOC assessments and the evaluation of conservation practices.

Beyond soil properties, inherent site differences can also extend to historical difference in range condition/degradation degree (as indicated by vegetation productivity and bare ground cover), further influencing paired-site comparability. A key finding of our study is that restored sites were substantially more degraded than unrestored sites before conservation efforts began, as evidenced by their lower productivity (NPP and NDV), higher bare ground, and identified soil disturbances. This initial disparity highlights a potential selection bias, with ranchers focusing on restoration on their most degraded areas. As a result, the selection of truly comparable unrestored controls is difficult, as these initial differences can obscure the true impact of restoration. This finding is consistent with other results from our previous study in Chapter 2, which reported lower herbaceous biomass in riparian restored sites compared to unrestored sites. Similarly, our observation of four restored sites with greater bare ground and lower vegetation production, but without significant disturbance events (Figure 3.6), having lower SOC than their unrestored counterpart, further supports a selection bias.

This concept also applies to a single disturbance event such as tillage which could have long-lasting effects on SOC while not significantly impacting longer term biomass productivity. These findings emphasize the importance of considering not only disturbance history, but also the overall vegetation health and productivity when selecting controls in paired-site studies. Failing to do so may obscure the true long-term impacts of conservation practices on SOC dynamics.

Our results highlight the potential of integrating remote sensing tools with field measurements to assess the impact of conservation practices on ecosystem health. These tools can not only identify historical soil disturbances and vegetation conditions for paired-site selection but also analyze ecosystem-level trends over time. Future research could build on this work by examining annual NPP and NDVI variability on restored sites to quantify vegetation recovery following implementation. Since our carbon measurements were taken at a single time point and do not capture temporal changes, incorporating NPP and NDVI could provide a more comprehensive assessment of conservation practice effects on vegetation health and SOC storage. While field measurements provide valuable point-specific SOC data, integrating remote sensing applications allows us to extrapolate historical land conditions across broader landscapes, bridging the gap that field data alone cannot address.

1. Conclusion

Our study underscores the importance of considering longer term land use history, including distinct disturbance events and potential differences in past range conditions, when selecting paired sites for assessing impacts of conservation practice adoption on SOC stock changes and biomass productivity. While historical trends in NPP emerged as a potential important difference between members within a site pair, additional within-pair differences in variables such as aspect, actual soil texture, and disturbance events impacted the inferences made on SOC stock change. For pairing in which we found evidence of historically lower NPP and poorer range conditions on the sites selected for restoration, the inferred response to conservation practices adoption was often negative, i.e., adoption of the conservation practice 'resulted' in decrease in soil carbon stocks. Of course, the likely explanation is that the pair member receiving the conservation practice started out with a lower SOC stock than the unrestored pair member. Hence even if the conservation practice adoption led to net accrual of SOC it might take many years for the restored site to 'catch up to' the unrestored site. Conversely, when soil disturbance or lower productivity was observed in the unrestored sites, the restored site showed, on average, a positive response ratio of SOC, indicating an increase in SOC relative to the unrestored site.

The identification of disturbance events and productivity differences through remote sensing highlights the need for rigorous site selection to avoid confounding factors that may skew SOC assessments. Given that restored sites were often more degraded before conservation efforts, selecting appropriate unrestored controls remains a challenge, as mismatches in initial conditions can obscure true treatment effects. Combining automated screening approaches, using remote sensing, with on-site validations may enhance the selection of appropriate study sites. Further, incorporating direct soil measurements– particularly for soil texture analysis– alongside remote sensing and historical land-use data can further refine site comparability. Future studies should integrate these methods to improve the evaluation of conservation practices, ensuring more accurate assessments of their long-term impact on SOC storage and rangeland resilience.

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APPENDIX

Supplemental Table 2.1. *This table presents the distribution of soil and herbaceous biomass properties by aggregate data of restored versus unrestored sites*

Restored/ Unrestored	(%) TOC	(%) MAOC	(%) POC	SOC Stock	MAOC Stock	POC Stock	*f(maoc)*	Biomass	(%) Clay
Restored	1.77 ±0.05	1.35 ±0.05	0.44 ±0.01	65.7 ±1.52	49.3 ±1.39	16.7 ±0.49	0.72 ±0.08	280 ±6.83	23.8 ±0.6
Unrestored	1.82 ±0.05	1.37 ±0.05	0.46 ±0.01	69.3 ±1.57	52 ±1.4	17.9 ±0.48	0.72 ±0.08	334 ±8.99	23.8 ±0.6

Values are mean ± standard error. Bulk density is expressed as g cm^{-3}, and SOC stocks are expressed as Mg ha^{-1} *from 0 to 30 cm depth. Biomass is defined as standing herbaceous biomass (oven-dried) in g m^{-2}. f(maoc) is defined as the fraction of MAOC relative to total SOC.*

Supplemental Table 2.2. This table displays the distribution of tree biomass in Mg ha^{-1} across tree plantings sites by age group and by climate. Young sites do not have a standard deviation since there is only one young site in our analysis. Moist climate includes sub-humid and humid sites.

Age group	**Mean ± SD**	**Climate**	**Mean ± SD**
Young	0.2 ± 0	**Dry**	9.07 ± 18.45
Medium	0.43 ± 0.64	**Moist**	5.03 ± 5.47
Old	17.06 ± 20.73		

Supplemental Table 3.1. List of pairs with according to conservation practice, year of practice implementation and name of county.

Pair	Conservation Practice	Year of Implementation	County
1	Perennial Planting	2009	Monterey
2	Perennial Planting	2004	Yolo
3	Tree Planting	1999	Monterey
4	Riparian Restoration	2019	Tulare
5	Perennial Planting	2012	Sonoma
6	Riparian	2008	Tulare
7	Tree Planting	2015	San Benito
8	Tree Planting	2018	San Benito
9	Tree Planting	2016	Tehama
10	Riparian Restoration	2018	Tehama

11	Perennial Planting	2014	Tehama
12	Riparian Restoration	2018	Sonoma
13	Riparian Restoration	2007	Butte
14	Riparian Restoration	2020	Butte
15	Riparian Restoration	2018	Sonoma
16	Tree Planting	2002	Tehama
17	Tree Planting	2014	Tehama
18	Riparian Restoration	2014	Solano
19	Tree Planting	2015	Solano
20	Riparian Restoration	2015	Marin
21	Riparian Restoration	2016	Fresno
22	Riparian Restoration	2000	Marin
23	Tree Planting	1995	Yolo
24	Riparian Restoration	1993	Marin
25	Perennial Planting	2007	Yuba
26	Perennial Planting	2017	Yuba
27	Perennial Planting	2017	Shasta
28	Perennial Planting	2017	Solano
29	Tree Planting	2003	Monterey
30	Perennial Planting	2004	Monterey
31	Tree Planting	2012	Monterey
32	Tree Planting	2018	Monterey
33	Perennial Planting	2018	Monterey
34	Perennial Planting	2016	Alameda
35	Tree Planting	2016	Marin
36	Perennial Planting	2017	San Mateo

Supplemental Table 3.2. Detailed description of disturbance events occurring within paired sites, including the dates of disturbance and year practice was planted.

Site	Conservation Practice	Planting year	County	Disturbance Type	Disturbed site	Years confirmed
1	Perennial Planting	2009	Monterey	Differences in vegetation, cropping management in unrestored site	Unrestored	1986-2008
2	Perennial Planting	2004	Yolo	Prescribed burns within restored site	Restored	2004, 2005, 2008, 2010
5	Perennial Planting	2012	Sonoma	Herbicide application or vegetation removal from restored site	Restored	2011

9	Tree Planting	2016	Tehama	Significant tillage and grading event within unrestored site	Unrestored	2010-2012
10	Riparian Restoration	2014	Tehama	Significant tillage and grading event within both sites	Both Sites	2010-2012
11	Perennial Planting	2014	Tehama	Significant tillage and grading event within restored site while unrestored site encountered no disturbance event	Restored	2010-2012
12	Riparian Restoration	2018	Sonoma	Restored site irrigated and mowed	Restored	1993-2000
17	Tree Planting	2014	Tehama	Differences in irrigation or vegetation cover in 1999. In 2002 tillage occurred within the restored site.	Restored	1999, 2002
19	Tree Planting	2015	Solano	Clear differences in grazing management within the restored site, while the unrestored site remained ungrazed	Restored	2011, 2014
26	Perennial Planting	2017	Yuba	Mastication event within the restored site.	Restored	2016
30	Perennial Planting	2004	Monterey	Possible prescribed burning in the restored	Restored	2002

31	Tree Planting	2012	Monterey	Mastication within both sites, but a more intense event within the restored site.	Both sites	2012
34	Perennial Planting	2016	Alameda	Previous windmill farm with control in parking lot. Parking lot expanded in 2015 until practice implementation	Both sites	1993-2015 2015-2016
35	Tree Planting	2016	Marin	Restored site under varying management of plowing, possible hay planting and mowing	Restored	2002-2016