Evaluating Grazing for Conservation and Fuel Management

Results from a 2-year Study at Rancho Jamul Ecological Reserve and Hollenbeck Canyon Wildlife Area

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UC Berkeley Project Team: James Bartolome[1*], Lynn Huntsinger[1*], Matthew Shapero[2*], Michael D. White[3], Lawrence Ford[4*], Felix Ratcliff[4*], Kaveh Motamed[4], Joyce Qiao[4], Chris Nygard

[1] University of California, Berkeley
[2] University of California, Agriculture and Natural Resources
[3] Michael White Consulting
[4] Rangeland Conservation Science
* Certified Rangeland Manager

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Executive Summary

This Executive Summary provides a brief summation of the overall project questions, approach, findings, and next steps. This project evaluated the potential for grazing to enhance conservation values in grasslands and coastal sage scrub habitats while simultaneously reducing fire hazard from non-native annual grasses.

Questions:

The project was designed to answer four primary questions (and their associated objectives):

1) How effective is grazing at reducing fire risk?
    a. Reducing fuels from non-native herbaceous plants.
    b. Reducing native and non-native fuel loads in fuel breaks.
2) Can grazing effectively enhance disturbed native grassland and forb (non-graminoid herbaceous flowering plant) habitats?
    a. Reducing non-native grasses and forbs in disturbed grasslands and forblands.
    b. Increasing native forb, grass, and shrub cover and bare ground.
3) Can grazing enhance disturbed native coastal sage scrub habitat?
4) Can grazing reduce nonnative grass and forb cover in disturbed coastal sage scrub to increase native shrub cover and bare ground and improve habitat for MSP species such as Quino checkerspot butterfly (*Euphydryas editha quino)*, California gnatcatcher (*Polioptila californica*), and black-tailed jackrabbit (*Lepus californicus*)?

Approach:

There are important limitations to using the existing literature to answer the primary questions. A review of the literature reveals that potential trade-offs among management goals and the limitations and the possibilities of grazing are not well described or understood (see the literature review conducted as part of this project, https://sdmmp.com/view_article.php?cid=SDMMP_CID_187_625051ca1fb35). While there are good examples of meeting similar goals with livestock grazing, the literature includes findings that: a) grazing may cause harm to shrublands or other sensitive resources; b) there are limitations to what grazing can achieve and potential trade-offs among management goals, but those limitations are neither well described nor understood; and c) there is a need for improved adaptive management and monitoring to facilitate managing livestock to optimize achievement of management goals.

Rancho Jamul Ecological Reserve (RJER) and Hollenbeck Canyon Wildlife Area (HCWA) have good information about geology, vegetation structure, species of special concern, and current grazing use in the literature, but soils and vegetation change information is mostly lacking. The conceptual models for Ecological Site Descriptions (ESDs) and State and Transition Models (STMs) require soil information, in particular as linked to how and why vegetation changes, that is not available for the study site. These models have been widely applied in some rangelands but have not been well-developed in San Diego County and the rest of Southern California. Reliable vegetation reference conditions (potential natural vegetation states) are also not available. This information is essential for answering the study questions and evaluating the efficacy of grazing practices to meet study objectives.

To address these needs, we identified Ecological Sites (ES) based on landform and soil characteristics, and ES-specific management-scale STMs to better describe spatial and temporal processes and to place the role of management within the context of other drivers (pgs. 14-19). We used a ground-up, plot-based process to do this. Our classifications are built from empirical plot-based quantitative data. We provide descriptions of vegetation states for each ecological site (ES), and a catalog of the changes (and drivers of those changes) between states. We document details of livestock use and stocking and pasture rotations. To determine grazing effects on vegetation we also measured Residual Dry Matter (RDM) on our plots and mapped levels of RDM on grazed and ungrazed pastures. We used available remotely sensed historical information to identify areas that were historically cultivated on the properties. Cultivation may have long-lasting impacts to soil structure and may extirpate perennial species. It is therefore critical to understanding current vegetation conditions.

The ESD and STM approach applied to this study was essential for working out vegetation spatial and temporal relationships among species, community composition, structure, and grazing. More years of monitoring and expansion to new sites will allow us to better work out these interdependent factors, but already a picture is emerging of the ability of grazing to meet management objectives on the reserves.

Findings:

1) How effective is grazing at reducing fire risk?

Effective, with limitations. Cattle grazing can significantly reduce fuels from herbaceous species, but the end result depends on the management decisions and constraints of the reserve manager and rancher, and the spatial and temporal factors affecting annual forage (and fuel) production.

Dried herbaceous plants are characteristic of California grasslands in the dry season and are fine fuels that ignite easily. Results from the first two years of study show that grazing can effectively reduce herbaceous fuel loads from annual grass biomass in grasslands and coastal sage scrub at RJER/HCWA. At the same time, detrimental impacts to important coastal sage scrub species like California sagebrush and California buckwheat were not seen – in fact California sagebrush cover had a positive relationship to grazing in our study. The efficacy of grazing to reduce herbaceous fuel loads is determined by many factors, principal among them is how much production occurs in a given year (which is weather and site driven), and when, where, and how much livestock grazing occurs. The cow-calf grazing operator can only increase the herd incrementally from year to year, so the occasional high forage production years complicate the rancher’s ability to meet biomass reduction needs across the two reserves. The reserves have multiple management goals relating to biomass reduction through grazing, and the reserve managers and ranching lessee do a good job of prioritizing target areas in high-production years.

2) Can grazing effectively enhance disturbed native grassland and forbland habitats?

Yes. Some grassland conservation objectives are only achievable on specific ecological sites.

Each of the four ESs has grassland vegetation states. In some of these sites, grazing appears to affect species composition, but it manifests differently among the sites. In the Alluvial ES, ungrazed plots were far more likely to fall into the "Ripgut Grasslands" (*Bromus diandrus* dominant) state with high annual grass cover and low forb cover. Grazed plots in this ES had much higher cover of forbs and generally had higher species richness – including more native forb species. The forb component of grazed grasslands in this ES was still overwhelmingly comprised of non-native annual species, but regardless of origin, forbs may provide more lower-statured vegetation, flower pollen, and other resources thought to benefit many wildlife species.

Non-native annual grasses had a statistically significant negative relationship with livestock grazing, but the species responses varied among ESs. Ripgut brome was very common on ungrazed Alluvial and Granitic Hills ESs, but less common in ungrazed Volcanic Hills and Volcanic Alluvium ESs. On the two volcanic ESs, wild oats were much more common and were also reduced by livestock grazing. Purple false brome is also common in the Volcanic Hills ES, however its cover was not significantly related to livestock grazing.

Native perennial grass presence was not strongly related to livestock grazing but was strongly tied to ESs. Needlegrasses (*Stipa sp.*) were far more common in the Volcanic Hills ES and Volcanic Alluvium ES than the others, a finding supported by the phytolith analyses. Given the distribution of perennial grasses and the phytolith record, enhancing native perennial grasses is likely to be a feasible management goal only on some ESs.

3) Can grazing enhance disturbed native coastal sage scrub habitat?

Yes, but this question is complex and not yet fully answered by this study. Exotic grass cover and residual biomass (mostly composed of exotic species) were reduced in the ESs that supported CSS, while many CSS shrub species were not affected or had positive relationships with grazing. Further work should experimentally investigate the mechanisms by which grazing affects shrub demography.

There are multiple ways livestock grazing may affect shrublands. These vary in scale and in the mechanisms of impact. The following are observations and results from this study that relate to livestock impacts to coastal sage scrub habitat:

Direct impacts to shrubs. There was little direct evidence of cattle browsing on most coastal sage scrub shrub species. California sagebrush (*Artemisia californica*) and California buckwheat (*Eriogonum fasciculatum*) had very little evidence of browsing. Cover of California sagebrush was positively associated with grazing as well, indicating that it is not being directly harmed by livestock. Some species, like laurel sumac, do appear to be negatively impacted by livestock. We did not observe browsing on this species, but cattle appeared to break low branches to create shade opportunities under these tall-statured shrubs. Shrubs in high impact areas such as next to a water trough may show indiscriminate browsing.

Reduction of wildfire impacts to coastal sage scrub. While we did see significant reduction of fuels in both grasslands and shrublands, it is difficult to predict how that translates to fire frequency and severity. Wildfire behavior is highly dependent on weather and topography, and the location of fires is also dependent on ignition. Wildfires may be more easily controlled on the reserves due to grazing, and the probability of a fire that is difficult to control may be reduced, but the ultimate impacts of wildfire to coastal sage scrub ecosystems depend on many interacting variables that are difficult or impossible to predict in advance.

Increased germination and seedling survival. Another plausible mechanism for enhancing shrublands is that livestock grazing could eliminate thatch build-up and reduce competition from annual grasses for water, light, and nutrients, thereby enhancing shrub germination and seedling survival. We observed many shrub seedlings on study plots, especially of California sagebrush, but will need additional sampling to sort out the interaction of grazing, ES, and year effects. The Volcanic Hills ES generally produced much more herbaceous biomass than the Granitic Hills ES, so if competition from annual grasses is a major factor in shrub reproduction and survival, grazing may have a bigger impact there.

4) ***Can grazing reduce non-native grass and forb cover in disturbed coastal sage scrub to increase native shrub cover and bare ground and improve habitat for MSP species such as Quino checkerspot butterfly, California gnatcatcher, and black-tailed jackrabbit?***

Yes, grazing can benefit habitat quality for some MSP species. Grazing appears to have a positive effect on habitat quality for burrowing owl (Athene cunicularia) and Otay tarplant (Deinandra conjugens), and potentially on California gnatcatcher and other CSS-associated species. Different species have distinct habitat requirements and patterns of occurrence, and the effects of grazing must be considered separately for individual species. The ESD/STM framework provides a useful basis for evaluating these effects.

Several MSP species are potentially affected by livestock grazing. Many of these species, including Quino checkerspot butterfly, California gnatcatcher, and burrowing owl are considered to be threatened by ecosystem changes due to invasion by non-native annual grasses. Grazing did effectively reduce annual grass height, cover, and biomass across ESs. In the case of burrowing owl, ungrazed study plots in the Alluvial ES generally had less favorable habitat than grazed plots. We did not directly study the owl populations or demographic processes such as predation, prey availability, and burrow availability. Those factors may take precedence over herbaceous vegetation structure.

Similar to pastures with burrowing owls, grazed study plots in the ES containing Otay tarplant had lower residual biomass and vegetation height than ungrazed plots in the Volcanic Alluvial ES. Based on prior research at Rancho Jamul Ecological Reserve, these conditions are likely beneficial for Otay tarplant, however the relationship needs to be more rigorously tested. We collected Otay tarplant frequency data prior to grazing treatments planned in the 2023-24 grazing season including experimental exclosures to test the impact of grazing on this species.

Relationships between grazing and other MSP species deserve a closer look as well. There are important questions remaining about the impact of grazing to habitat elements like cryptogamic soil crusts (which are considered important elements of Quino checkerspot butterfly habitat), ground squirrel occurrence (a critical factor for burrowing owl occupancy), invertebrate species and abundance (important prey for several birds including Tricolored blackbird (*Agelaius tricolor*), and habitat elements relating to other key species like California gnatcatcher.

Next Steps:

The infrastructure, information, and analytic protocols developed through this project provide a foundation for developing improved information for fire managers on the role of grazing in shaping vegetation structure. Opal phytolith analyses suggest that further work using these materials and archival information to further explore the impacts of historic tilling on species composition and phytolith preservation will shed light on vegetation drivers and ES potential. We found that additional work investigating relationships between grazing, ES, and recruitment and cover of California sagebrush and California buckwheat is warranted. In the next phase of the study, we will selectively begin modifying conservation grazing practices on the documented existing ESs in an attempt to link patterns of grazing use to distribution of fuels, to ESs and STMs, and to MSP objectives.

Grazing appears to have a positive effect on habitat quality for burrowing owl and Otay tarplant, and potentially on California gnatcatcher and other CSS-associated species. Additional work, including applying grazing modifications, is merited to better understand the mechanisms associated with increases in CSS shrub species and the potential negative effects on cryptogamic soil crusts and habitat of species such as tricolored blackbird and Quino checkerspot butterfly. New study areas could also potentially expand our work into additional MSP habitats supporting species such as San Diego fairy shrimp and Stephens' kangaroo rat, and expand our catalogue of ESs.

Introduction

This report presents results to date of the Grazing Monitoring Plan and Pilot Project (GMPPP) initiated in 2021 as part of the implementation of the Management and Monitoring Strategic Plan (MSP Roadmap; SDMMP and TNC 2017) of the San Diego Association of Governments (SANDAG) and the San Diego Management and Monitoring Program (SDMMP). The Roadmap provides a biologically based foundation that supports decision making and sets funding priorities for managing and monitoring species and vegetation communities on conserved lands in western San Diego County. In early 2019, out of concern about frequent wildfires and the impact of non-native grasses on habitat conditions for wildlife and native plants, the SDMMP convened an informal Grazing Monitoring Plan Working Group (Working Group) comprised of federal, state, and local resource agency representatives, local land managers, biologists, SANDAG staff, and SDMMP, and in 2020 the Working Group recommended funding a pilot project that would design and implement a study to determine the efficacy of livestock grazing as a management strategy to enhance MSP Roadmap (MSP) species and their habitats and reduce the risk of fire on conserved lands in western San Diego County. Livestock grazing has been suggested as a means of controlling invasive plants, reducing fuels, and enhancing habitat for native forbs and grasses by managing non-native grasses.

The GMPPP is conducted under an MOU between SANDAG and The Regents of the University of California on behalf of the University of California, Berkeley (UCB), effective March 24, 2021 and amended effective February 10, 2022 and January 27, 2023. A team supervised by the Principal Investigators, UCB Professors James Bartolome and Lynn Huntsinger, is responsible for completing tasks under the MOU. The project was initiated in 2021 on two adjoining properties managed by California Department of Fish and Wildlife: Rancho Jamul Ecological Reserve (RJER) and Hollenbeck Canyon Wildlife Area (HCWA).

Grazing Monitoring Plan and Pilot Project Problem Statement

Coastal Sage Scrub and its associated grasslands form a critical habitat matrix for many MSP species and other sensitive species in San Diego County. These habitats, and particularly coastal sage shrublands, are severely threatened by the increased frequency and intensity of wildfires that is occurring (Syphard et al. 2022). In October/November of 2003 the Otay Fire burned approximately 80% of the Rancho Jamul Ecological Reserve; in 2007 the Harris Fire burned portions of Rancho Jamul. Herbaceous cover in the study area is often dominated by non-native annual grasses, which reduce habitat quality for many native species and may increase the frequency of wildfire occurrence. There are good reasons to believe that livestock grazing can effectively reduce fuel loads from non-native annual grasses, while potentially improving habitat for some MSP species on conserved lands in San Diego County (Popay and Field, 1996; Weiss 1999; Firn et al 2018; Barry and Huntsinger 2021; Ratcliff et al. 2022). There are also important areas of uncertainty regarding how livestock grazing may benefit or harm elements of these systems. In particular, grazing-weather-site interactions complicate our understanding of the opportunities and benefits provided by grazing. Generalizations from the scientific literature are often not adequate for conservation management planning for specific sites, and the changing climate is rapidly altering system dynamics. Assumptions about the effectiveness or limitations of grazing need to be rigorously and objectively tested.

The Working Group developed four primary questions to guide this research:

1) How effective is grazing at reducing fire risk?
   a. Reducing fuels from non-native herbaceous plants.
   b. Reducing native and non-native fuel loads in fuel breaks.
2) Can grazing effectively enhance disturbed native grassland and forb (herbaceous flowering plants that are not a grasses, sedges, or rushes) habitats?
   a. Reducing non-native grasses and forbs in disturbed grasslands and forblands.
   b. Increasing native forb, grass, and shrub cover and bare ground.
3) Can grazing enhance disturbed native coastal sage scrub habitat?
4) Can grazing reduce nonnative grass and forb cover in disturbed coastal sage scrub to increase native shrub cover and bare ground and improve habitat for MSP species such as Quino checkerspot butterfly (*Euphydryas editha quino)*, California gnatcatcher (*Polioptila californica*), and black-tailed jackrabbit (*Lepus californicus*)?

There are important limitations to using the existing literature to answer the primary questions. A review of the literature reveals that potential trade-offs among management goals and the limitations and the possibilities of grazing are not well described or understood (see the literature review conducted as part of this project, https://sdmmp.com/view_article.php?cid=SDMMP_CID_187_625051ca1fb35).

There is some evidence that grazing can help achieve the management goals in question and that a lack of grazing can be detrimental in some systems (Bartolome et al. 2014; Spiegal et al. 2016a). However, it is possible that grazing may be detrimental to coastal sage shrublands or other sensitive resources and this has yet not been systematically examined. In fact, much of the literature is generalized across soils and topography, among other site characteristics, and is not specific to characteristics and conditions that may lead to different management outcomes. It is well known that soils, in particular, play a large role in defining vegetation characteristics in California and that they are unusually heterogenous in this state. There is a need for an approach that increases our predictive ability and facilitates adaptive management and monitoring.

Ecological Site Descriptions (ESDs) differentiate areas of differing environmental characteristics and capacities and describe them as *Ecological Sites* (ES). ESDs include a description of site characteristics and potential plant communities. They also include *State and Transition Models* (STMs), critical, management-oriented diagrams that that delineate site responses to environmental factors and interventions, providing a framework for understanding how the outcomes of management interventions vary across conserved lands. Part of an STM is the Vegetation Reference plant community or vegetation state, providing a benchmark for vegetation conditions. This framework is standardized by the federal Natural Resources Conservation Service, facilitating its broad use (https://edit.jornada.nmsu.edu/catalogs/esd). Yet ESDs, STMs, and reliable benchmarks for vegetation conditions are not available for RJER or HCWA. ESDs provide a basis for testing grazing practices and patterns and documenting their influence on vegetation and site characteristics, as well as for improving managers' ability to anticipate management outcomes, including for habitat.

Our general approach and conceptual framing for answering the study questions is based on building ground-up classifications of ESs linked to state and transition models as follows (detailed in the methods section):

1. Build classifications from empirical plot-based data
2. Define and identify the ESs in the study area and develop ESDs and STMs
    a. ESDs are models that describe how spatial variation in environmental factors such as climate, soils, and topography constrain variation in vegetation dynamics and ecological function (https://edit.jornada.nmsu.edu/catalogs/esd).
    b. This approach builds on an existing body of work (https://edit.jornada.nmsu.edu/catalogs/esd; Spiegal et al. 2016a; Aoyama et al. 2020).
3. Develop STMs for vegetation classes occurring within each ES
    a. These models have been widely applied to rangeland planning and management but have not been developed for San Diego County (Spiegal et al. 2016b).
    b. STMs provide a description of vegetation states for each ES, and catalog changes (and drivers of those changes) between states.
    c. STMs are needed to evaluate and guide management.
4. Document numbers and movements of cattle across the landscape to examine how grazing patterns and practices contribute to meeting management goals.
5. Evaluate habitat relationships and grazing effects for the study area in the context of ESDs and existing cattle grazing practices.

Study Area and Methodological Approach

Rancho Jamul Ecological Reserve and Hollenbeck Canyon Wildlife Area

The study sites, RJER and HCWA, total 9,891 acres in southern San Diego County (**Figure 1**) They were acquired by the California Department of Fish and Wildlife as part of the San Diego Multiple Species Conservation Program South County Plan. RJER and HCWA are located in the southeastern portion of the South County Plan, supporting some of the most inland populations of MSP species and habitats (https://www.sandiegocounty.gov/content/sdc/pds/mscp/sc.html Table 2-1).

Situated on the flanks of the San Ysidro and Jamul mountains, RJER and HCWA and are traversed by Jamul and Dulzura creeks, which drain to Lower Otay Reservoir to the southeast. The terrain of the properties is varied, with flat to gently sloping terraces, rolling hills and knolls, and steep slopes. Elevations vary from around 800 feet to 2,800 feet. They are underlain by three general rock types of different geologic origins: 1) metavolcanic rock associated with the San Ysidro and Jamul mountains, 2) granites, primarily quartz diorite and granodiorite, and 3) gabbroic rocks.

Land use

After at least 12,000 years of native management (https://jamulindianvillage.com/history), colonization led to the study area lands passing through a number of national governments and several owners. RJER and HCWA fall within the ancestral territory of the Kumeyaay Nation. Colonization by the Spanish and the establishment of Missions starting in 1769 displaced many native people from their homelands and forced many to work as laborers. Under the Mexican government, established in 1821, many Ranchos, large livestock rearing properties, were established, including the Rancho Jamul land grant to Pio Pico in 1829 (Smith et al. 2014). During this period, Mission lands were secularized. San Diego came under American governance in 1848, and once Pio Pico's grant was rejected by American courts in 1851, the land went through a succession of private ownerships until the California Department of Fish and Wildlife acquired RJER in 1998 and HCWA in 2001. Jamul Village, home of the Jamul Village Tribe, was established originally from land returned by the Catholic Church in 1912 (https://jamulindianvillage.com/timeline). Over time a few more acres were returned and Jamul Village, located near the study site, was designated a reservation by the U.S. Government 1981. Between the time of Spanish colonization and the late-1990s, beef-cattle grazing and farming were the predominant land uses in the area. Cattle were removed from the study area properties shortly before California Department of Fish and Wildlife acquired the land (TAIC 2006).

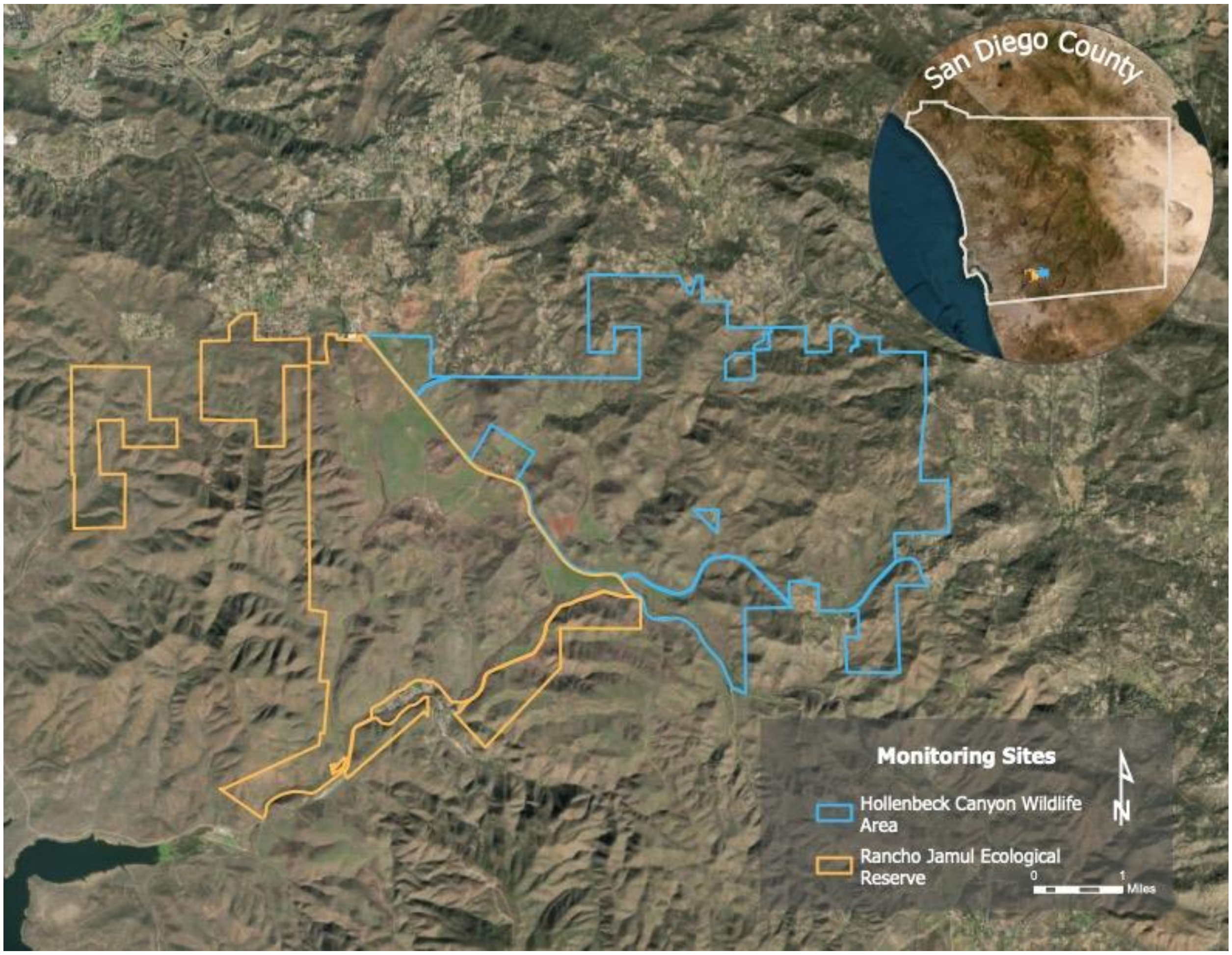


Figure 1. Rancho Jamul Ecological Reserve and Hollenbeck Canyon Wildlife Area

Cattle grazing

In recent years cattle grazing has been slowly re-introduced to RJER (2014) and HCWA (2019) to address specific natural resource and wildlife concerns, particularly excessive grass thatch and fuel build up associated with non-native Mediterranean annual grasses. The current *Excess Vegetation Removal Permit*, or grazing lease, between the California Department of Fish and Wildlife and the 4J Horse and Livestock Company runs from 2019-2024. The permit states that "the primary objective of livestock grazing is the enhancement and maintenance of vegetative conditions conducive to supporting the property's high diversity and unique range of bird species," with a secondary objective of fire hazard reduction.

Accordingly, cattle grazing is now allowed on 2,853 acres of RJER and 430 acres of HCWA. Cattle distribution is managed through a combination of *hard-line*, using permanent wire fences, unbarbed on top and lower wires, and *soft-line*, using temporary electric fences, to delineate pastures (**Figure 2**). At RJER, there are twenty-one pastures of varying sizes (range: 18-388 acres), and at HCWA, there are three pastures of roughly equal size (122 acres, 144 acres, and 163 acres (**Table 1**).

The cattle lessee, 4J Horse and Livestock, runs a cow-calf beef operation, meaning that they keep a base herd of breeding cows on the properties year-round, but have seasonal increases in livestock numbers due to the birth of calves from approximately September-April. Calving occurs in late summer to fall, and calves not retained to build the herd are shipped from the property the following spring. Animal feeding relies almost exclusively on the vegetation that grows on the properties; supplementation with hay and other sources of protein is minimal and seasonal, and the lessee markets many of his animals as grass-fed.

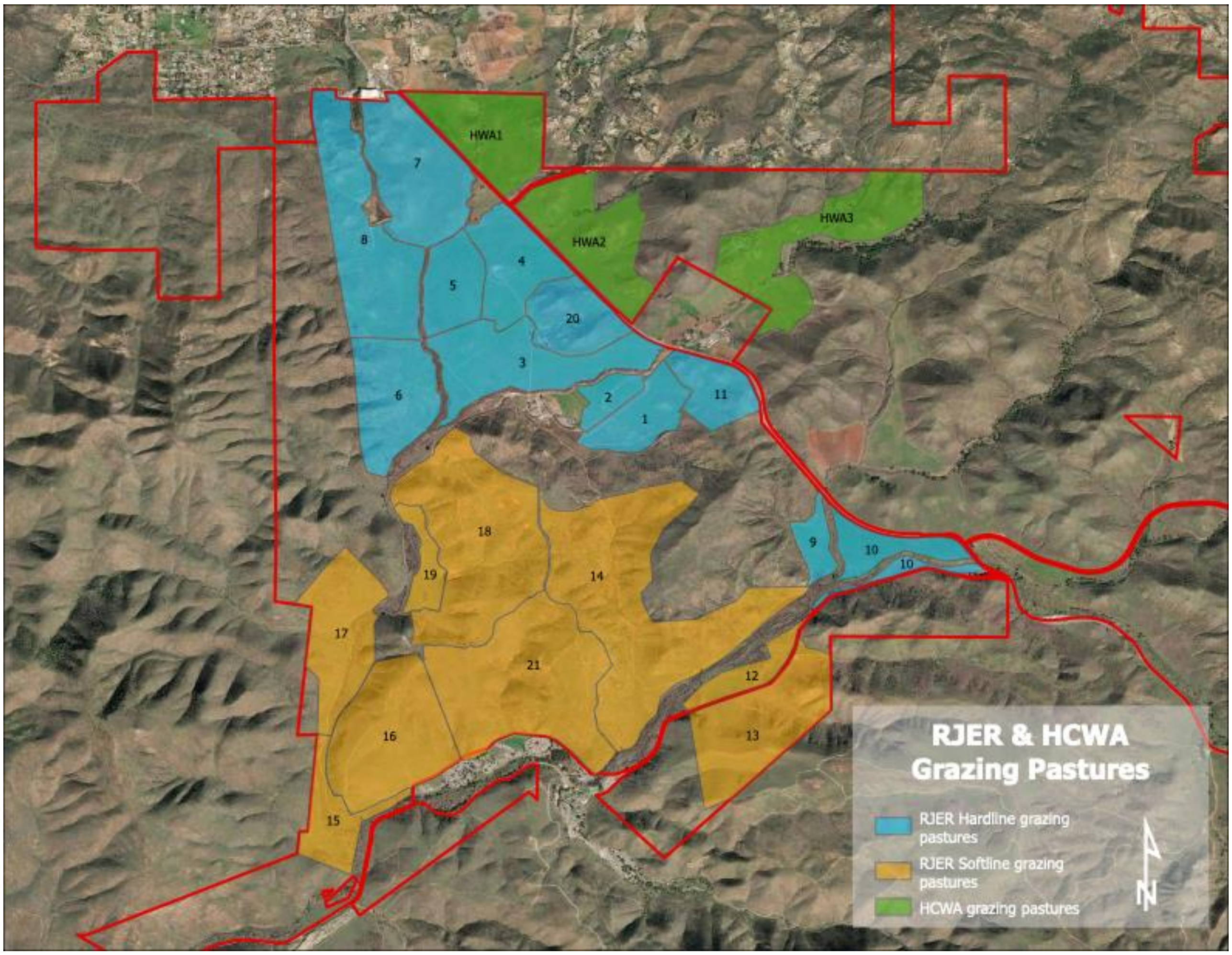


Figure 2. Grazing pastures at Rancho Jamul Ecological Reserve and Hollenbeck Canyon Wildlife Area

Generally, the cattle are managed with regular pasture rotations throughout the year. Rotations are dictated by forage conditions, weather, wildlife and vegetation management objectives, and livestock production demands (e.g. calving, branding, etc.). In recent years, the general pattern of grazing has followed this sequence 1) the RJER hard-line pastures in late winter-spring, 2) HCWA pastures in summer, and 3) RJER soft-line pastures in late summer-fall.

Table 1. *Hard-line* (barbed-wire) and *soft-line* (movable electric fence) pasture sizes at Rancho Jamul Ecological Reserve and Hollenbeck Canyon Wildlife Area.

Pasture	Property	Fence Type	Acres
1	RJER	barbed-wire	70
2	RJER	barbed-wire	30
3	RJER	barbed-wire	157
4	RJER	barbed-wire	98
5	RJER	barbed-wire	73
6	RJER	barbed-wire	128
7	RJER	barbed-wire	162
8	RJER	barbed-wire	201
9	RJER	barbed-wire	30
10	RJER	barbed-wire	54
10	RJER	barbed-wire	18
11	RJER	barbed-wire	63
12	RJER	electric	41
13	RJER	electric	229
14	RJER	electric	388
15	RJER	electric	69
16	RJER	electric	195
17	RJER	electric	147
18	RJER	electric	290
19	RJER	electric	37
20	RJER	barbed-wire	80
21	RJER	electric	293
HWA1	HCWA	barbed-wire	123
HWA2	HCWA	barbed-wire	144
HWA3	HCWA	barbed-wire	163

A regional vegetation map developed in 2012 (Oberbauer et al. 2012) mapped the properties primarily as coastal sage scrub, grassland, and forbland, with small amounts of chaparral vegetation, particularly on the eastern, higher elevation portion of HCWA . Our field observations during this project document how spatially variable species composition is across these vegetation types, nevertheless, the MSCP vegetation map allowed us to target plot selection in both shrub and herbaceous vegetation types. In addition, we documented past cultivated agriculture on both properties.

Study plot establishment

Detailed sampling methods are presented in Appendix B.

We began the study in March 2021, conducting site visits to review the landscape at Rancho Jamul and Hollenbeck Canyon. We also worked with SDMMP to assemble existing GIS data for the two properties, including spatial files for topography, soil, vegetation, species locations, and ranch infrastructure. In addition, we studied existing monitoring protocols and sampling designs. With this information, we developed a randomized plot design that spanned a diverse array of environmental conditions. We discussed plot locations with SANDAG partners prior to sampling.

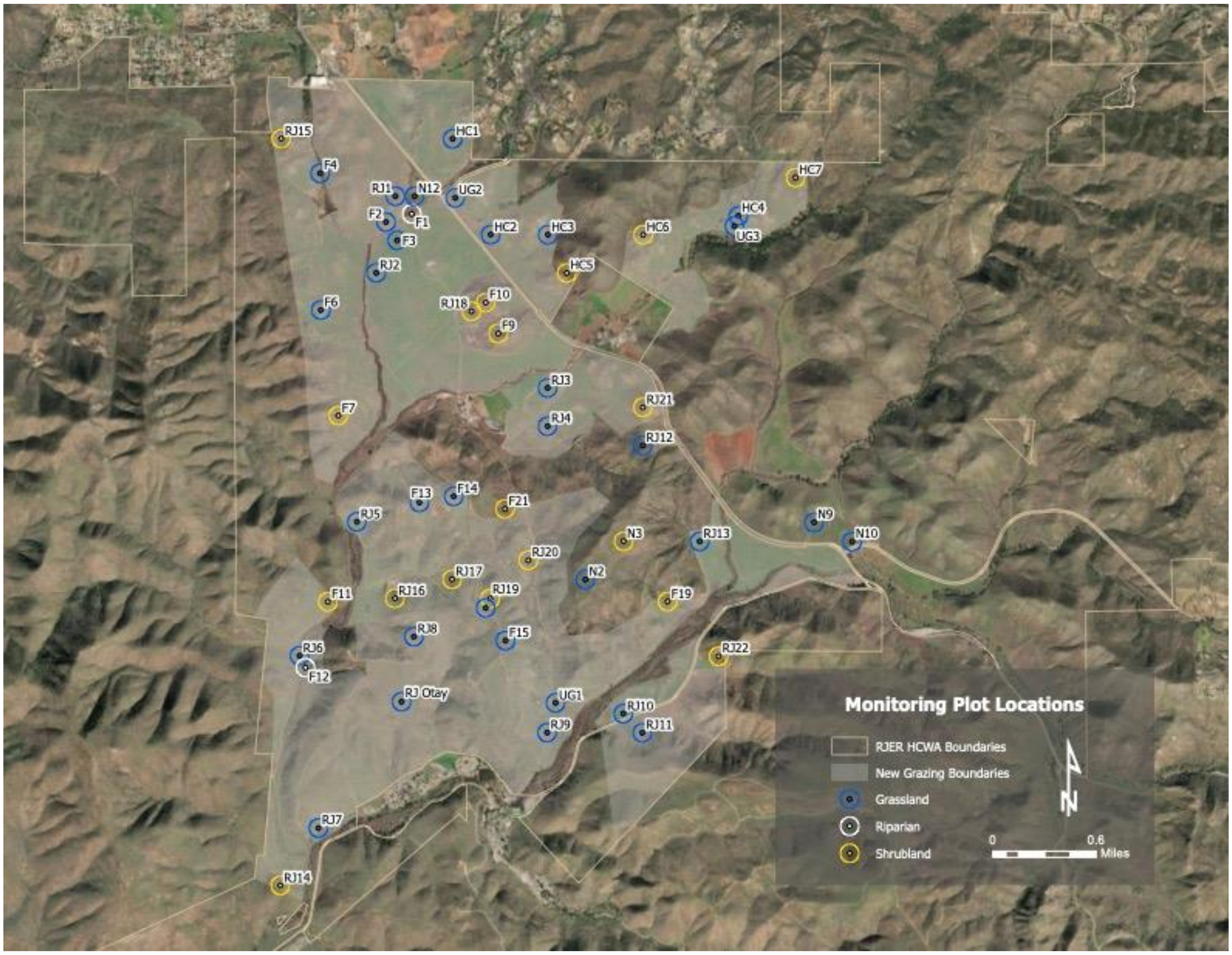


Figure 3. Study plot locations and grazing areas at Rancho Jamul Ecological Reserve (RJ) and Hollenbeck Canyon Wildlife Area (HC). "F" denotes plots from Robert Fisher's lab, "N" denotes plots added after Fall 2021, and "UG" are ungrazed plots.

We initially selected 29 plot locations across RJER and HCWA from a county-wide grid of points placed 150 meters apart. The grid was created by the SDMMP. Plots were 10 x 10 m (32.8 ft x 32.8 ft) and were sited to ensure we captured a diversity of factors that could influence vegetation composition, temporal transitions, and responses to management, spanning a variety of vegetation types, geologies, fire histories, grazing management practices, and topographies. We increased the number of plots over time as new information and opportunities emerged. In 2022, nine more plots were selected in a similar manner to complement the first 29 samples. In addition, 16 new plots, termed *Fisher* plots, were added that were originally established by Robert Fisher's lab in 2001 because they provide information about vegetation cover and change going back decades, In sum, 29 plots were sampled in the fall of 2021, 32 were sampled in the spring of 2022, and fifty-four were sampled in the fall of 2022 and spring of 2023 (**Figure 3**).

Sampling approach

At each 10 x 10 m (32.8 ft x 32.8 ft) plot, soil chemistry, texture, and phytolith content were assessed. In Spring 2022 and 2023, vegetation production and composition were sampled using a plot-wide plant inventory, line-point intercept, line-intercept, a modified relevé, herbaceous biomass, obstruction height using a Robel Pole, and substrate notation. Residual Dry Matter (RDM) sampling was conducted in the fall seasons of 2021, 2022, and 2023. RDM condition classes were specified and mapped in 2022 and 2023. All plots were photographed in spring and fall. Total annual solar radiation at each of the study plots was calculated with ArcGIS.

Using historical aerial imagery downloaded from Earth Explorer for the years: 1928, 1956, 1971, 1982, 1983, 1989, 1996, and 2000, we identified areas in the reserves that very likely had significant post-colonial soil disturbance from activities such as construction or cultivation. A final shapefile of formerly cultivated areas and construction influenced areas was created that best summarized all years into one hand-drawn polygon layer.

Ecological Site Description development

Ecological sites were based on a classification of environmental variables at each of the study plots to try to find "types" of plots that have similar environmental characteristics and are therefore likely to have similar potential vegetation states, vegetation dynamics, ecosystem processes, and responses to management. One difficult challenge is determining which environmental variables to include in the data used in the classification. To formally address this, we chose variables using Random Forest analysis. Variables were selected based on whether they were important predictors of individual species occurrence on the 54 study plots. Ecological site classification was conducted using cluster analysis. Plots were classified based on similarities among the selected environmental variables associated with each plot.

Vegetation State classification

Vegetation classification was based on a cluster analysis performed on plot data, using the vegetation data from each plot sampled in each year as the basis for clustering. The vegetation classification was performed on data from all plots together, and on data from the groups of plots in each ES.

Phytolith analysis

Microscopic particles of silica that form in plant tissues known as phytoliths provide information about species composition from several hundred to thousands of years ago. It is often possible to determine, for example, if native grasses once dominated a site (Appendix C). Phytolith samples were extracted from composite soil samples taken at each of the study plots. These samples were analyzed for counts of different phytolith morphotypes per gram of soil.

We used the reported thresholds of 0.3% phytolith dry weight in soils and a bilobate/total phytolith of 0.1% to assess the likelihood that the ESs identified in our study were prehistorically grassland. We used a t-test to determine the probability that the mean metrics from samples (plots within each ES) within each ES were greater than the published thresholds (i.e., evaluated the null hypothesis that measured metrics did not differ from published thresholds).

Grazing effects analysis

We analyzed the effects of grazing on several vegetation structure and composition attributes including percent cover of functional groups, percent cover of individual species, obstruction height, bare ground, and RDM. In these comparisons, grazing was generally treated as a categorical yes or no variable and results from grazed and ungrazed plots were compared. We used negative binomial models to examine the relationship between the continuous variable Animal Unit Days per acre (AUD/ac) of grazing and percent cover of functional groups and individual species.

Results

Grazing use

The degree of grazing use, measured as Animal Unit Days per acre (AUD/ac), varied among the 24 pastures and between the two years. An Animal Unit is the equivalent of one mature cow or one cow with a nursing calf. This is expressed graphically in a “heat map” of AUD/acre by pasture for the grazing years 2021-2 and 2022-3 (**Figure *4*** and **Figure *5***). Generally, the pastures in HCWA and in alluvial areas of RJER had higher AUD/acre use, while pastures in the southern and western hills had lower AUD/acre values. Grazing use varied between 0 and 99 AUD/acre in the 2021-2022 grazing year and between 4 and 88 AUD/acre in the 2022-2023 grazing year. The highest AUD/acre rate in both years is a bull field (where bulls are held separately from cows until breeding) near the Ranch House at RJER. Fields such as these are considered service areas, and the higher AUD/acre rates in this type of field are not unusual.

Ecological Site classification

One hundred and thirty-eight species were included in the Random Forest variable selection process. This analysis showed which environmental variables best explained the occurrence of individual species (**Figure 6)**. Soil texture, soil nutrients, elevation, and slope were shown to be important factors for the largest number of species. Other variables, including mapped variables like position on slope, landform, and history of cultivation were not shown to be as important. **Table *2*** shows the variables which were selected for inclusion in the ES cluster analysis based on the Random Forest analysis.

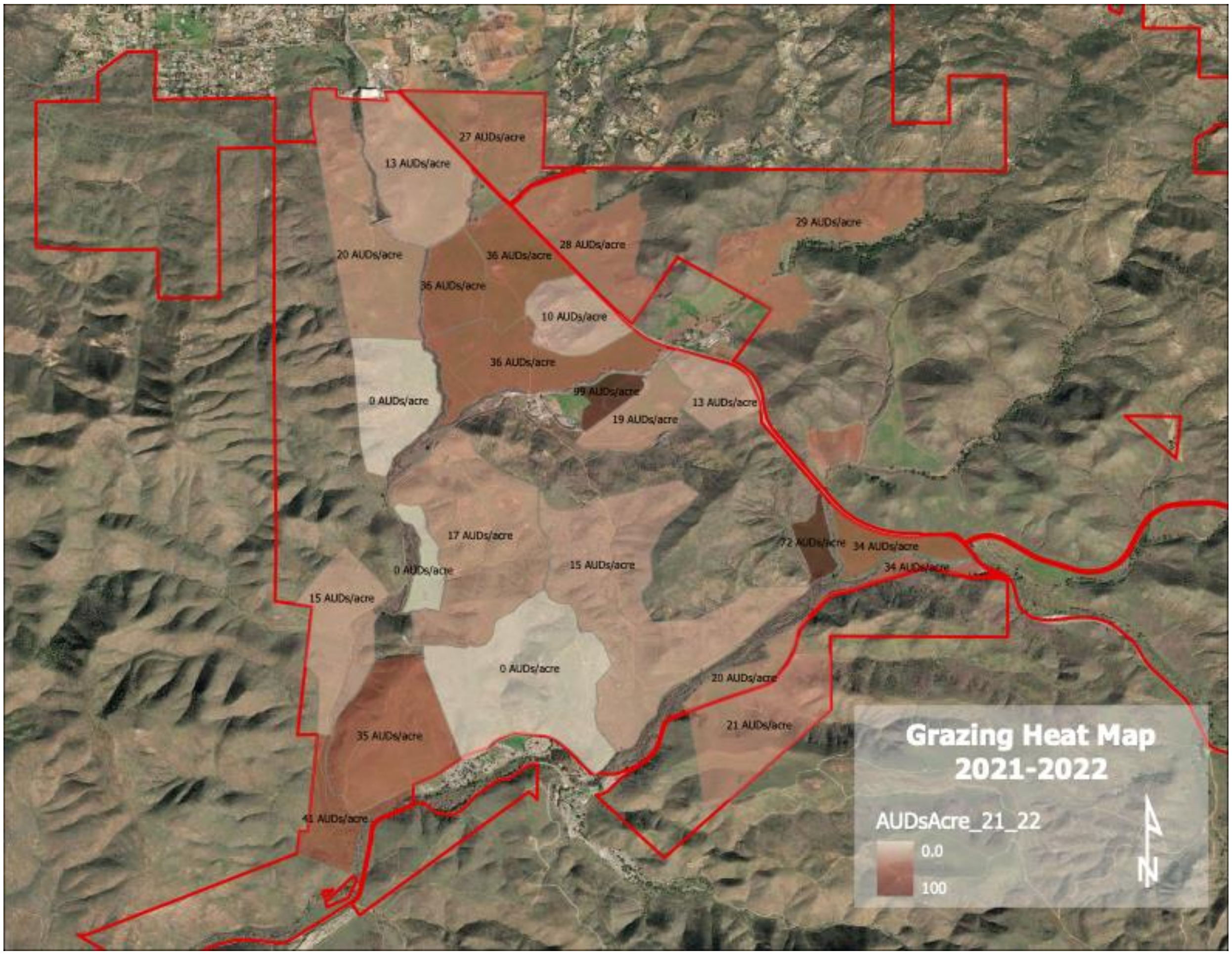


Figure 4. Heat map showing grazing use among the 24 pastures in the 2021 - 2022 grazing year. AUD=Animal Unit Days

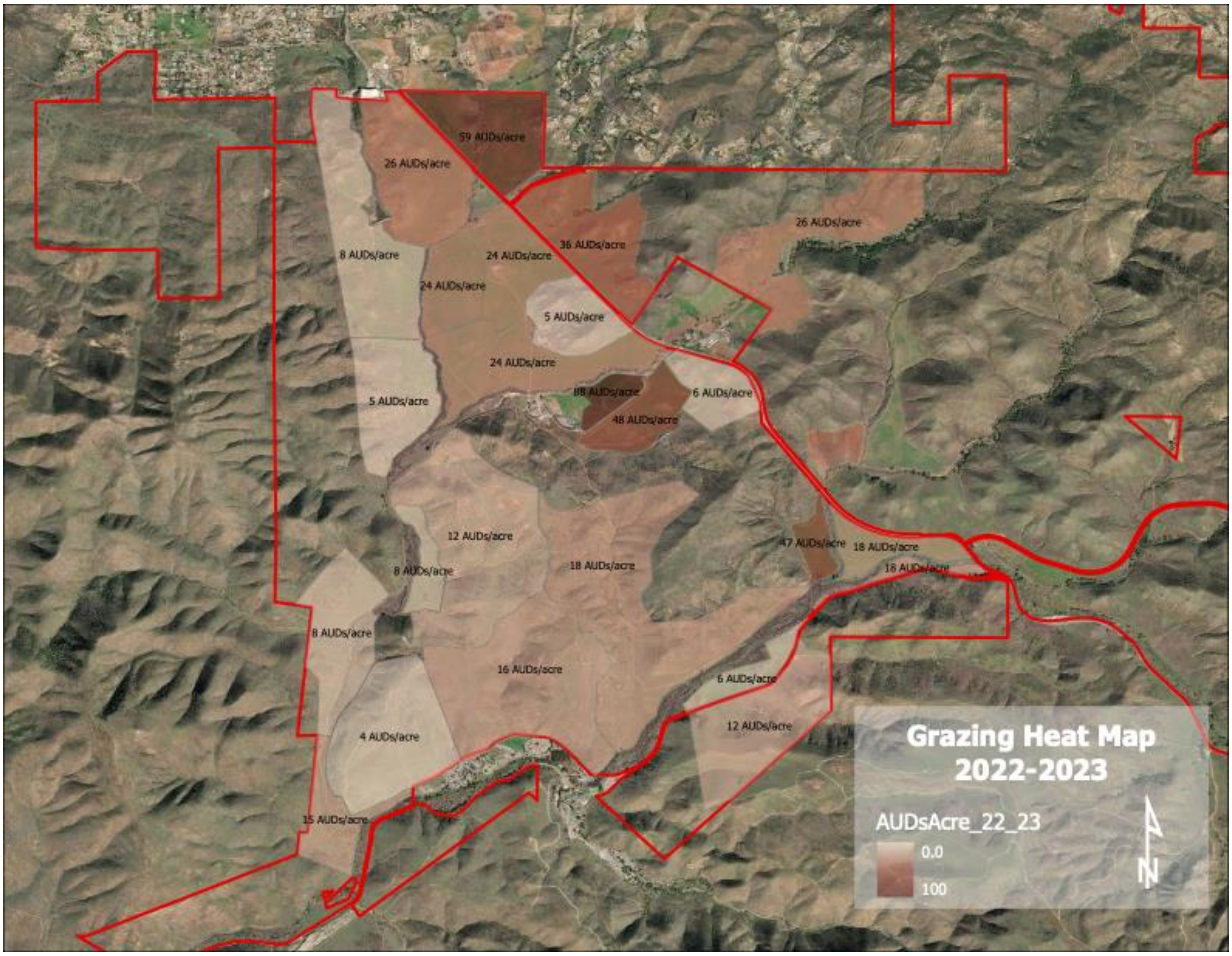


Figure 5. Heat map showing grazing use among the 24 pastures in the 2022 - 2023 grazing year. AUD = Animal Unit Days.

Table 2. Variables included in Random Forest analysis. The third column shows the average number of species that for each factor was identified as "important". Values are not whole numbers because the Random Forest model randomly subsets the data many times and the results vary per iteration.

Variable Type	Variable	Number of Species where the variable importance was in the top 50%	Included in Ecological Site Cluster Analysis
Soil Nutrients	Total Nitrogen	112.3	Yes
	Total Carbon	108.4	Yes
	Calcium	96.9	Yes
	Sulphate	94.6	Yes
	Magnesium	83.0	Yes
	Sodium	79.9	Yes
	Potassium	64.0	Yes
	Phosphorus	52.7	Yes
Soil Texture	% Sand	105.9	Yes
	% Clay	93.6	Yes
	% Silt	90.4	Yes
Topography and elevation	% Slope	58.8	Yes
	Solar Radiation	47.2	No
	Position on slope	33.7	No
	Shape	37.4	No
	Landform	49.9	No
Mapped Variables	Historic Cultivation (historic Aerial imagery)	48.7	No

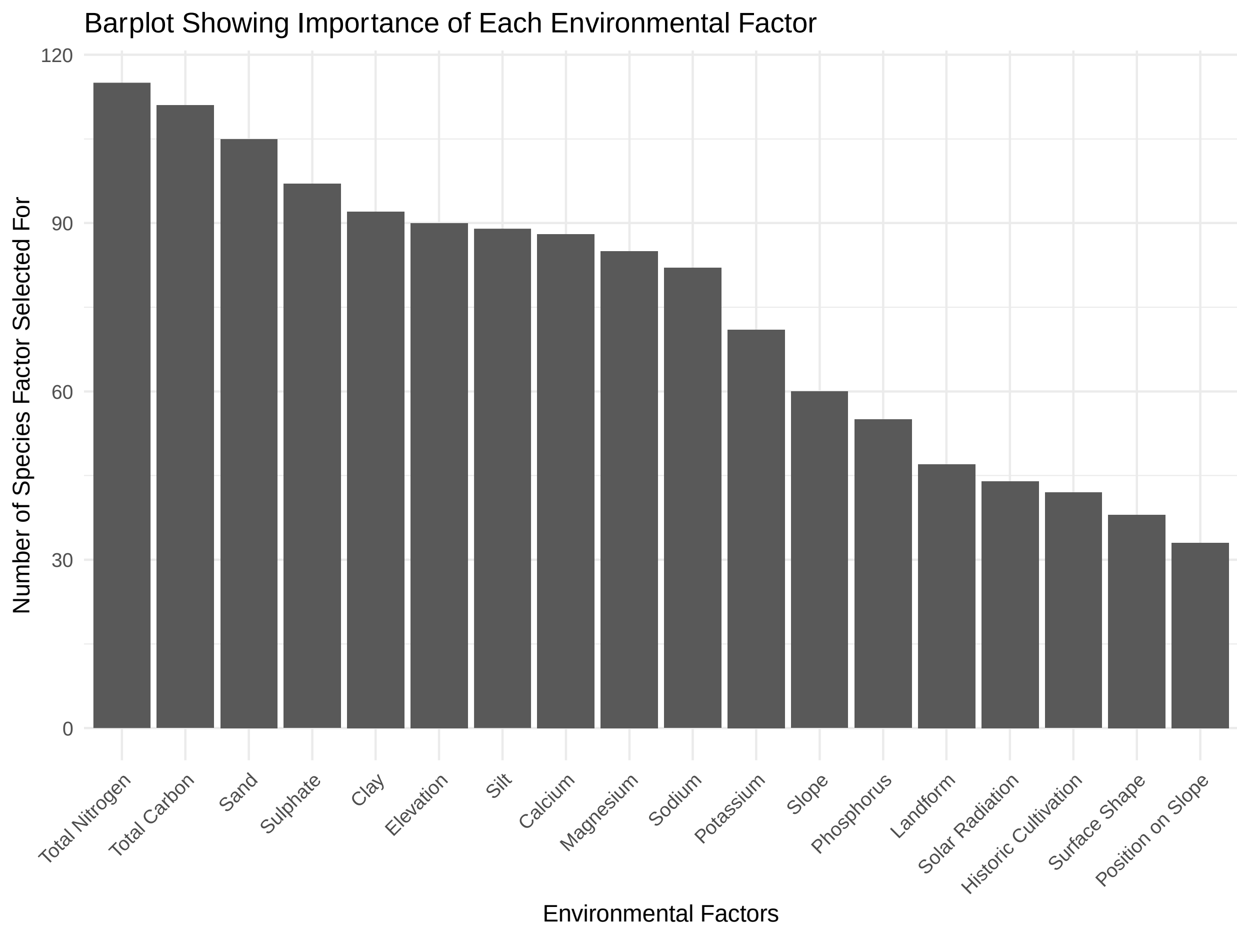


Figure 6. Random Forest analysis results showing the average number of species which included each variable in the top 50% of model importance.

Ecological Site classification results

The ES cluster analysis using the variables shown in **Table *2*** produced the cluster dendrogram showing relationships among plots in **Figure 7**. A Mantel test suggested that the ideal number of clusters was two, while indicator species analysis suggested that the optimal number of clusters was six. After reviewing the cluster dendrogram, mapping the ESs, and looking at differences in vegetation between the sites, we decided that the best number of clusters for describing management-relevant ESs was four. Each cluster is characterized by a combination of soil characteristics and geomorphology. Selecting more or fewer clusters illustrates some interesting relationships, but the differences attributed to these additional sites can be understood within the variation among the four primary ESs. Brief descriptions and physical characteristics of each ES are given in **(Table 3 Table 4)**. The distribution of ESs and their relationship to geology, soil parent material and historic cultivation are shown in **Figure 8.**

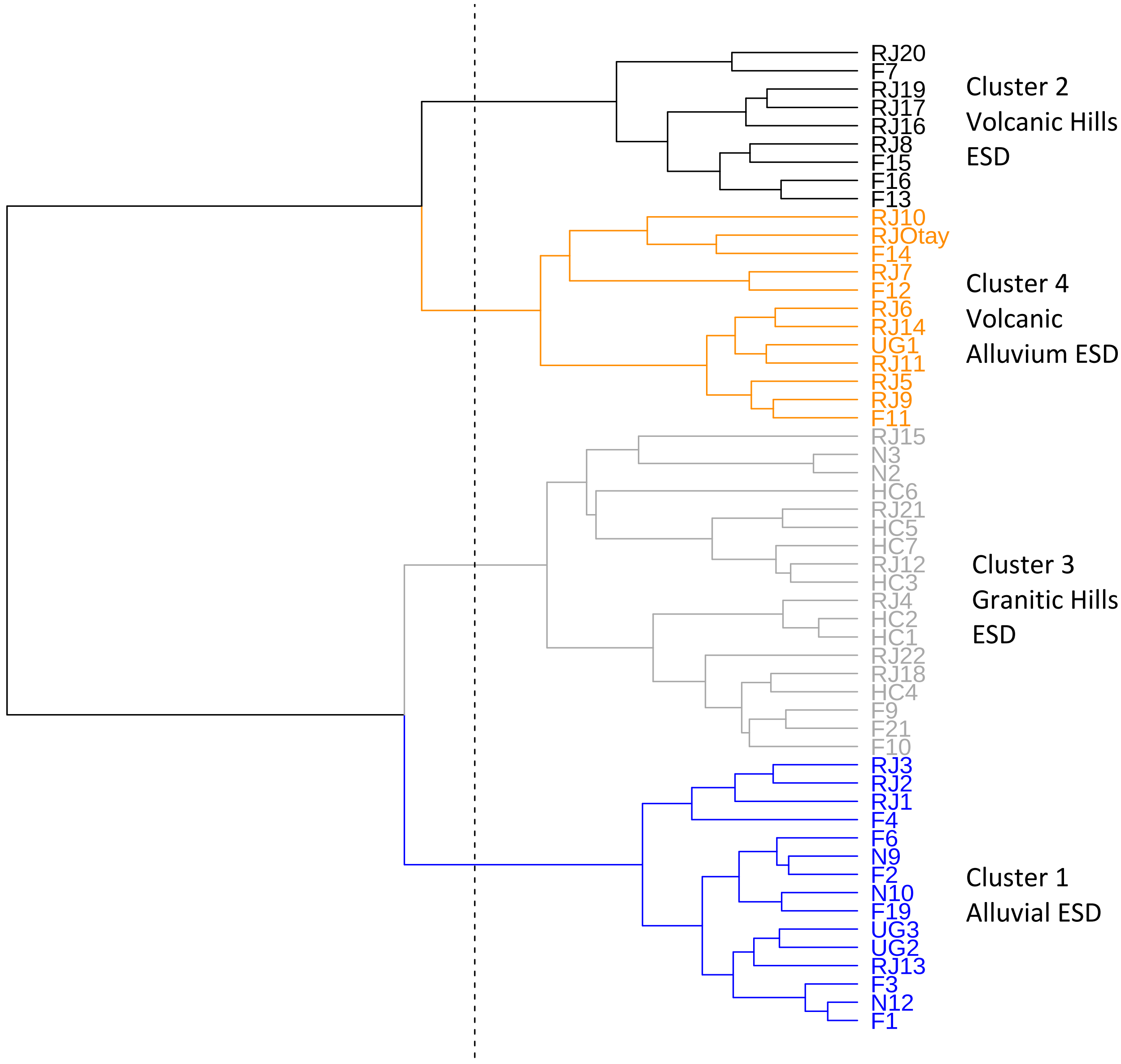


Figure 7. Dendrogram showing the ecological site cluster analysis results. The dotted line shows where the dendrogram was 'cut' to create the clusters.

Table 3. Descriptions of each ecological site.

Ecological Site Cluster Number	Ecological Site Name	Distinctive Characteristics*	General Distribution
Cluster 1	Alluvial ES	High phosphorus levels	Occurs on low-lying alluvial sites, near creeks and at toe slopes of hills with granitic soil parent material.
Cluster 2	Volcanic Hills ES	Silty soils, high % slope	Occurs on hills with metavolcanic soil parent material.
Cluster 3	Granitic Hills ES	Higher elevation, sandy soils	Occurs on hills in areas with granitic and gabbro soil parent materials.
Cluster 4	Volcanic Alluvium ES	Clayey soils; high levels of soil nitrogen, carbon, potassium, magnesium, sodium, calcium, sulphate	Occurs on foot slopes, floodplains, and terraces on alluvial and colluvial clay soils derived from metavolcanic soil parent materials. Within the Volcanic Hills ES, this ES occurs as small areas with distinct soil characteristics.

* Distinctive characteristics are characteristics that are more common (or higher magnitude) within a cluster and are less commonly found in other clusters. Characteristics given here are those that Indicator Species Analysis (Dufrene and Legendre 1997) suggests have a p<0.1 of being indicators for a given group.

Table 4. Environmental Characteristics of plots in each ecological site. Values shown are "mean (minimum, maximum)."

Variable	Alluvial ES	Volcanic Hills ES	Granitic Hills ES	Volcanic Clays ES
Elevation (m)	245.01 (221.18, 274.33)	271.24 (247.06, 303.94)	286.1 (236.45, 353.54)	215.48 (181.27, 293.51)
Slope (%)	5.82 (0.57, 15)	23.89 (7, 42)	16.11 (3.96, 35)	10.54 (1.3, 30)
Total Nitrogen (%)	0.09 (0.06, 0.14)	0.09 (0.05, 0.14)	0.06 (0.02, 0.17)	0.13 (0.09, 0.18)
Total Carbon (%)	0.98 (0.68, 1.46)	1.08 (0.48, 1.47)	0.73 (0.2, 2.01)	1.52 (1, 2.01)
Phosphorus (ppm)	18.91 (2.1, 30.1)	1.78 (1, 2.7)	6.36 (1.7, 20.1)	8.08 (1.5, 35.5)
Potassium (ppm)	213.8 (51, 425)	88 (24, 134)	87.28 (23, 192)	251.75 (142, 485)
Sodium (ppm)	23.23 (7, 49)	61.29 (31.3, 100.1)	25.58 (11, 59)	66.84 (15, 145.2)
Calcium (meq/100g)	5.48 (4.36, 7.27)	7.64 (4.8, 10.78)	4.77 (3.36, 9.51)	8.63 (5.03, 20.63)
Magnesium (meq/100g)	1.62 (0.92, 3.47)	3.63 (1.69, 5.19)	1.81 (1, 4.42)	4.05 (1.45, 12.06)
Sulphate (ppm)	5.86 (3.2, 14)	5.03 (2.7, 8.6)	3.54 (1, 5.2)	7.61 (3.9, 11.6)
Sand (%)	70.27 (61, 78)	33.67 (24, 55)	74.94 (67, 88)	41.67 (20, 67)
Silt (%)	20.53 (14, 24)	52.89 (28, 64)	17.61 (11, 23)	40.42 (25, 53)
Clay (%)	9.2 (6, 16)	13.44 (7, 19)	7.44 (1, 13)	17.92 (8, 42)

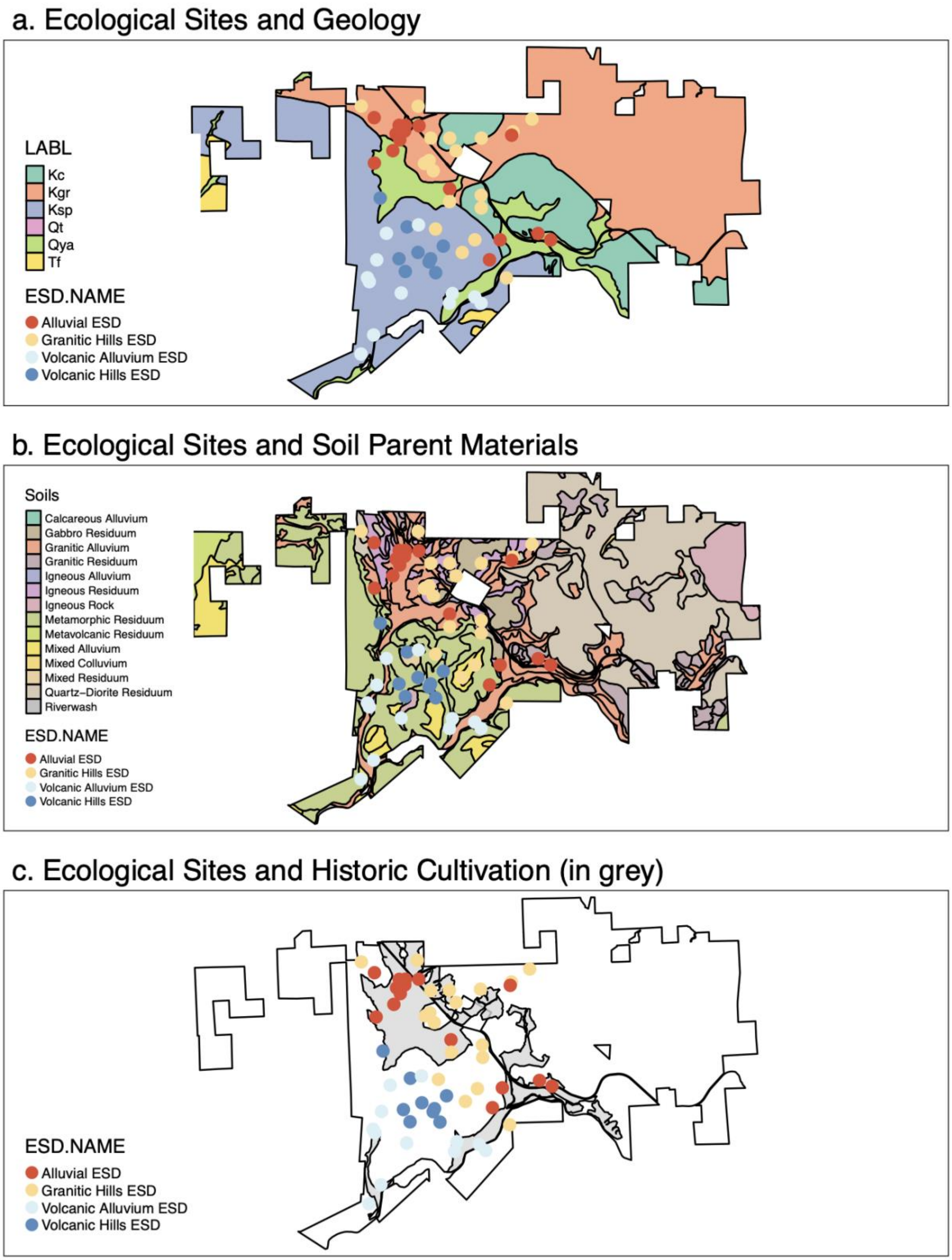


Figure 8. Location of study plots and their ecological site classification in relation to: (a) geology (USGS 2004), (b) soil parent material (SSURGO 2023) and (c) historic cultivation.

Although not based on maps of landform, geology, or historic cultivation, the four ESs correspond fairly well with these features (**Figure 8**). Combinations of these features can be used to more accurately predict the ES of an unsampled area. For example, on hilly landforms with residuum soils-derived from granitic or gabbro parent materials, the ES will very likely be the Granitic Hills ES. Neither the USGS geology map nor the USDA SSURGO soils maps correspond exactly to the ES classification, but taken together, they highlight important differences that are useful for predicting ES status. For example, alluvial areas in the southern portion of RJER are mapped by USGS as 'Santiago Peaks Volcanics' a metavolcanic geology type that does not distinguish between the alluvial floodplains of Jamul Creek and the adjacent hills. The SSURGO database maps these alluvial areas as having soils derived from granitic alluvium. Our ES classification shows that soils in this area share many characteristics with the metavolcanic soils in the adjacent hills. In our classification, these sites are a different ES (Volcanic Alluvium ES) from the alluvial sites occurring in or adjacent to granitic geology.

Descriptions of the four Ecological Sites

Descriptions are summarized in **Table *4***. The **Alluvial ES** occurs on both RJER and HWCA in low-lying positions classified and mapped by USGS as 'Young Alluvium' and 'Granitoid Rock' geology (USGS 2004). The USDA SSURGO database shows this site occurring predominantly on soils derived from Granitic Alluvium parent materials. These sites are relatively flat (0.6 to 15% slope), with sandy soils (61 to 78% sand), and intermediate to high soil fertility. They differ from the other ESs in their particularly high levels of the soil nutrient phosphorus (average of 19 ppm).

The **Volcanic Hills ES** occurs predominantly in the southern hilly portion of RJER mapped by the USGS as 'Santiago Peaks Volcanics', an early cretaceous metavolcanic formation (Herzig and Kimbrough 2014). The USDA SSURGO database shows this site occurring predominantly on soils derived from Metamorphic Residuum parent materials. This site spans a variety of slopes (7 to 42%), soils have a high silt content (28 to 64%) and are moderately fertile. However, soils in this ES have notably low levels of phosphorus (average of 1.8 ppm) and high levels of magnesium (1.7 to 5.2 meq/100g).

The **Granitic Hills ES** occurs predominantly in hilly portions of HCWA and in the western portion of RJER. This site occurs on geologic types classified by USGS as 'Granitoid Rocks', 'Cuyamaca Gabbro', and 'Santiago Peaks Volcanics' – however, we suspect that the areas that it occurs in that have been classified as 'Santiago Peaks Volcanics' are in fact a small inclusion of another geologic type. The USDA SSURGO database shows this site occurring on Igneous Residuum, Quartz-Diorite Residuum, and Mixed-Alluvium. The site occurs on a variety of slopes (4 to 35%) with very sandy soils (67 to 88% sand). Soil nutrients tend to be somewhat lower than the other ESs.

The **Volcanic Alluvium ES** occurs on alluvial soils in low-lying areas adjacent to the Volcanic Hills ES, and colluvial clay lenses within the hilly portion of the Volcanic Hills ES in the southern portion of RJER. It occurs on geologic types classified as 'Santiago Peaks Volcanics' and 'Young Alluvium' by USGS and mapped as soils derived from various geologic types in the USDA SSURGO database. It generally encompasses areas with low slopes (average of 10.5%), however it includes some areas with slopes as high as 30%. It has high levels of soil nutrients and sodium, and generally much higher soil clay content than the other ESs (8 to 42% clay). It also has relatively high magnesium concentrations, including two very high outlier samples (9.5 and 12.1 meq/100g). Plots occurring on this site often stand out for having unusual species and for occurring in areas distinct from their surroundings. Species that occur on this site include Otay tarplant (*Deinandra conjugens*) and saltgrass (*Distichlis spicata*).

Vegetation classification

In order to identify vegetation states, the vegetation data were classified using two different methods. In the first method, we performed a cluster analysis on vegetation data from all plots (regardless of ES). In the second method, we divided the plots into their respective ES, then performed cluster analysis on the subset of plots occurring in each ES.

Cluster analysis of all plots

The cluster dendrogram resulting from the first method had 5 to 8 clearly-defined branches (**Figure 9**). A Mantel test selected 9 clusters as optimal, whereas Indicator Species Analysis selected 7 clusters as best. We selected 8 clusters since it seemed consistent with a reasonable break on the dendrogram and was between the Mantel test and Indicator Species Analysis results.

This vegetation cluster analysis breaks down fairly well along the lines of the ESs. Several states are either exclusively or overwhelmingly represented in one ES, and each Ecological Site has states that are particularly common to it (**Table 5**). This suggests that they support different plant communities and thus represent distinct units from the standpoint of potential vegetation.

Table 5. Number of times a vegetation cluster (based on all plots together) falls into each of the different ecological sites. Bold numbers indicate states that occur predominantly or exclusively on one ecological site.

	Number of Plots in each State occurring in each ES			
Vegetation Cluster	**Alluvial ES**	**Volcanic Hills ES**	**Granitic Hills ES**	**Volcanic Alluvium ES**
Cluster 1	**3**	--	--	--
Cluster 2	3	**14**	--	1
Cluster 3	--	5	2	**10**
Cluster 4	--	--	**7**	--
Cluster 5	11	10	1	1
Cluster 6	2	2	**8**	--
Cluster 7	--	2	2	--
Cluster 8	--	--	--	**2**

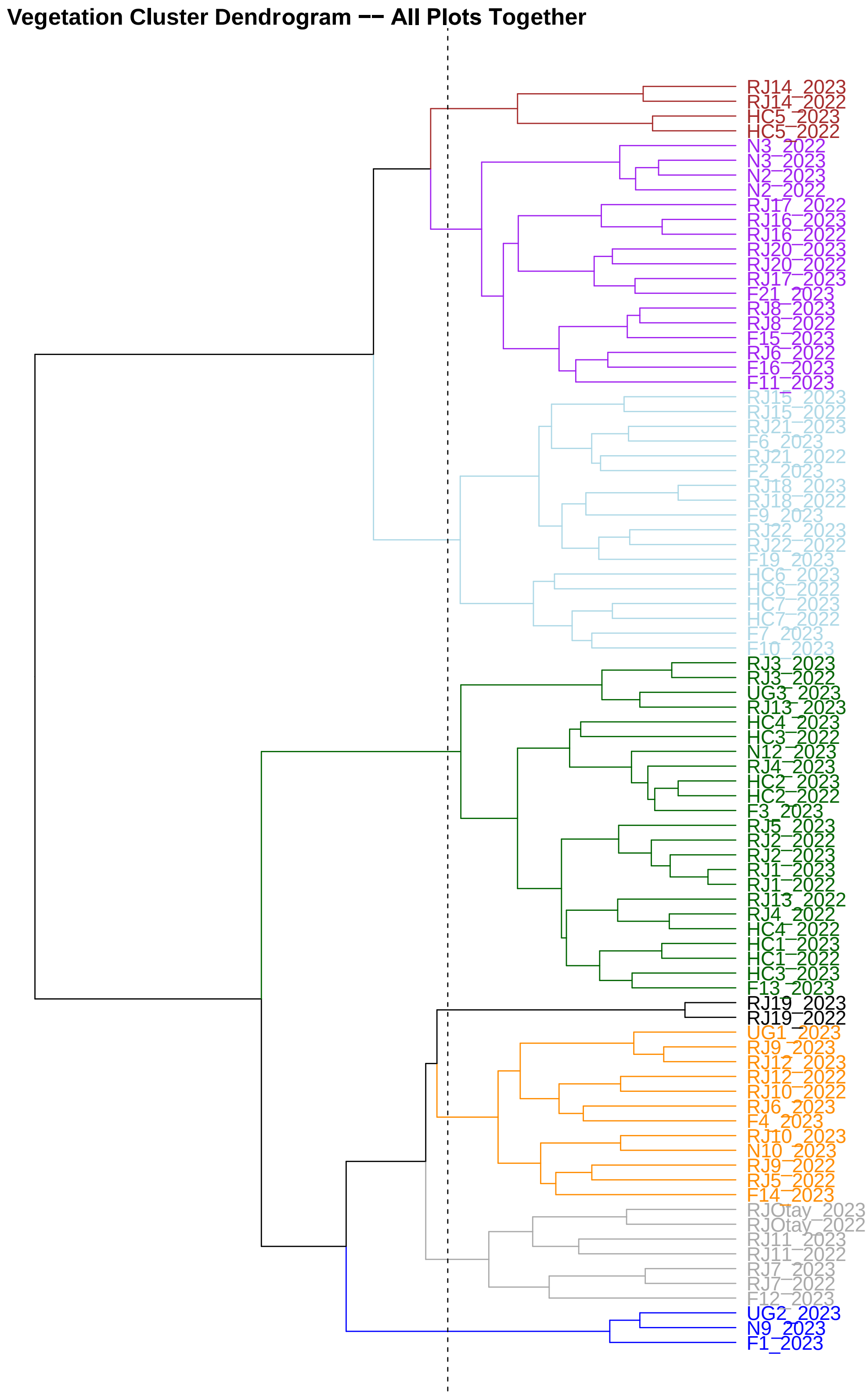


Figure 9. Dendrogram showing results of vegetation cluster analysis performed on all plots together. The dotted line shows where the dendrogram was ‘cut’ to create the clusters.

Vegetation cluster analysis for developing ES vegetation states.
Breaking down the vegetation clusters into vegetation states contributes to the creation of the ESD and STM for an ES. An ESD includes the vegetation states that characterize the ES and the STM models and how they might change given environment and management. To create ES-specific STM models, we performed a vegetation cluster analysis based on the groups of plots broken out by ES. This analysis produced four separate cluster dendrograms (**Figure 10**). We trimmed these dendrograms using a combination of a Mantel Test, Indicator Species Analysis and examination of the dendrograms to identify clear breaks. This resulted in the following numbers of vegetation states per ES: Alluvial ES – 2 States, Volcanic Hills ES – 3 States, Granitic Hills ES – 3 states, Volcanic Alluvium ES – 3 States (**Table *6***).

Very few of the 29 plots surveyed in 2022 occurred in different vegetation states between 2022 and 2023. This is a sign that the classifications used for our vegetation were robust to field sampling error, and that plant composition was relatively stable on the plots despite the very different weather in the 2021-2022 growing season and the 2022-2023 growing season. However, it doesn't identify any specific events as major drivers of changes between vegetation states. One plot in the Volcanic Alluvium ES, RJ11, transitioned from Volcanic Alluvial Native Grasslands to Volcanic Alluvial Exotic Grasslands between 2022 and 2023. However, this transition did not coincide with a significant alteration of grazing practices (although it had fewer AUD/acre of grazing in 2023 than in 2022) or in another major event like a fire. Most likely, cover on this plot was altered by the different weather years and it was enough to tip this plot, already on the margins of a state change, into the Volcanic Alluvial Exotic Grasslands state.

There are no extended time-series data with which to observe transitions over time for the majority of the study plots. However, some of the spatial differences in vegetation states that we see within ESs correspond to different management histories and shed light on the significance of grazing. For example, the Alluvial ES had several ungrazed plots (F1, N9, N10, UG2) that formed a cluster with one grazed plot (F4). These largely ungrazed plots together constituted the entire Ripgut Grasslands State, where ripgut brome has an average cover of 72%. This stands in contrast to the other vegetation state in this ES, Annual Forblands, which is characterized by high cover of annual forbs, including some native species. This suggests that in the Alluvial ES, removal of grazing likely leads to lower forb cover and near-monocultural stands of ripgut brome in many places. It should be noted that three other plots in the Alluvial ES, N12, UG2 and UG3, are also ungrazed. These plots cluster with the other 10 plots in the Annual Forblands state, meaning that some locations within this ES do not appear to transition to the Ripgut Grasslands State in the absence of grazing.

Table 6. Characteristics of the vegetation states based on the classification broken out by each ecological site.

Ecological Site	Cluster	# of Plots	Name	Indicator Species*	Notable Characteristics	Functional Group Cover**
Alluvial ES	1	5	Ripgut Grasslands	None	Near-monocultures of ripgut brome (72% avg). Very few other species present.	Forb (exotic): 18.8% [18.5] Grass (exotic): 76.8% [13.9]
	2	10	Annual Forblands	*Amsinckia menziesii, Erodium cicutarium, Hirschfeldia incana, Medicago polymorpha, Raphanus sativus, Vicia sp.*	Lower cover of ripgut brome (36%) than 'Ripgut Grasslands'. Significant cover (69% avg) and richness of annual forbs (both native and exotic).	Forb (exotic): 60.1% [8.7] Forb (native): 9.2% [3.6] Grass (exotic): 51.2% [10.6] Shrub (native): 3.2% [2.7]
Volcanic Hills ES	1	5	Wild Oat Grasslands	None	Very high cover of wild oats (*Avena fatua*, 40% cover average). High forb cover (40%). May contain low cover of shrubs and needlegrass (*Stipa* sp.).	Forb (exotic): 36.8% [11.3] Forb (native): 3.3% [1.5] Grass (exotic): 93.3% [15.5] Grass (native): 3.7% [1.9] Shrub (native): 18.4% [7.7]
	2	3	Volcanic Sage Scrub	*Bromus madritensis, Calystegia sp., Centaurea melitensis, Salvia apiana*	High cover (52% average) of sage-scrub shrub species. Low cover of annual grasses. Fairly high cover of native forbs (12%). May contain needlegrass (*Stipa* sp.).	Forb (exotic): 13.6% [4.8] Forb (native): 11.8% [9.8] Grass (exotic): 33.8% [11.4] Grass (native): 3.4% [2.5] Shrub (native): 52.0% [11.9]
	3	1	Purple false-brome grasslands	*Brassica nigra, Dipterostemon capitatus*	Similar to Wild-Oat Grasslands, but with high cover of purple false brome (67% average). Low to no forb cover.	Forb (native): 2.0% [NA] Grass (exotic): 116.5% [NA] Shrub (native): 10.0% [NA]
Granitic Hills ES	1	10	Granitic Sage Scrub	*Artemisia californica, Bahiopsis laciniata, Bromus madritensis,*	High cover (32% average) of shrub species. Low non-native grass cover (34%), although red brome can be	Forb (exotic): 21.5% [6.8] Forb (native): 11.0% [1.8] Grass (exotic): 34.2% [9.3] Shrub (native): 32.4% [4.5]

				Eriogonum fasciculatum, Mirabilis laevis, Schismus barbatus	abundant. Fairly high native forb cover (11% on average). By far the highest levels of bare ground of any state (19% on average).	
	2	7	Granitic Native Forbland	*Amsinckia menziesii, Erodium cicutarium, Lupinus bicolor, Malva parviflora, Medicago polymorpha, Raphanus sativus, Vicia. sp.*	High cover of all forbs and native forbs (55% and 17% average respectively). Can have fairly high cover of ripgut brome (27% average).	Forb (exotic): 38.0% [9.8] Forb (native): 17.2% [4.6] Grass (exotic): 57.2 [9.7]
	3	2	Mesa Crypto-gammic	*Acmispon micranthus, Acmispon strigosus, Bromus rubens, Crassula connata, Epilobium sp., Isocoma menziesii, Juncus bufonius, Selaginella bigelovii, Selaginella cinerascens, Silene gallica, Stipa. sp.*	Forb-rich plots occurring in matrix of boulders and cryptogamic soil crusts (especially spike mosses) in the 'mesa' area of RJER.	Forb (exotic): 54.7% [15.8] Forb (native): 4.7% [0.75] Grass (exotic): 39.4% [13.5] Grass (native): 1.0% [1] Shrub (native): 5.2% [5.3] Lycophyte (native): 2.4% [2.5]
Volcanic Alluvium ES	1	4	Alluvial Native Grassland	*Lactuca serriola, Silene gallica*	High cover of native grasses (3.7% average), especially saltgrass and needlegrass. Little to no shrub cover.	Forb (exotic): 36.6% [13.2] Forb (native): 6.6% [2.4] Grass (exotic): 47.9% [11.1] Grass (native): 3.7% [1.4] Shrub (native): 1.0% [0.5]
	2	8	Volcanic Alluvial Exotic Grassland	*Bromus diandrus*	High cover of non-native annual grasses, mostly wild oat and ripgut brome (41% and 32% respectively). Very low cover of native species.	Forb (exotic): 15.3% [4.4] Forb (native): 2.7% [1.6] Grass (exotic): 85.5% [10.2] Grass (native): 0.4% [0.4] Shrub (native): 0.2% [0.3]

	3	1	Volcanic Alluvial Shrubland	*Acmispon glaber, Artemisia californica, Bromus hordeaceus, Centaurea melitensis, Cerastium glomeratum, Corethrogyne filaginifolia, Crassula connata, Daucus pusillus, Dipterostemon capitatus, Festuca myuros, Gutierrezia sp., Hirschfeldia incana, Marah macrocarpa, Oxalis albicans, Pentagramma triangularis, Rhamnus crocea, Salvia apiana, Selaginella cinerascens*	High cover of shrubs (27%), especially redberry buckthorn and California sagebrush.	Forb (exotic): 30.5% [NA] Forb (native): 11.0% [NA] Grass (exotic): 61.0% [NA] Grass (native): 1.5% [NA] Shrub (native): 26.5% [NA]

* Indicator species are included here if the p-value from the Indicator Species Analysis is <0.1.
** Functional group cover is given as the mean followed by [standard error] for each vegetation state. Standard errors were calculated based on plots classified as each state, not based on plot_years.

Figure 10. Dendrograms for the vegetation cluster analyses done for plots in each of the ecological sites. The dotted line shows where the dendrogram was 'cut' to create the clusters.

Resampling of the Fisher Lab Study Plots

The 2022 – 2023 spring vegetation sample shed some light on the drivers of vegetation transitions within the ESs, but these samples only spanned two years and vegetation was remarkably stable at the plot level within those years. To complement this dataset, we resampled 16 plots that were established by Robert Fisher's lab in the early 2000s and sampled through 2012 (Rochester et al. 2010). These included nine plots that burned in the 2003 Otay Fire and 11 that burned in the 2007 Harris Fire. All the plots in this sample that burned in 2003 burned again in 2007, along with two additional plots that did not burn in 2003. This resample provided a useful lens back in time to observe drivers of changes in vegetation cover.

The Fisher Lab vegetation sampling protocol was different from our protocol, meaning that a separate vegetation classification using line-point (percent cover) data had to be created. This classification using the Fisher Lab data (and line point data from all plots in 2022 and 2023) found 8 vegetation states. Of these states, two (states 3 and 7) were considered "shrub" states, with average shrub cover 45.9% and 13.8% respectively. Three states (states 4 and 6) were considered "sparse" shrub states with average shrub cover of 9.5% and 3.1% respectively. One state was considered riparian (State 5) because it has significant cover of sycamore (*Platanus racemosa*), coast live oak (*Quercus agrifolia*) and other riparian species. The remaining three states were considered grassland states with little to no shrub cover (**Figure 11**).

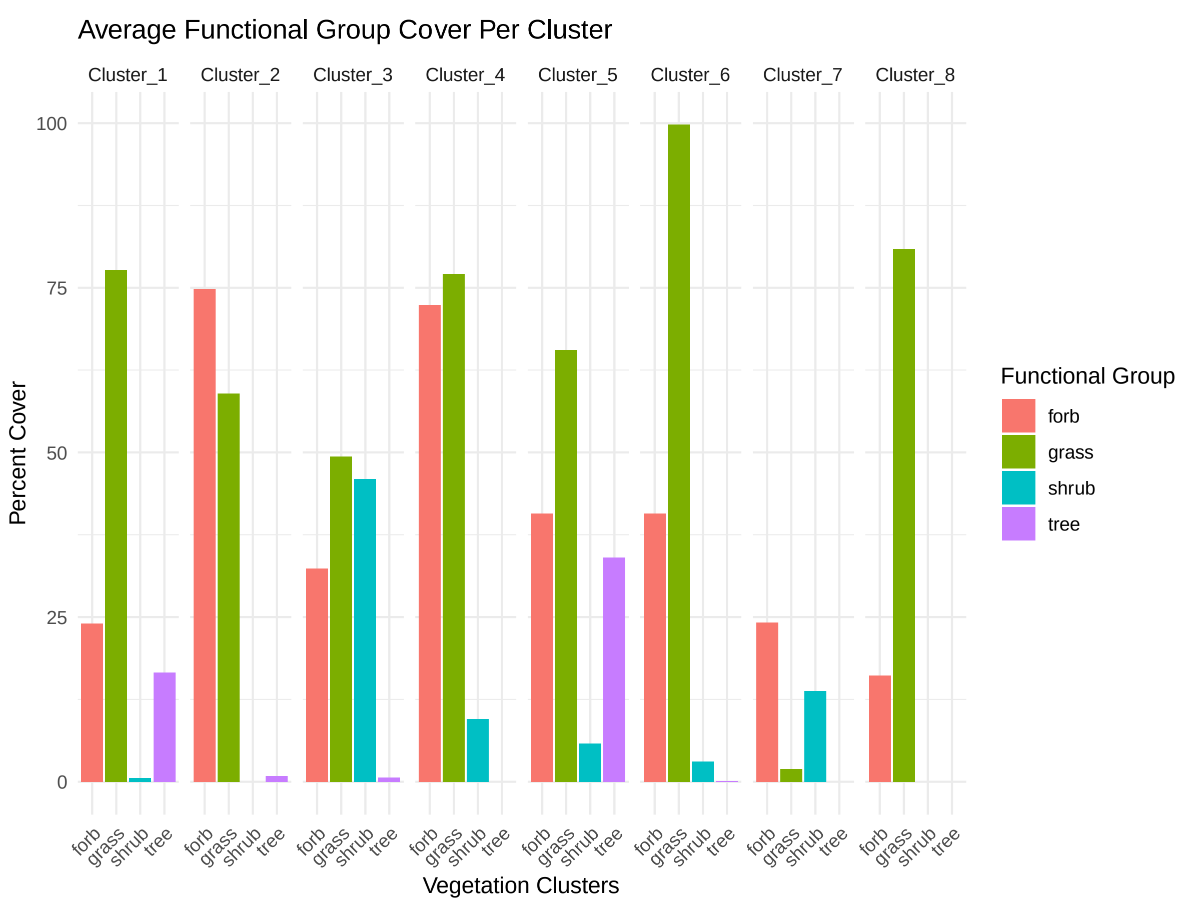


Figure 11. Average percent cover of functional groups in each vegetation cluster from the Fisher Lab data.

We were especially interested in type conversions over time between shrub and grass states. **Table 7** shows transitions between the major vegetation types in the cluster analysis, highlighting different time periods that include the two fires. Six of the eight plots that were classified as “shrub” plots in 2001 converted to one of the “sparse” states in 2005. Every one of these plots burned in the 2003 Otay Fire. The one shrub plot that did not burn in the fire remained in the “shrub” state. Additionally, two of the plots that did not burn in the Otay Fire were in “sparse” states in 2001 and converted to “shrub” states in 2005, suggesting that in the absence of fire, shrub cover was either maintained or increased in this interval (**Table 7**).

Although 11 plots burned in the Harris Fire in 2007, none of those plots changed vegetation types. This is likely because all but two of these plots had already burned in the 2003 Otay Fire. All but one of the plots that burned in the Harris Fire were in the “sparse” class in 2005. The “sparse” plots all remained in that class, suggesting that this class, while containing some shrub species, is somewhat resilient to fire. Shrub cover was already low on the plots that burned in 2007 and it did not significantly decrease after the 2007 fire **(Figure 12**).

In the years since 2008, plots have changed from shrub to sparse states, and vice versa, without wildfire as a driver. These years have been characterized by large swings in the timing and amount of precipitation. In 2014 grazing was reintroduced to portions of RJER and HCWA. We did not detect a systematic pattern in the data that suggests that precipitation or grazing was a consistent or major driver of type conversions to or from shrub types since 2008. Over the 22-year span of these data, the over-riding factor driving transitions from shrub to sparse states was wildfire. The cumulative effects on these plots of burning twice in a short period of time has been both profound and long-lasting. Nine of the eleven plots that burned in either the 2003 or 2007 wildfires burned in both years. Seven of the plots that burned twice during this period were shrub plots and only two of these returned to a shrub state. Total shrub cover on the burned plots remains much lower in 2023 than in 2001 **(Figure 12**). There was a significant decrease in shrub cover on unburned plots between 2012 and 2023, which could be related to grazing, drought, or some other factor during that period. It could also be due to differences in sampling technique or subtle changes in the location of the sampling transects when original transect locations were difficult to relocate.

Table 7. Vegetation change (or lack of change) the 16 resampled plots established by the Fisher Lab. The Harris Fire occurred in 2003 in the first period, the Otay Fire occurred in 2007 in the second period. Bold text shows which plots transitioned from one type to another. Highlighted cells indicate the plots that burned during the given time interval.

	Time Interval (Years)			
Plot	**2001 to 2005**	**2005 to 2008**	**2008 to 2012**	**2012 to 2023**
F1	Grass-Grass	Grass-Grass	Grass-Grass	Grass-Grass
F2	**Sparse-Shrub**	Shrub-Shrub	Shrub-Shrub	**Shrub-Sparse**
F3	Grass-Grass	Grass-Grass	Grass-Grass	Grass-Grass
F4	Sparse-Sparse	Sparse-Sparse	**Sparse-Shrub**	**Shrub-Sparse**
F6	Sparse-Sparse	Sparse-Sparse	**Sparse-Shrub**	**Shrub-Sparse**
F7	**Shrub-Sparse**	Sparse-Sparse	**Sparse-Shrub**	Shrub-Shrub
F9	**Sparse-Shrub**	**Shrub-Sparse**	Sparse-Sparse	**Sparse-Shrub**
F10	Shrub-Shrub	Shrub-Shrub	Shrub-Shrub	Shrub-Shrub
F11	**Shrub-Sparse**	Sparse-Sparse	Sparse-Sparse	Sparse-Sparse
F12	**Shrub-Riparian**	Riparian-Riparian	Riparian-Riparian	**Riparian-Sparse**
F13	**Grass-Sparse**	Sparse-Sparse	Sparse-Sparse	Sparse-Sparse
F14	Sparse-Sparse	Sparse-Sparse	Sparse-Sparse	Sparse-Sparse
F15	**Shrub-Sparse**	Sparse-Sparse	Sparse-Sparse	**Sparse-Shrub**
F16	**Shrub-Sparse**	Sparse-Sparse	Sparse-Sparse	Sparse-Sparse
F19	**Shrub-Sparse**	Sparse-Sparse	Sparse-Sparse	Sparse-Sparse
F21	**Shrub-Sparse**	Sparse-Sparse	Sparse-Sparse	Sparse-Sparse

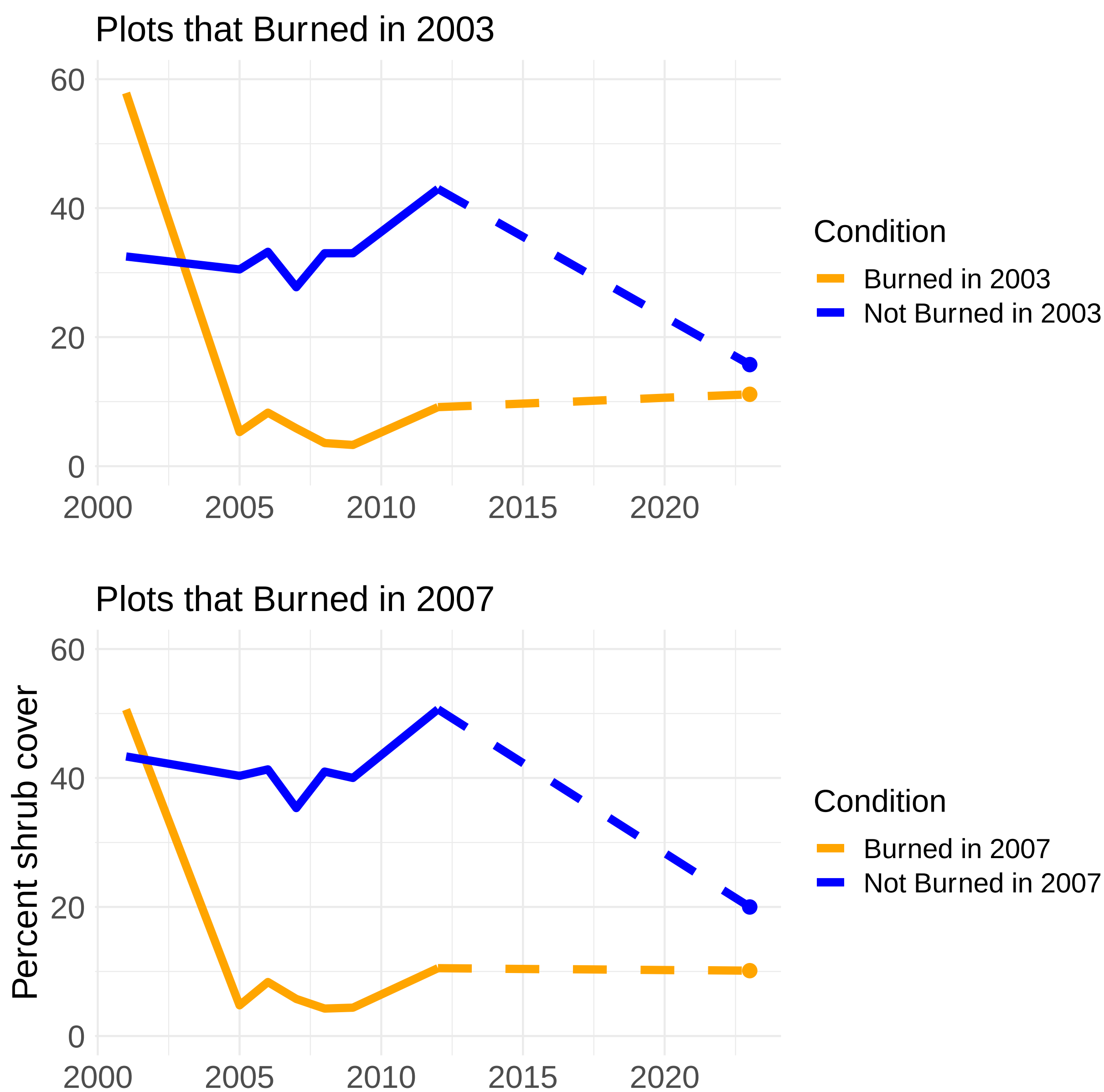


Figure 12. Percent cover of shrubs on Fisher Lab plots at RJER. The dashed lines show years when no sampling occurred. The two plots are very similar because almost all the same plots burned in 2003 as in 2007.

Grazing and wildfire fuels

Most of the herbaceous cover and fuel at RJER and HCWA is non-native annual grasses, which accounted for 56% of total cover in 2022 and 60% cover in 2023. The species vary by ES and vegetation state but the most common grass species are ripgut brome (*Bromus diandrus)* in Alluvial and Volcanic Alluvium ESs, wild oats (*Avena fatua*) in Volcanic Hills and Volcanic Alluvium ESs, slender wild oats (*Avena barbata*) in Granitic Hills ES, and purple false brome (*Brachypodium distachyon)* in the Volcanic Hills ES.

Annual production of herbaceous biomass varied between and within the ESs in 2022 (**Figure 13**) and 2023. In 2023, mean estimated production varied from approximately 3000 lbs./acre in the Granitic Hills ES to 6700 lbs./acre in the Volcanic Clays ES. Ecological site greatly influences the distribution of fuels across the reserves.

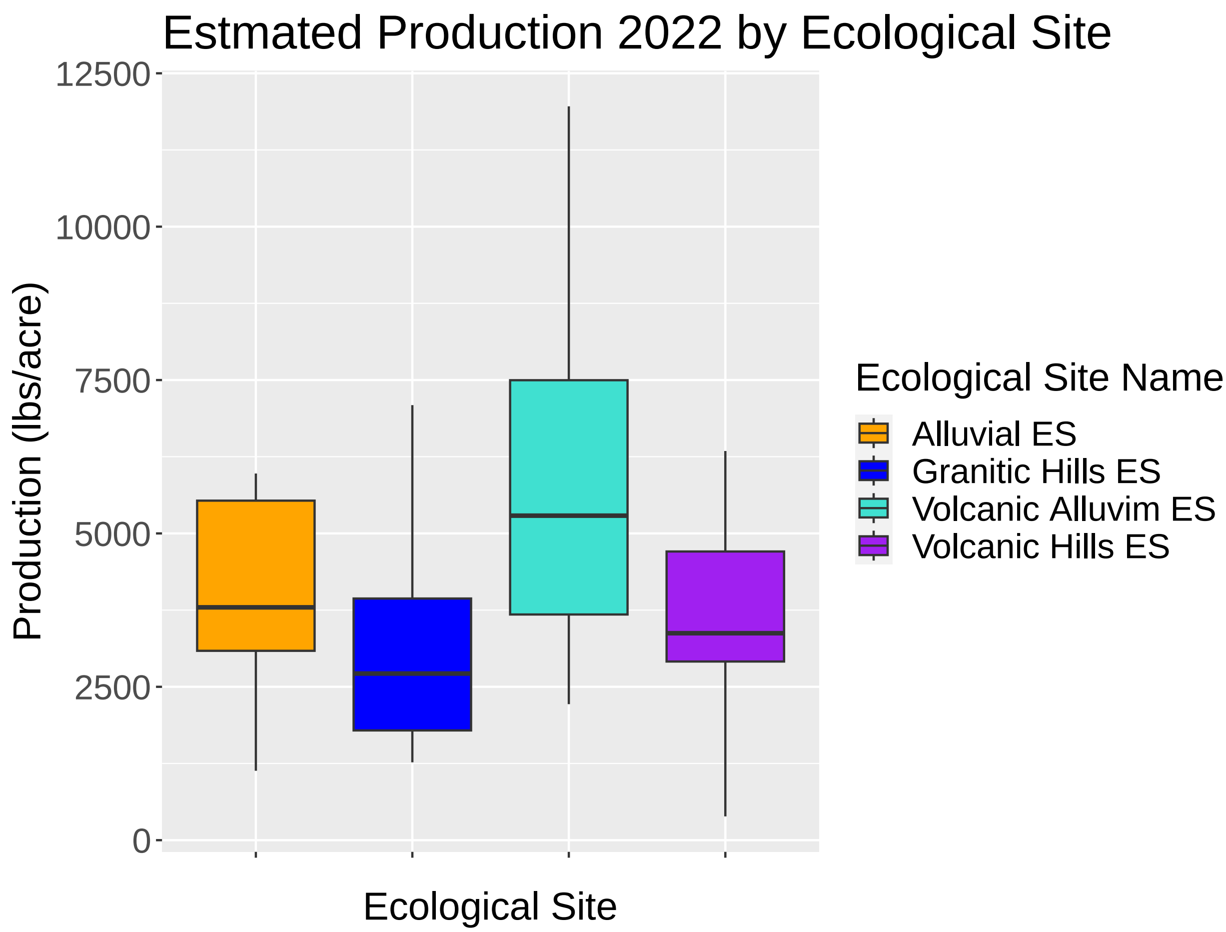


Figure 13. Estimated forage production among the four ecological sites in 2022.

In 2022 and 2023, grazing intensity was highly variable between pastures in the reserves (**Figure *4*** and **Figure *5***). The pattern of grazing is intentionally focused on meeting fuel and habitat standards in different areas of the reserves. Generally speaking, low-lying areas typical of the Alluvial ES, Volcanic Alluvium ES, and some portions of the Granitic Hills ES received more grazing pressure (as measured by Animal Unit Days per acre, while areas in the Volcanic Hills ES received less AUD/ac of grazing).

Interestingly, while overall AUD/acre was light to moderate across the reserves, grazed pastures had significantly lower RDM than ungrazed areas (**Figure 14)**. The RDM difference between grazed and ungrazed areas in 2022 was much greater than one would expect based on consumption by livestock alone. This could be the result of several factors, including: 1) reduction of thatch in grazed areas prevents the buildup of "carry-over" fuels from one year to the next, 2) other processes such as trampling, are removing herbaceous fuel or accelerating decomposition, 3) there are relatively few "ungrazed" plots, and some of these may have inherently higher production than some of the grazed plots for a given ES.

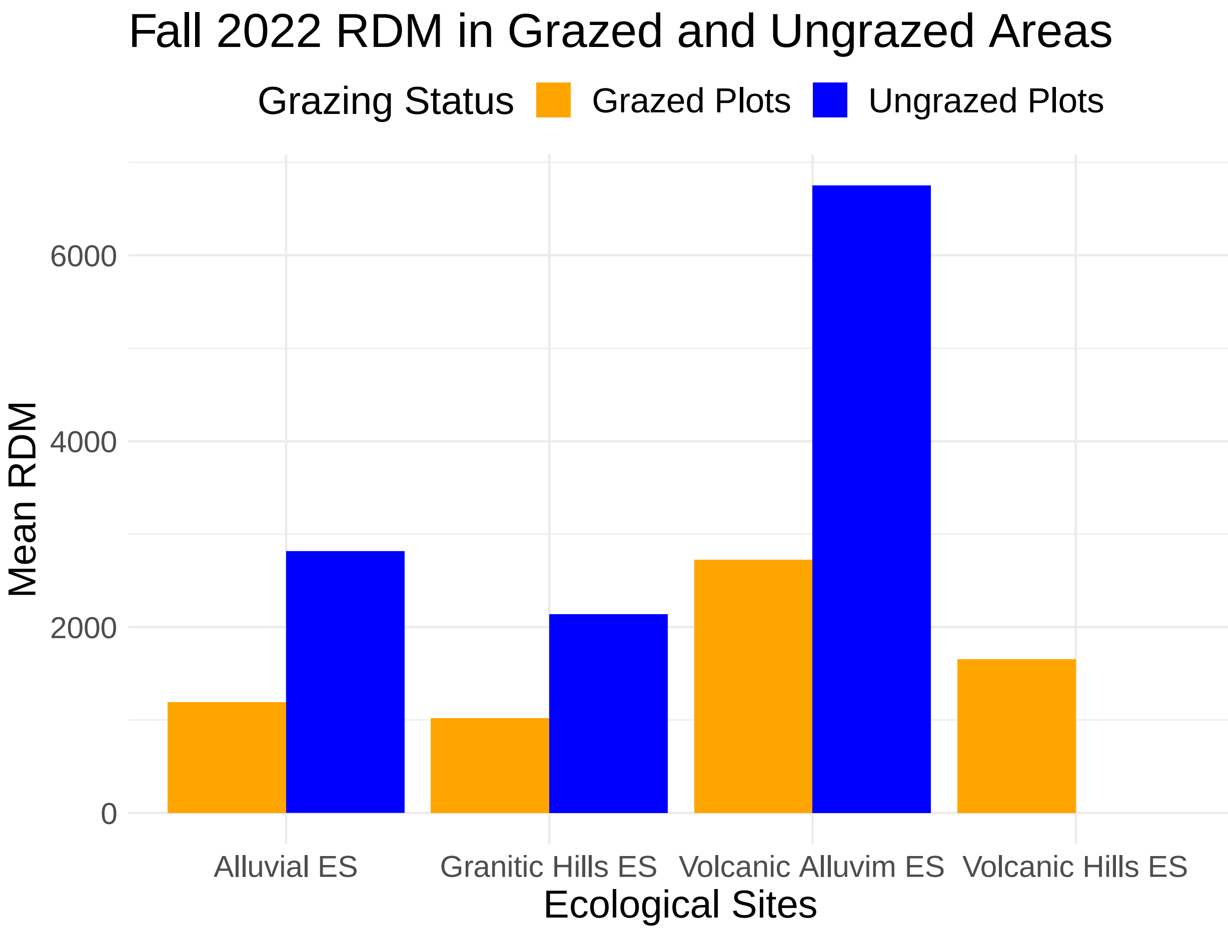


Figure 14. Fall 2022 Residual Dry Matter in grazed and ungrazed plots, broken out by ecological site.

RDM in grazed areas was generally less than 3000 lbs./acre and was kept below 1500 lbs./acre in key strategic areas (i.e., between HWY94 and Rancho Jamul Estates; **Figure 15**). This reflects effective grazing management by the grazing lessee, who targets this area for fuel reduction. Ungrazed areas were consistently >3000 lbs./acre.

Emerging research on grassland fuel and fire behavior in California shows that when areas with fuel loads between 1250 and 2500 lbs./acre burn, the resulting fire may have flame lengths of less than 4 feet. Four feet is a critical flame length threshold for firefighters because at less than 4 feet flame length fires can be fought using hand tools (Andrews and Rothermel 1982; Andrews et al. 2011). When biomass is below 1250 lbs./acre, flame lengths will likely be less than 4 feet and there is a chance that a fire will self-extinguish (Hulme Foss 2023). Conversely, in areas with >3500 lbs./acre, as with most of the ungrazed areas in the reserves, resulting flame heights are likely much higher than 4 feet. These numbers are for non-extreme fire weather conditions. During extreme fire weather, much lower fuel thresholds are expected to be necessary to maintain low flame lengths (Ratcliff et al. 2022). Viewed in this way, grazing is likely having a beneficial impact on fuel loads across the reserves, with a more pronounced benefit in key areas along HWY 94.

The Volcanic Hills ES in the southern portion of Rancho Jamul generally has higher RDM (and fall fuel loads) than other grazed areas of the reserves. To some degree this reflects the high productivity of this ES, but it also reflects the annual rotation of cattle through the reserves. The lessee runs a year-round cow-calf operation and needs to have areas to bring his cattle in the fall and early winter when cows are calving. For this purpose, forage is left in the Volcanic Hills and interior alluvial areas and these areas have higher fall fuel loads as a result.

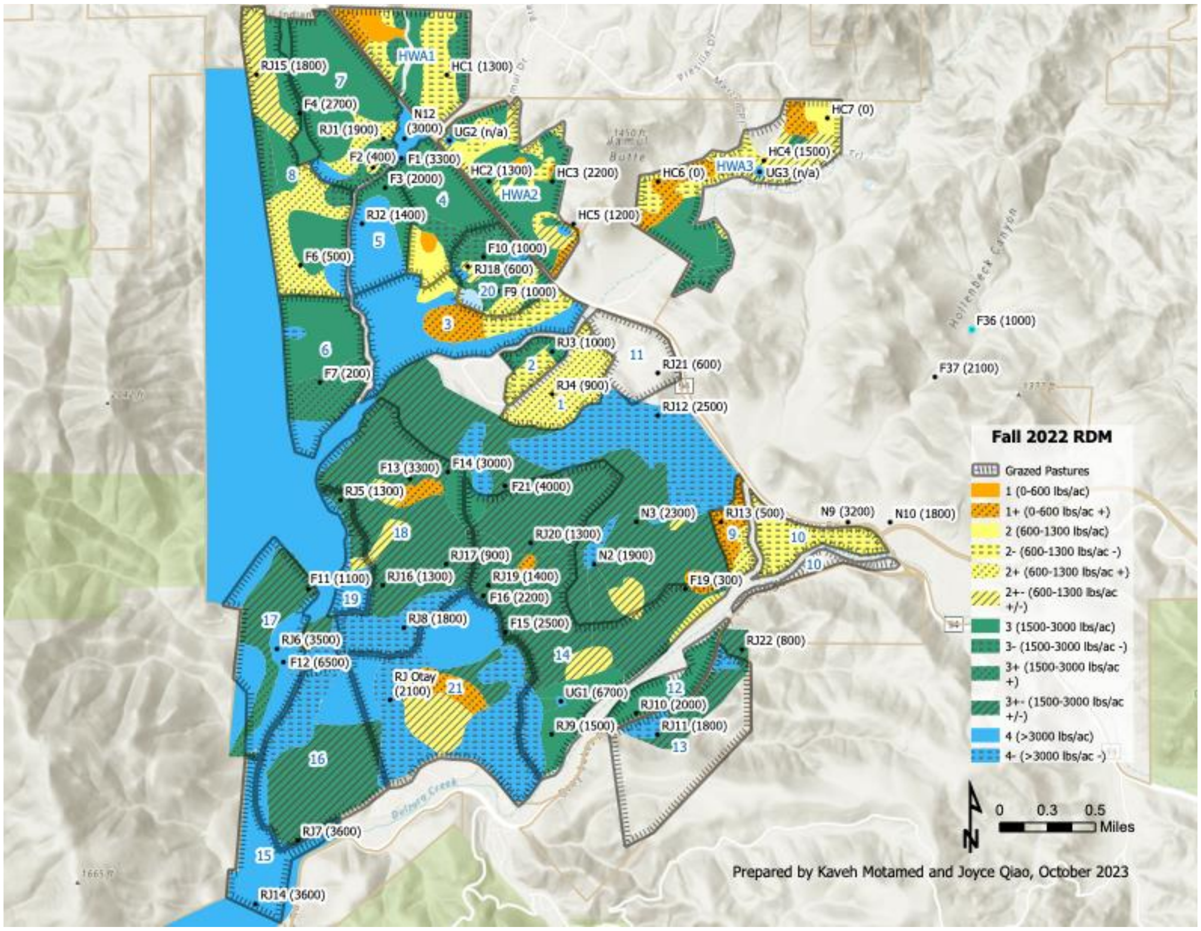


Figure 15. Map of Residual Dry Matter (RDM) across grazing areas in the two reserves in Fall of 2022. The map includes some areas outside the grazing area as well. The "+" and "-" modifiers on some map features indicate that a given map unit contained areas of a higher or lower RDM class than that map unit as a whole, but these smaller areas were too small or patchy to map as separate map units.

Grazing and vegetation change

Documenting potential vegetation change on an ES in an STM is a critical part of an ESD. Based on data from 2022 to 2023 the STMs for the ESs had only one transition among the 29 plots that were monitored in both years where a plot changed from one state to another. That transition, from Volcanic Alluvium Native Grassland to Volcanic Alluvium Exotic Grassland occurred on plot RJ11 in the Volcanic Alluvium ES. Grazing intensity was lighter in that pasture in 2023 than it was in 2022: cattle use observed on the plot in 2022 was "medium" while there was no evidence of cattle use noted at the time of monitoring in 2023 (it was grazed after monitoring occurred). The average vegetation height in 2022 was only 7.8 cm (3.1 in), while it was 26.8 cm (10.6 in) in 2023. In terms of vegetation composition, the plot had much higher cover of wild oats in 2023 than 2022 (87% compared to 12%), but less ripgut brome (6% compared to 50%). Purple needlegrass cover was similar between years. These transitions may well have been driven by the lack of grazing in 2023, but the vastly different rain-years were likely a contributor as well. The ability of the exotics to grow to large sizes during a heavy rain year no doubt played a part, and wild oats are particularly noted for this ability.

Grazing intensity varied between years in many pastures, but vegetation on the 29 plots sampled in both 2022 and 2023 was remarkably stable in the cluster dendrograms. Changes in grazing did not appear to shift any plots from one state to another. Lack of temporal transitions notwithstanding, the variation in states between grazed and ungrazed plots within some ESs (especially the Alluvial ES) suggests that grazing may have a significant effect, especially on herbaceous vegetation.

The Alluvial ES has two states: one dominated by ripgut brome and long-beaked filaree (means of 71.6% and 18.2% cover, respectively), with few other species present, and one with much lower cover of these two species and much higher cover of native and exotic forbs (9% and 60% cover respectively). The ripgut brome dominated state is composed almost entirely of ungrazed plots. The one grazed plot with this state (F4) is in a pasture that had a very low grazing rate in the 2022-2023 grazing year (8 AUD/acre). The other state does occur on two ungrazed plots (N12 and UG3), but the other eight plots where this state occurs are in grazed pastures.

The Granitic Hills ES also has some states that are mostly composed of grazed and ungrazed (or lightly grazed) plots. One of these states (Mesa Cryptogamic) is composed of the two plots on the mesa in RJER. While ungrazed, these plots have different environmental attributes than many of the other plots in the Granitic Hills ES so it is not a surprise that their vegetation differs from other plots in this ES. They have boulders and cobbles at the surface, higher bare ground, and the presence of cryptogamic soil crusts, including spike mosses (*Selaginella spp.*). Four of the other 16 plots in this ES are ungrazed, but they do not cluster together in a systematic way.

Grazing impacts to species and functional groups

At RJER and HCWA, grazing appeared to impact vegetation in different ways, varying based on taxonomic groups, lifecycle, origin, species, and grazing intensity. We use "functional groups" as a way of organizing species into broad categories that may be of conservation management significance, not to suggest that species in these categories provide similar "functions" or have similar functional traits. With functional groups as response variables, we did not find any significant statistical interactions between grazing and ES or year (e.g., weather), however, the occurrence of individual species and functional groups often varies significantly between ESs and years.

Across all ESs, increases in grazing intensity (i.e., more AUD/acre) led to decreases in exotic annual grass cover and increases in exotic annual forb cover (**Table 8**). This was also true when looking at grazing as a categorical variable (grazed/not grazed). Although the interaction between grazing and ES was not statistically significant for any of the functional groups, the negative relationship between AUD/acre and grass cover appeared to be particularly strong in the Alluvial ES. This negative relationship supports the conclusion that removal of grazing can cause transitions from grasslands with higher proportions of forbs to those with near-monocultures of ripgut brome and long beaked filaree.

Some functional groups did not vary significantly with different levels of grazing. The presence and abundance of native perennial grasses and native annual forbs both varied significantly between the ESs but did not significantly covary with grazing. As you would expect, native perennial grass cover did not change significantly between years, while native annual forb cover did (**Table 8**).

Shrub cover did not vary between years but it was highly variable between ESs. Interestingly, there was a statistically significant positive association between grazing intensity and shrub cover (**Table 8**). Individual species varied in their association with grazing: California sagebrush (*Artemisia californica*) and white sage (*Salvia apiana*) cover was positively associated with increased grazing intensity, while laurel sumac cover was negatively associated (**Table 9**). Anecdotal observations suggest cattle use laurel sumac (*Malosma laurina)* for shade and to rub on (breaking low branches in the process), which may be responsible for the negative association.

Table 8. P-value and model coefficients from negative binomial models, with functional groups as response variables. Grazing (AUD/acre), ES and Year are independent variables. P-values shown here are not corrected for multiple-testing.

Response Variable	Independent Variable	Estimate	Std. Error	p-value
Exotic Annual Grasses	Grazing	-0.017	0.008	**0.034**
	Year	0.621	0.141	**<0.001**
	ES	NA	NA	**0.005**
Exotic Annual Forbs	Grazing	0.039	0.015	**0.009**
	Year	0.665	0.264	**0.012**
	ES	NA	NA	0.332
Native Perennial Grasses	Grazing	-0.001	0.030	0.835
	Year	-0.054	0.519	0.917
	ES	NA	NA	**<0.001**
Native Annual Forbs	Grazing	0.017	0.020	0.384
	Year	1.011	0.357	**0.005**
	ES	NA	NA	**0.001**
Native Shrubs	Grazing	0.080	0.028	**0.004**
	Year	0.096	0.489	0.845
	ES	NA	NA	**<0.001**[1]

[1] factor-wide p-value for ES was not possible for the regression with native shrubs because the theta parameter of the negative binomial model could not be estimated without ES in the model, and thus an F-test between models with and without ES was not possible. The result shown here is the lowest p-value for differences between individual levels of the ES factor.

Table 9. P-value and model coefficients from negative binomial models, with the most common species in each functional group as response variables. Grazing (AUD/acre), ES, and Year are independent variables. P-values shown here are not corrected for multiple-testing.

Management Guild	Response Variable	Independent Variable	Estimate	Std. Error	p-value
Exotic Annual Grasses	Ripgut brome (*Bromus diandrus*)	Grazing	-0.0312	0.019	0.100
		Year	0.347	0.338	0.305
		ES	NA	NA	**<0.001**
	Wild oat (*Avena fatua*)	Grazing	-0.065	0.030	**0.032**
		Year	1.479	0.531	**0.005**
		ES	NA	NA	**<0.001**[1]
	Slender wild oat (*Avena barbata*)	Grazing	-0.052	0.029	0.073
		Year	0.121	0.464	**0.009**
		ES	NA	NA	**<0.001**
	Purple false brome (*Brachypodium distachyon*)	Grazing	0.084	0.061	0.173
		Year	1.28	0.836	0.125
		ES	NA	NA	1[1]
	Rattail fescue (*Festuca myuros*)	Grazing	0.064	0.027	**0.015**
		Year	1.121	0.479	**0.019**
		ES	NA	NA	**0.021**
	Needlegrass (*Stipa sp.*)	Grazing	0.002	0.003	0.571
		Year	-0.176	0.500	0.724

Native Perennial Grasses		ES	NA	NA	**<0.001**
	Saltgrass (*Distichlis spicata*)	Grazing	Did not occur at enough sites to evaluate		
		Year			
		ES			
Exotic Annual Forbs	Long beaked filaree (*Erodium botrys*)	Grazing	0.017	0.030	0.576
		Year	0.985	0.532	0.064
		ES	NA	NA	0.793
	Red-stemmed filaree (*Erodium cicutarium*)	Grazing	0.062	0.029	**0.034**
		Year	-0.431	0.530	0.417
		ES	NA	NA	**<0.001**
	Tocalote (*Centaurea melitensis*)	Grazing	0.080	0.026	**0.002**
		Year	0.983	0.471	**0.037**
		ES	NA	NA	0.188
Native Forbs	Annual lupine (*Lupinus bicolor*)	Grazing	0.002	<0.001	**0.011**
		Year	0.568	0.161	**<0.001**
		ES	NA	NA	**<0.001**[1]
	Blue dicks (*Dipterostemon capitatus*)	Grazing	0.042	0.031	0.180
		Year	0.437	0.615	0.477
		ES	NA	NA	**0.004**
	Small-flowered lotus (*Acmispon micranthus*)	Grazing	-0.004	-0.006	0.525
		Year	28.690	211900	1
		ES	NA	NA	0.686
	Strigose lotus (*Acmispon strigosus*)	Grazing	-0.002	-0.006	0.734
		Year	37.060	11790000	1
		ES	NA	NA	0.304[1]
Native Shrubs	California sagebrush (*Artemisia californica*)	Grazing	0.090	0.027	**<0.001**
		Year	0.071	0.453	0.876
		ES	NA	NA	**<0.001**[1]
	California buckwheat (*Eriogonum fasciculatum*)	Grazing	0.021	0.035	0.536
		Year	0.061	0.581	0.917
		ES	NA	NA	**0.007**
	Laurel sumac (*Malosma laurina*)	Grazing	-0.427	-0.190	**0.023**
		Year	-1.031	-0.935	0.912
		ES	NA	NA	**<0.001**
	White sage (*Salvia apiana*)	Grazing	0.094	0.022	**<0.001**
		Year	-0.797	0.231	**<0.001**
		ES	NA	NA	0.993[1]

[1] factor-wide p-value for ES was not possible for the regression because the theta parameter of the negative binomial model could not be estimated without ES in the model, and thus an F-test between models with and without ES was not possible. The result shown here is, the lowest p-value for differences between individual levels of the ES factor.

The positive relationship between shrub cover and grazing is somewhat counter intuitive. However, it is consistent with field observations of browsing on the study plots made in the fall of 2022. For most of the common shrub species only a small proportion of leaders were browsed: California sagebrush, California buckwheat (*Eriogonum fasciculatum*), white sage, and laurel sumac. A few species, coyote brush (*Baccharis pilularis*), desert broom baccharis (*Baccharis sarothroides*), and toyon (*Heteromeles arbutifolia*), showed more frequent signs of browsing, but these species were rare or absent on the study plots. Recent research shows that removing non-native annual grasses aids in survival of California sagebrush seedlings and small adults, likely by increasing available soil moisture (Thomson et al. 2021), such that grazing may lead to increases in sagebrush cover through these mechanisms. Cattle were observed breaking the limbs of laurel sumac and sheltering in their shade. It is plausible that they could decrease laurel sumac (**Table 9**) cover even though this species showed only infrequent browsing on the study plots.

Grazing intensity numbers were derived for each plot based on the total AUD/acre for each pasture, not at the scale of the plot, but they do give us a good sense of the level of grazing at the pasture scale. In the Granitic Hills ES, where shrublands are more likely to have low herbaceous production and typically occur on steeper slopes and higher elevations than grasslands, many shrub plots showed little sign of cattle activity in the year preceding spring vegetation sampling. Grazing may have had limited effects on shrub cover in this ES simply because cattle are not using these areas as much as grasslands. In contrast, the Volcanic Hills ES has a much more integrated matrix of grasslands and shrublands, and the pastures do not generally include large lowland grasslands where cattle may prefer to spend time. Neither of these patterns explain the *positive* relationship between grazing intensity and shrub cover (**Table 9**).

Management of MSP species habitat with grazing

The ES framework provides a useful basis for evaluating MSP species habitat and the impacts of grazing on habitat quality. For many species, suitable habitat only occurs in select ESs, and the primary habitat components potentially influenced by grazing (vegetation structure and species composition) vary between the ESs. Thus, the ES provides a useful filter for evaluating distribution of potential habitat, threats to habitat, and the impacts of grazing with respect to habitat parameters.

Burrowing owl (*Athene cunicularia hypogea*). Western burrowing owls (BUOW) are listed as a Bird of Conservation Concern U.S. Fish & Wildlife Service and a Species of Special Concern by the California Department of Fish and Wildlife. The SDMMP lists BUOW as a species at risk of loss, and BUOW populations and habitat suitability parameters have been monitored annually by the San Diego Zoo Wildlife Alliance and California Department of Fish and Wildlife at RJER for several years (SCZICR 2017). BUOW are a grassland species, preferring areas with short-statured vegetation in foraging areas and near burrows (Gervais et al. 2008, Hammond et al. 2022). Grazing is considered compatible or beneficial for maintaining BUOW habitat (SCZICR 2017; Hammond et al. 2022), however grazing-related habitat management standards have not been established. At Rancho Jamul, there are two areas that currently support breeding BUOW. One population was successfully introduced via active translocation by the San Diego Zoo Wildlife Alliance to the lower slopes of the Granitic Hills ES just south of the historic racetrack, and another was successfully introduced in the northeast most grazing field (close to the Jamul Casino). Grazing management at Rancho Jamul prioritizes maintaining low-statured vegetation in the vicinity of these populations, sometimes requiring multiple grazing periods within a year to maintain low vegetation height during rapid spring growth. Additionally, the San Diego Zoo Wildlife Alliance has created artificial burrows and other habitat structures in these areas, using line-trimmers to keep vegetation very low around the burrows.

All indications from Rancho Jamul are that grazing is a helpful management tool for improving burrowing owl foraging and burrowing habitat. In the Alluvial ES, where one of the colonies occurs, ungrazed areas tend to form dense stands of ripgut brome – a fairly tall-statured grass that can accumulate thatch over multiple years. Spring vegetation height in ungrazed areas of the Alluvial and Granitic Hills ESs (the two ESs where BUOW occur) were as much as twice as tall as in grazed areas of these ESs, and residual dry matter on grazed plots was less than half that of ungrazed plots. In both ESs, grass cover decreased with increasing grazing pressure (AUD/ac), while forb cover increased. BUOW occurrence and persistence at a location is dependent on a combination of factors, including presence of burrows, available insect and small rodent prey, vulnerability to predation, and maintenance of beneficial vegetation structure (Hammond et al. 2022). While grazing appears to improve key structural components of vegetation for BUOW, it is unclear whether it significantly effects other key aspects of BUOW habitat leading to greater incidence or persistence of BUOW in grazed areas. More research on the relationship between grazing and California ground squirrel occupancy, abundance, and diversity of BUOW prey items, and spatial patterns of BUOW occupancy with respect to environmental variables would help clarify the effects of grazing on this species.

California gnatcatcher (*Polioptila californica californica*). Coastal California gnatcatcher (CAGN) is a Federally Threatened species and a State Species of Special Concern. It occurs in coastal sage scrub habitat, with higher occupancy rates in "intact" habitat with higher cover of California sagebrush and California buckwheat (personal communication Barbara Kus and Kristine Preston July 27, 2023). This intact habitat is threatened by frequent human-caused wildfires, which reduce cover of sagebrush and buckwheat and can lead to a higher cover of annual grasses (Syphard et al. 2022), which we have documented at RJER/HCWA. While there are not thresholds in percent cover of any vegetation species for suitable CAGN habitat, occupied sites typically have higher cover of California sagebrush (17% compared to 10% in unoccupied habitat), higher cover of California buckwheat (13% compared to 6% in unoccupied habitat), and lower cover of annual grasses (24% compared to 36% in unoccupied habitat). Laurel sumac, a native shrub, is negatively associated with CAGN occupancy, especially above 30% or 40% cover (personal communication Barbara Kus and Kristine Preston July 27, 2023).

There are only two ESs that have vegetation states with coastal sage scrub cover that approach the vegetation composition required for suitable habitat: the Granitic Hills ES and Volcanic Hills ES . In the Granitic Hills ES, there is only one vegetation state that has a high cover of shrubs, Granitic Sage Scrub. Average shrub cover per plot in this state is 33%. This state has the lowest average annual grass cover of any of the states, also approximately 33%. In the Volcanic Hills ES, there are two states that potentially constitute habitat for CAGN: Volcanic Sage Scrub and Wild Oat Grasslands. These states have 52% and 18% average shrub cover respectively.

Across all ESs grazing is positively associated with shrub cover, in particular California sagebrush, and negatively associated with annual grass cover, especially wild oats. In the

Volcanic Hills ES, where wild oats are the most common annual grass and California sagebrush is present, grazing may be particularly effective in reducing annual grass cover and increasing sagebrush cover. Similarly, laurel sumac cover is negatively associated with grazing. This reduction in sumac cover may improve habitat for CAGN, especially in the Volcanic Hills ES, which has by far the highest cover of this species.

Quino checkerspot butterfly (*Euphydryas editha quino*). The federally endangered Quino checkerspot butterfly inhabits open structured coastal sage scrub in southern California and northern Baja California (Mattoni et al. 1997). Surveys at RJER have identified the species on the mesa and in limited areas in the plateau (TAIC 2006). Prior to 2008, Quino checkerspot butterfly were observed in at least five locations in Hollenbeck Canyon (EDAW 2008), none of which are currently in grazing pastures. Suitable habitat for this species includes a network of small "micro patches" of the primary host plant California plantain (*Plantago erecta*) but must also include diverse topography and nectar resources (USFWS 2003). Invasion by non-native grasses is the greatest threat to Quino checkerspot butterfly reserves because these non-native species displace host and nectar plants (USFWS 2003).

Quino checkerspot butterfly habitat occurs primarily in two of the ESs: Granitic Hills ES and Volcanic Hills ES. This is based on Quino checkerspot butterfly observations in the reserves, the occurrence of host plants and nectar resources among the 54 study plots (forbs and native shrubs), the occurrence of cryptogamic soil crusts on the study plots, and the occurrence of open coastal sage scrub.

Grazing is not considered an appropriate restoration tool for Quino checkerspot butterfly habitat by USFWS because it is thought to reduce cryptogamic soil crusts, allowing further invasion of non-native grasses (USFWS 2003). This study did not evaluate impacts to cryptogamic soil crusts from livestock, and further investigation of this dynamic is warranted to understand the long-term impacts of livestock to Quino checkerspot butterfly habitat and to sensitive cryptogamic soil communities. In the short-term, grazing did effectively reduce cover of non-native annual grasses without negatively impacting shrub cover on the study plots, which may benefit Quino checkerspot butterfly habitat quality. The host plant California plantain was only observed on three study plots among the 2022-2023 sample (F7, HC5 and HC7), and was only seen in 2023. Two of these plots were grazed, but there is not enough occurrence data to determine the impact of grazing to this species in the reserves. Of the nectar plants listed in the Federal Register (USFWS 2002), only California buckwheat is common enough on the plots to be evaluated with relation to grazing. It did not have a significant relationship with grazing on the study plots.

Otay tarplant (*Deinandra conjugens*). Otay tarplant is federally listed as threatened. Between 2013 and 2015, an Otay tarplant study at Rancho Jamul (CBI 2017) by the Conservation Biology Institute (CBI) found that line-trimming and herbicides significantly reduced non-native annual grass cover and increased Otay tarplant density, with herbicides having a stronger impact and being more cost effective. CBI recommended "perpetual treatment," a situation amenable to annual livestock grazing. In 2023, we established study plots in the same vicinity as the CBI study. Apart from a small area that is annually managed with herbicides, the area is not under targeted management for Otay tarplant. Surprisingly, growing among the tall wild oats and ryegrass, we found several plots with Otay tarplant densities that were similar to the herbicide and line-trimming treatment values from the CBI study. This area has largely been ungrazed and showed no visible signs of grazing at the time of monitoring in 2023 (it was subsequently grazed that year). We are in discussion with the grazing lessee at Rancho Jamul to put in experimental grazing exclosures in this area to test the effect of grazing on Otay tarplant density. Based on the observed changes in vegetation structure and composition, it is likely livestock grazing can benefit Otay tarplant, but we will have better information after grazing treatments are installed and monitored.

Phytolith analysis

In our study, total phytoliths, short cell phytoliths, and bilobate phytoliths were extremely abundant across all sites but also varied significantly among ESs (**Figure 16**). Maximum values for short cell and bilobate phytoliths in the study area were several to many times higher than reported total bilobate values for Pepperwood Preserve in northern California (Evett et al. 2013) and total short-cell values for sites across California (Evett and Bartolome 2013). Preservation of phytoliths can also vary by soil texture and other properties, and interestingly the highest phytolith abundances in this study were recorded in ESs with high clay and silt and low sand fractions (**Figure 17**). Kumeyaay agriculture of an unknown grass species may have also influenced phytolith abundance (Shipek 1989). The high phytolith values measured in the study area are very intriguing and merit a reanalysis of the field samples to verify the phytolith findings and potentially, additional investigation to elucidate mechanisms for such high abundances.

Proportional phytolith metrics vary significantly across ESs at RJER/HCWA, suggesting that grasslands were prehistorically present at only a subset of them. We report these findings by ES below.

Alluvial ES. Mean phytolith percent soil weight appears to be greater than 0.3% ($0.1 > p > 0.05$, $df = 14$) suggesting that a significant proportion of this ES was prehistorically grassland, but mean phytolith percent soil weight was 4-5 times lower than that of the Volcanic Hills and Volcanic Alluvium ESs (**Figure 16**). Mean bilobate/total phytoliths did not exceed the 0.1 threshold ($p >> 0.25$, $df = 14$), suggesting *Stipa* and other native perennial grasses were not important components of the grassland flora of this ES. It is unclear what grass species would have been prehistorically present to generate the non-bilobate phytoliths. This site currently supports vegetation states with high exotic grass cover but not native grasses. In addition, a large proportion of this ES was historically, and perhaps prehistorically, cultivated, which may be responsible for generating high phytolith concentrations that are not those of the familiar native grasses.

Volcanic Hills ES. There is a high probability that mean phytolith percent soil weight is significantly greater than 0.3% ($p < 0.001$, $df = 8$), suggesting this ES was prehistorically grass-dominated (**Figure 16**). It also had the highest mean total phytoliths per gram of soil, not statistically different from the Volcanic Alluvium ES but much higher than either the Alluvial or Granitic Hills ESs ($p < 0.001$, $df = 53$). Mean bilobate/total phytoliths was significantly greater than 0.1 ($p << 0.001$, $df = 8$), indicating the historical presence of *Stipa* or other bilobate generating grass species. This ES currently supports vegetation states with purple needlegrass

(*Stipa pulchra*) and small amounts of small-flowered melica (*Melica imperfecta*), both of which generate bilobate phytoliths, as well as very abundant exotic annual grass cover in some states.

Granitic Hills ES.
There was a low probability that the mean phytolith percent soil weight in the Granitic Hills ES is greater than the 0.3% threshold ($p > 0.25$, $df = 17$) suggesting the site did not prehistorically support grasslands. This site also has relatively low mean total phytoliths, comparable to the Alluvial ES. Mean bilobate/total phytoliths also had a low probability of exceeding the 0.1 threshold ($p > 0.20$, $df = 17$) suggesting *Stipa* and other native perennial grasses were not important components of the grassland flora. However, *Stipa* is currently present in very low abundance in some vegetation states at this ES.

Volcanic Alluvium ES.
There is a high probability that mean phytolith percent soil weight at this ES exceeds the 0.3% threshold ($p < 0.001$, $df = 11$), indicating it prehistorically supported grasslands. The site had a very high mean total phytolith per gram of soil, comparable to the Volcanic Hills ES. Mean bilobates/total phytoliths was also likely to exceed 0.10 ($p < 0.01$, $df = 11$) indicating that native perennial grasses were an important component of the site. Vegetation states at this site currently support *Stipa*.

In summary, measured concentrations of phytoliths in soils were high in all plots sampled in this study. All plots supported short cell phytoliths and only two plots had no bilobate phytoliths. These results are not consistent with those of previous phytolith studies in California and we are excited to explore these results further. However, phytolith percentage metrics we evaluated suggest that ESs identified in this study differentially supported prehistoric grasslands. The Granitic Hills ES does not appear to have prehistorically supported grasslands. However, it is likely that the Volcanic Hills and Volcanic Alluvium ESs prehistorically supported native grasslands with bilobate generating species such as *Stipa* and *Melica*. However, the shape of the bilobate phytoliths in the samples collected from RJER/HCWA are somewhat unusual (Evett personal communication), which potentially merits additional analysis. The results for the Alluvial ES are somewhat equivocal regarding its prehistoric vegetation composition. Mean phytolith percent soil weight and mean total phytolith concentration at this ES suggest that it probably supported grasslands. However, mean bilobate/total phytolith is low at this ES suggesting native bilobate-generating grasses were less important in the grasslands that were present. It is unclear what grass taxa would have been prehistorically responsible for generating non-bilobate phytoliths at the Alluvial ES.

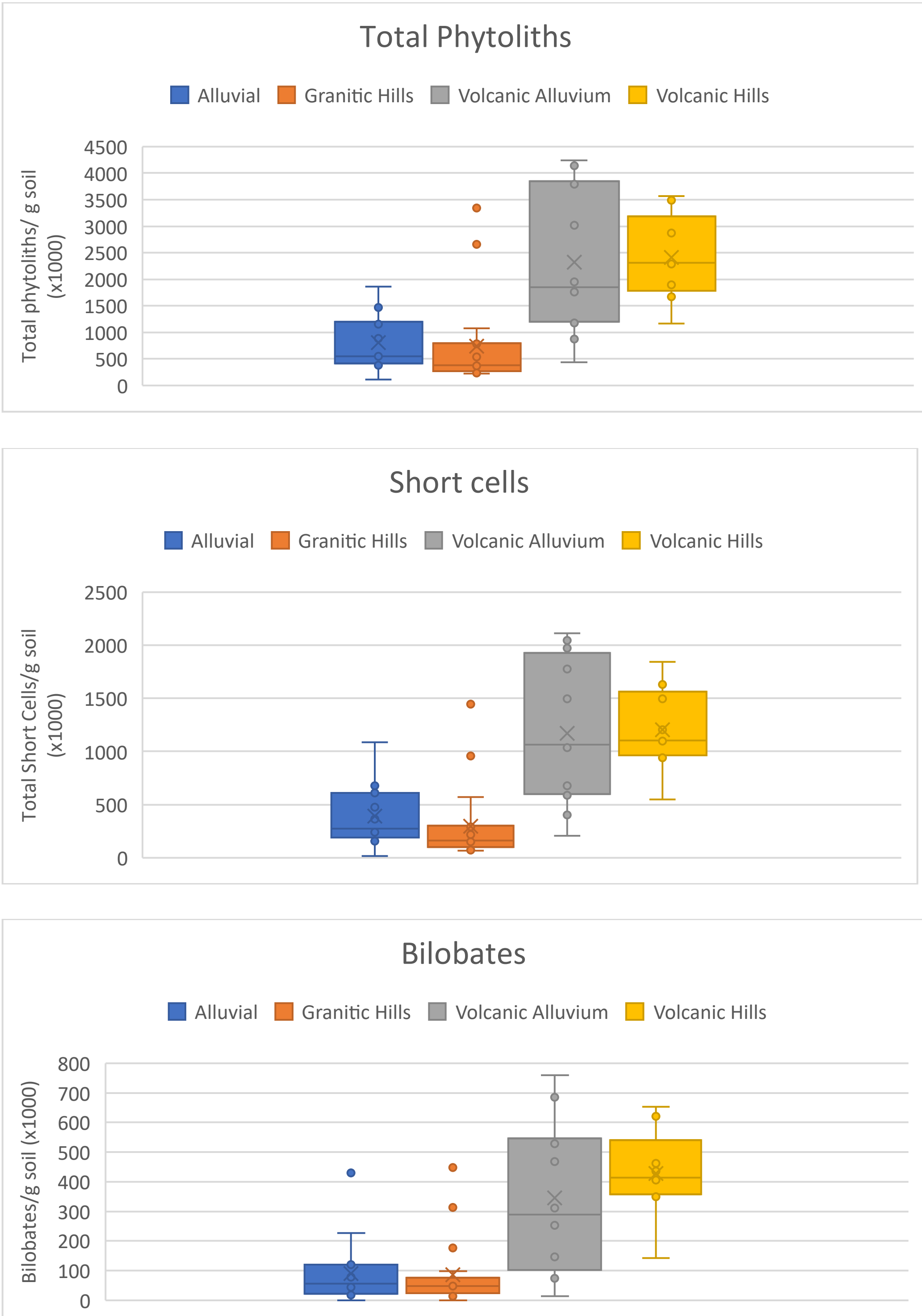


Figure 16. Box and whisker plots of phytolith abundance results by ecological site.

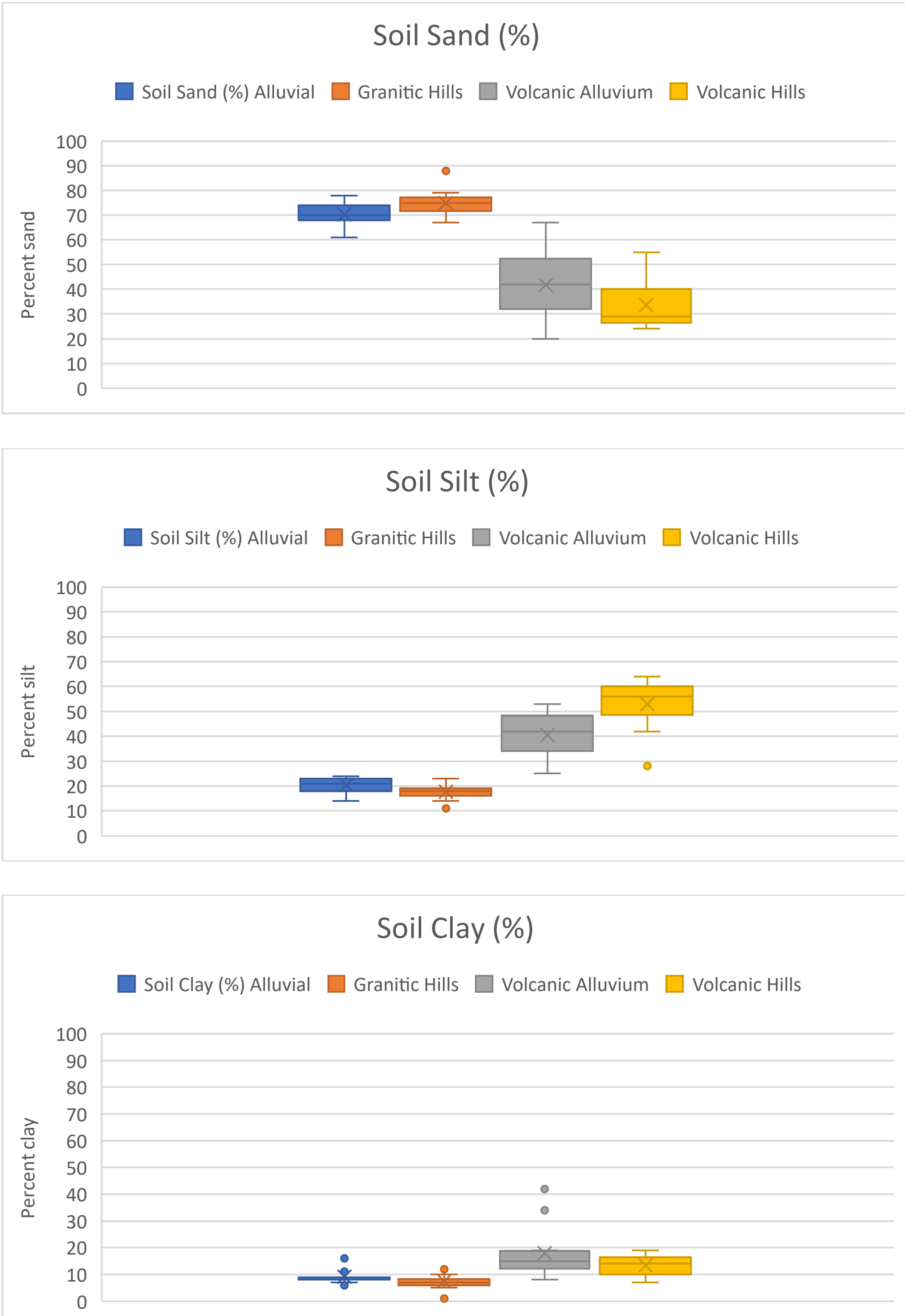
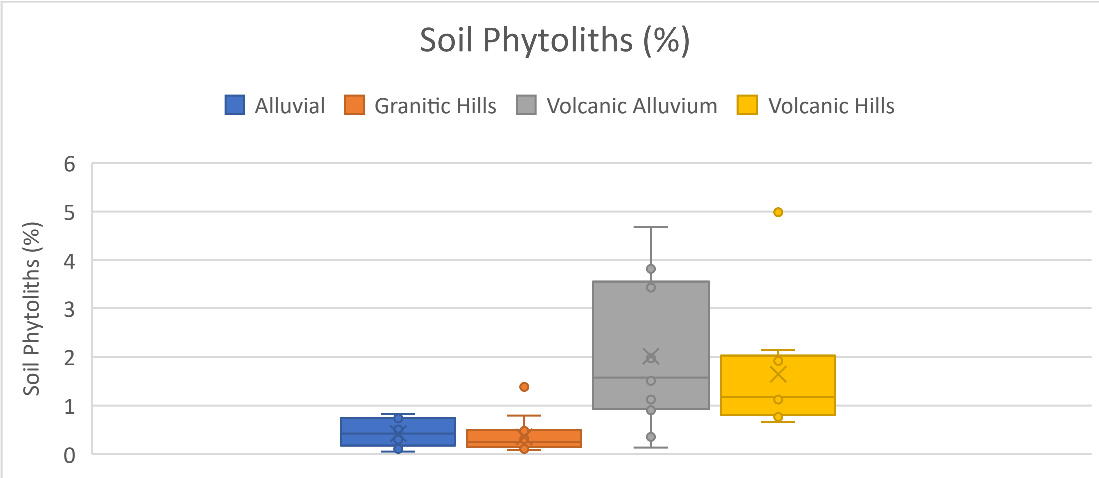


Figure 17. Soil texture results by ecological site.



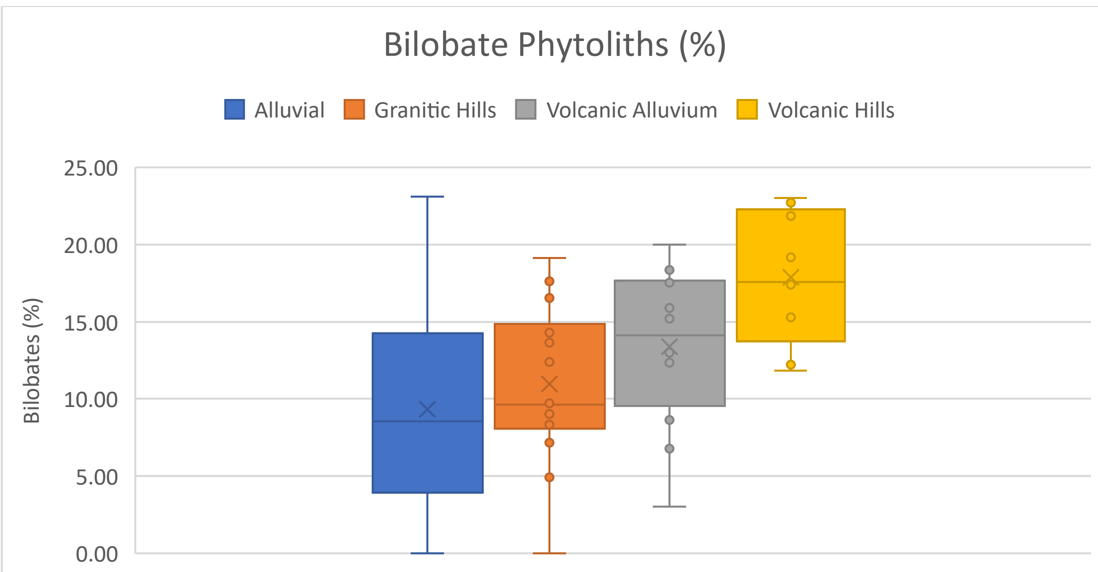


Figure 18. Box and whisker plots of phytolith percentage results by ecological site.

Discussion

In this project we evaluated the potential for grazing to enhance conservation values in grasslands and coastal sage scrub habitats while simultaneously reducing fire hazard from non-native annual grasses.

A summation of our primary findings to date includes the following:

1. The 24 grazed pastures showed variable **grazing use** at levels within range management guidelines given the normal variations in site potential and weather (Bartolome et al. 2006). A number of pastures were ungrazed or had very low grazing intensity.

2. We applied **Ecological Site Classification** to define ESs, determine environmental characteristics, and identify drivers. We applied Random Forest variable selection and cluster analysis to classify and create ESDs (see ESD Classification section). We identified and named 4 clusters representative of distinctive ESs: Alluvial, Volcanic Hills, Granitic Hills, and Volcanic Alluvium. These useful landscape categories are described in detail.

3. **Historic wildfires:** We re-sampled 16 plots, established by Robert Fisher's lab in the early 2000's, that had experienced two wildfires since 2003. The effects of grazing, weather, or a single wildfire on shrub-grass conversions appear minimal. In contrast, two fires in succession (2003, 2007) significantly reduced shrub abundance, with detectable effects nearly twenty years later.

4. **Grazing and wildfire fuels**: Herbaceous fuel loads, largely from annual plants, vary spatially across RJER and HCWA and fluctuate through the year as a result of the annual production cycle, decomposition, and herbivory (including cattle grazing). The plant species vary by ES and vegetation state. In 2023, mean estimated production varied from approximately 3000 lbs./acre in the Granitic Hills ES to 6700 lbs./acre in the Volcanic Alluvium ES. This pattern greatly influences the distribution of fuels across the reserve. Grazed pastures had much lower fall fuel loads than ungrazed areas, generally falling into the lower fuel load categories expected to have less extreme fire behaviors.

5. **Grazing in STMs**: During our study only one plot changed vegetation states between years. There is no evidence to suggest this was due to changes in grazing use.

6. **Grazing effects on plant guilds:** Some functional groups remained stable in composition even with different levels of grazing. The distribution of native perennial grasses and native annual forbs both varied significantly among the ESs but did not significantly co-vary with grazing. Non-native grass cover was significantly negatively related to grazing, while non-native forb cover was significantly positively related to grazing. There was a statistically significant positive association between grazing and shrub cover. Individual shrub species varied in their association with grazing: California sagebrush and white sage were positively associated with increased grazing intensity, while laurel sumac was negatively associated with grazing. These results are from only two years of monitoring.

Grazing effects may be different in years with different weather patterns (such as drought or high-precipitation years).

7. **Phytolith results:** Grasses form and deposit abundant phytoliths in soils, whereas forbs and shrubs produce them relatively rarely, so high abundance of phytoliths in soils can be an indicator of prehistoric grassland vegetation states. In our study, total phytoliths, short-cell phytoliths, and bilobate phytoliths were extremely abundant across all sites but also varied significantly among ESs. The high phytolith values measured in the study area are very intriguing and merit a reanalysis of the field samples to verify the phytolith findings and potentially additional investigation to elucidate mechanisms for such high abundances.

The ESD and STM approach applied to this study was essential for working out spatial and temporal relationships among plant species, community composition, structure, and grazing. More years of monitoring and expansion to new sites will allow us to better work out these interdependent factors, but already a picture is emerging of the ability of grazing to meet management objectives on the reserves.

Below are answers for each of the primary study questions:

1) How effective is grazing at reducing fire risk?

Results from the first two years of study show that grazing can effectively reduce fuel loads from annual grass biomass in grasslands and coastal sage scrub at RJER/HCWA. At the same time, detrimental grazing impacts to important coastal sage scrub species like California sagebrush and California buckwheat were not seen. The efficacy of grazing to reduce herbaceous fuel loads is determined by many factors, principal among them is how much production occurs in a given year (which is weather and site driven), and when, where, and how much livestock grazing occurs. The cow-calf grazing operator can only increase the herd incrementally between years, so very high forage production years complicate the ability to meet biomass reduction needs. The two reserves have multiple management goals relating to biomass reduction through grazing, and the reserve managers and ranching lessee do a good job of prioritizing target areas in high-production years.

2) Can grazing effectively enhance disturbed native grassland and forbland habitats?

Each of the four ESs contains grassland vegetation states. In some of the sites, grazing appears to have a pronounced effect on species composition, but it manifests differently among the sites. In the Alluvial ES, ungrazed plots were far more likely to fall into the “Ripgut Grasslands” state with high annual grass cover and low forb cover. Grazed plots in this ES had much higher cover of forbs and generally had higher species richness – including more native forb species. The increase in non-native forb cover in grazed grasslands in this ES was overwhelmingly driven by increases in filaree (Erodium spp.). While not native, plants of this genus provide high-quality seed for granivorous rodents (Brown et al. 1979; Brock and Kelt 2003).

Non-native annual grasses had a statistically-significantly negative relationship with livestock grazing, but the species in question varied among ESs. Ripgut brome was very common on ungrazed Alluvial and Granitic Hills ESs, but less common in Volcanic Hills and Volcanic Alluvium ESs. On the two volcanic ESs, wild oats were much more common and were also reduced by livestock grazing. Purple false brome is also common in the Volcanic Hills ES, however its cover was not significantly related to livestock grazing intensity in the two years of monitoring.

Native perennial grass abundance was not strongly related to livestock grazing, however their presence was strongly tied to ES. Needlegrasses were far more common in the Volcanic Hills ES and Volcanic Alluvium ES than the others, a finding supported by the phytolith analyses. Given the distribution of perennial grasses and the phytolith record, enhancing native perennial grasses is likely to be a feasible management goal only on some ESs.

3) Can grazing enhance disturbed native coastal sage scrub habitat?

This question is complex and not yet fully answered by this study. There are multiple ways livestock grazing may affect shrublands. These vary in spatial and temporal scale and in the mechanisms of impact. Below are preliminary observations and results from this study that relate to livestock impacts on coastal sage scrub habitat:

Direct impacts to shrubs. There was little evidence of cattle browsing on most coastal sage scrub shrub species. California sagebrush and California buckwheat had very little evidence of browsing. Cover of California sagebrush was positively associated with grazing intensity at the pasture-level as well, indicating that it is not being directly harmed by livestock. However, grazing intensity on some shrub plots may have been less than on the pasture as a whole. Some species, like laurel sumac, do appear to be impacted by livestock. We did not observe browsing on this species, but cattle appear to break low branches to create shade opportunities under these tall-statured shrubs. Shrubs in the extremely high impact areas like next to a water trough may be browsed indiscriminately.

Reduction of wildfire impacts to coastal sage scrub. One hypothesized benefit of livestock grazing in coastal sage scrub is the reduction in frequency and severity of wildfires due to reduction of herbaceous fuels from annual grasses. While we did see significant reduction of fuels in both grasslands and shrublands, it is difficult to say how that translates to fire frequency and severity. Wildfire behavior is highly dependent on weather and topography, and the location of fires is also dependent on the site of ignition. Wildfires may be more easily controlled on the reserves due to grazing, and the probability of a fire that is difficult to fight may be reduced, but the ultimate impacts of wildfire to coastal sage scrub ecosystems depend on variables that are more difficult to predict in advance and will require more work.

Increased germination and seedling survival. Another plausible mechanism for enhancing shrublands is that livestock grazing could eliminate thatch build-up and reduce competition from annual grasses for water, light, and nutrients, thereby enhancing shrub germination and seedling survival. We observed many shrub seedlings on study plots, especially of California sagebrush, but we did not yet test this hypothesis. The answer is likely to vary by ES. The Volcanic Hills ES generally had much more herbaceous biomass produced per acre than the Granitic Hills ES, so if competition from annual grasses is a major factor in shrub reproduction and survival, grazing may have a bigger impact there.

4) ***Can grazing reduce non-native grass and forb cover in disturbed coastal sage scrub to increase native shrub cover and bare ground and improve habitat for MSP species such as Quino checkerspot butterfly, California gnatcatcher, and black-tailed jackrabbit?***

There are several MSP species that could be affected by livestock grazing. Many of these species, including Quino checkerspot butterfly, California gnatcatcher, and burrowing owl are considered to be threatened by ecosystem changes due to non-native annual grasses. Grazing did effectively reduce annual grass height, cover, and biomass across ESs. In the case of burrowing owl, ungrazed study plots in the Alluvial ES generally had far less favorable habitat than grazed plots. We did not study the owl populations or demographics themselves; processes such as predation, prey availability, and burrow availability may take precedence over herbaceous vegetation structure.

Grazed study plots also had lower residual biomass and vegetation height in the Volcanic Alluvial ES, where the Otay tarplant occurs. Based on prior research at Rancho Jamul Ecological Reserve, these conditions are likely beneficial for Otay tarplant, however the relationship needs to be more rigorously tested. We collected Otay tarplant frequency data prior to any experimental grazing treatment in 2023. It would be helpful to install experimental exclosures in future years to test the impact of grazing on this species.

Relationships between grazing and other MSP species deserve a closer look as well. There are important questions remaining about the impact of grazing to habitat elements like cryptogamic soil crusts (which are considered important elements of Quino checkerspot butterfly habitat), ground squirrel occurrence (a critical factor for Burrowing Owl occupancy), invertebrate species and abundance (important prey for several birds including Tricolored Blackbird), and habitat elements relating to other key species like California Gnatcatcher and black-tailed jackrabbit.

Next Steps

Fuel risk

We have made good progress measuring the distribution and amounts of herbaceous fuels under cattle grazing. Current grazing use appropriately targets areas with high ignition probability, which are primarily along main highways. We documented both a significant reduction in herbaceous biomass/fuels and a significant effect on the vertical distribution and porosity of potential fuels with grazing. Grazing has so far shown minimal effects on woody fuels and more work is needed. We are also in a position to better understand variations in the relationship of vegetation structure (e.g., height) and RDM across ESs, which would provide improved information for fire managers.

Native grasses and forbs
The abundance and distribution of native grasses and forbs appear to be closely tied to small scale differences in site characteristics and may be strongly influenced by site history (wildfire, prior cultivation, and grazing). The relationship of native plant species and grazing appears to be complex and highly site specific. Native grasses like *Stipa pulchra* appear to be restricted to specific ESs and monitoring and restoration efforts should be focused on those sites. Phytolith analyses were also intriguing and merit additional work. Values appear to be high, and we found a novel shaped bilobate phytolith, possibly indicating prehistoric presence of a native grass species that is now locally extinct. Further investigation into the impacts of historic tilling on species composition and phytolith preservation would shed light on vegetation drivers and ES potential.

Coastal shrub habitat
Conclusions about this vegetation type require a longer time series and additional demographic data. We have initiated shrub seedling monitoring in our plots to follow seedling dynamics more closely. Additional work investigating relationships between grazing, ES, and recruitment and cover of California sagebrush and California buckwheat is warranted.

MSP species
Grazing appears to have a positive effect on habitat quality for burrowing owl and Otay tarplant, and potentially on California gnatcatcher and other CSS-associated species. Additional work, including applying grazing modifications, is merited to better understand the mechanisms associated with increased in CSS shrub species and the potential negative effects on cryptogamic soil crusts and habitat of species such as tricolored blackbird and Quino checkerspot butterfly.

New study sites
Additional study sites would expand our catalog of ESs and the potential to use grazing to achieve conservation management goals on them. New study areas could also potentially expand our work into additional MSP habitats supporting species such as San Diego fairy shrimp and Stephens' kangaroo rat.

Experimental grazing treatments
The current study documented existing ESs and tied patterns of grazing use to distribution of fuels, to ESs and STMs, and to MSP objectives. In the next phase of the study, we will selectively begin modifying conservation grazing practices.

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Appendices

Appendix A. MSP Species Occurring at Rancho Jamul Ecological Reserve and Hollenbeck Canyon Wildlife Area

Table A1.a. MSP plant species occurring at Rancho Jamul Ecological Reserve and Hollenbeck Canyon Wildlife Area.

Common Name	Scientific Name
Otay tarplant	*Deinandra conjugens*
San Diego thornmint	*Acanthomintha ilicifolia*
San Diego sunflower	*Viguiera laciniata*
Variegated dudleya	*Dudleya variegata*
Felt-leaved monardella	*Monardella hypoleuca* ssp. *lanata*
Cleveland's goldenstar	*Muilla clevelandii*
San Miguel savory	*Satureja chandleri*
San Diego butterweed	*Senecio gander*
Parry's tetracoccus	*Tetracoccus diocus*
Palmer's ericameria	*Ericameria palmeri* ssp. *palmeri*
Snake cholla	*Cylindropuntia californica* var. *californica*

Table A1.b. MSP animal species occurring at Rancho Jamul Ecological Reserve and Hollenbeck Canyon Wildlife Area.

Common Name	Scientific Name
Quino checkerspot butterfly	*Euphydryas editha quino*
Hermes copper butterfly	*Lycaena hermes*
Coast horned lizard	*Phrynosoma coronatum*
Orange-throated whiptail	*Aspidoscelis (Cnemidophorus) hyperythrus*
Golden eagle	*Aquila chrysaetos*
Burrowing owl	*Athene cunicularia hypugea*
Peregrine falcon	*Falco peregrinus*
Bell's vireo	*Vireo bellii*
California gnatcatcher	*Polioptila californica*
Western bluebird	*Sialia mexicana*
Rufous-crowned sparrow	*Aimophila ruficeps*
Savannah sparrow	*Passerculus sandwichensis*
Mountain lion	*Puma concolor*
Mule deer	*Odocoileus hemionus*

Appendix B. Detailed Sampling and Analysis Methods

Plot layout

We established 10x10-meter square plots at each of the sampling plots. At locations where we resampled plots established by Robert Fisher's lab, we used a 50x2-meter belt transect to best represent the areas in their original sample. Each 10x10-meter plot had two wooden stakes installed in the spring of 2022: one at the NW and one at the SE corner of the plot, with a GPS location of the NW corner captured.

Plots were rejected if they had one of the following characteristics: 1) spanned more than one distinctly different aspect; 2) included more than one distinctly different slope class; 3) included significant human disturbance (e.g., a road or other infrastructure); 4) included a cattle "service area," e.g., watering trough, feeding location, mineral lick, etc. If the plot was rejected, we shifted the NW corner of the plot to the SE corner of the plot and reestablished it (i.e., previous SW corner becomes the new NW corner). If that plot was also rejected, we selected the closest random plot location.

Soil chemistry, texture, and phytoliths

At each plot we collected soil with a soil auger from 0-15 cm depth at four locations and combined them into a single composite sample. Soil was dried at 65º C for 72 hours. A portion of each soil sample was sent to the UC Davis Chemistry Laboratory. Another portion of each sample was sent to Dr. Rand Evett for phytolith analyses, where they were processed as described in Evett and Bartolome (2013).

Spring vegetation composition and production

We measured vegetation composition and production in the spring seasons of 2022 and 2023. Vegetation data were collected using a combination of line-point intercept, line-intercept, a modified relevé, and herbaceous biomass. The 16 Fisher plots at RJER (Rochester et al. 2010) were sampled the same way, except that the transect arrangement and relevé area are different (see below). We measured species richness by recording all plant species rooted within the 10x10m plot (or relevé).

Line-point intercept sampling was conducted along five parallel transects oriented north-south at 2.5-meter intervals and sampled from the westmost (0) to the eastmost (5). Points were sampled with a sampling pin along these transects every 0.5 meters (starting at 0.5 meters), yielding 100 samples per plot. At each point location, the first (top) plant hit by the pin was recorded. Then, all subsequent hits representing new species for that point location were recorded. Height of the tallest plant hit was recorded. If the top hit was over 2 meters tall, then the height of the top hit below 2 meters was also recorded. In 2023, we added notation of the soil substrate type at each point location (e.g., bare rock, cobblestone, cryptogamic, leaflitter, moss/lichen, organic dirt, sandy soil) to determine any substrate relationships to vegetation composition and production.

Line-intercept sampling was used to measure the distance along each transect intercepted by shrub species. Any part of the shrub canopy that intersected the tape was recorded if it intersected more than 10 cm of the tape. Similarly, gaps in shrub canopy were not recorded unless they were larger than 10 cm.

Herbaceous biomass was collected in the spring by clipping and weighing herbaceous vegetation in three representative 1/16th-meter^2 quadrats adjacent to but outside of each plot. Vegetation was clipped as close to the ground as possible, and the field (wet) weight was recorded. Biomass samples were dried in a drying oven for 72 hours at 65º C to remove all moisture, then re-weighed for the dry weight.

Fisher study plots (Rochester et al. 2010) included point-intercept, relevé, and biomass vegetation sampling. These study plots were originally 25-meter radius circles. We identified the center points of those plots with GPS, maps, and plot photos. Using a compass, a 50-meter transect tape was positioned 25-meters north of the center point and 25-meters south. The 50-m transect served as a line-intercept transect and was sampled every 0.5 meters starting at 0.5 meters as described above (100 samples total). This is the same methodology used in the original surveys by Robert Fisher's lab (Hathaway et al. 2002). The soil substrate type was also recorded (e.g., bare rock, cobblestone, cryptogamic, leaflitter, moss/lichen, organic dirt, sandy soil). We measured species richness in these plots by recording all species within a 2-meter belt transect oriented along the east side of the 50-meter line-intercept transect. Herbaceous biomass was also collected in the Fisher plots as described above.

Fall RDM sampling and mapping

Residual Dry Matter (RDM) sampling and mapping was conducted in the fall seasons of 2021, 2022, and 2023. At each plot we recorded three co-dominant plant species and any site disturbance (i.e., presence of squirrel activity, heavy cattle grazing, and signs of erosion). One RDM sample was clipped per plot by placing a 0.96 foot^2 hoop in an area representative of herbaceous vegetation cover in the study plot. We removed any tree or shrub leaves, sticks, dirt, roots, and summer annuals out of the hoop. The remaining plant material was clipped as close to the ground as possible, then bagged and weighed with a scale. Clipped materials were dried in a drying oven for 24-48 hours at 65º C and weighed for the dried mass.

In 2022 and 2023, we also created a map of RDM condition classes across grazed areas of the reserves. These maps recorded which areas fell into one of four RDM categories: 1) 0 – 600 lbs./acre, 2) 600 – 1300 lbs./acre, 3) 1500 – 3000 lbs./acre, and 4) >3000 lbs./acre. These categories reflect both the RDM standards for the area and herbaceous biomass thresholds thought to correspond to wildfire behavior (Hulme Foss 2023). The minimum mapping unit for mapping was 2.5 acres, meaning unique RDM conditions occurring in smaller areas were not included in the map. We did note if a map polygon included significant areas above or below the condition class for that polygon that were too small to be mapped separately.

In Fall 2022, we added additional methods to assess browsing and shrub recruitment. Within each plot, the number of shrubs was recorded by species and by age class (i.e. seedling, young, mature; Coulloudon et al. 1999). If branch leaders showed evidence of browsing in the past year, the percent of total leaders browsed was recorded for each individual shrub. Lastly, the proportion of shrub species and age classes that fell into each form class was recorded (Coulloudon et al. 1999).

Other Field Sampling Methods

Obstruction Height. Obstruction height is a combined measure of the density and height of herbaceous vegetation, is obtained by placing a Robel pole in the center of the plot (or in a grassland-portion of the plot if shrubs are present), standing back 20 feet and looking at the pole from the height of the herbaceous vegetation and finding where the vegetation obscures over 80% of the black and white squares of the pole. This is repeated so that there is a value for north, south, east, and west directions. Herbaceous obstruction height was recorded during both spring and fall monitoring.

Photo Monitoring. All plots visited in spring and fall sampling had photographs taken at the time of sampling. In the spring, these photographs were taken from each side of the relevé, showing the plot in the foreground. Additional photos were taken of notable species and conditions (e.g., unusual disturbance and unknown plants). In the fall, similar photos were taken in four cardinal directions. These photos were taken around the northwest plot stake (rather than the 10x10-meter plot). Photographs of RDM clipping locations were also taken in the fall.

Remote Sensing and Imagery Analysis

Solar radiation. Total annual solar radiation at each of the study plots was calculated with ArcGIS. A 10-meter circular buffer was created around each plot point. The San Diego County Digital Elevation Model (DEM) raster was clipped to plot buffers to improve processing speed. Using the ArcGIS tool “Area Solar Radiation,” the annual solar radiation was calculated for each buffer circle. Using the “Zonal Statistics” tool, each polygon was summarized into one numerical value for solar radiation in watt hours per buffer circle. Dividing by the area of the buffer circle, each plot was assigned an annual solar radiation value in watt hours per square meter.

Historically cultivated areas. Using historical aerial imagery downloaded from Earth Explorer for the years: 1928, 1956, 1971, 1982, 1983, 1989, 1996, and 2000, we identified areas in the reserves that very likely had soil disturbance from activities such as construction or agriculture. Images that were downloaded as panels were stitched together using the Mosaic tool on ArcGIS. For each year, polygons were manually drawn over areas that appeared cultivated (e.g., had row crops, a smooth texture with linear borders, had infrastructure). The San Diego County DEM was clipped to project boundaries and helped in determining the cultivated polygon borders, as the smoothest parts of the terrain matched closely with hand-drawn polygons. Lastly, the polygons were checked by the land manager to assess their accuracy. With these sources combined, a final shapefile was created that best summarized all years into one hand-drawn polygon layer.

Data analysis

Data analyses fall into four categories: 1) Ecological Site Analysis, 2) Vegetation Classification, 3) Phytolith Analysis, and 4) Grazing Effects. In addition to these main analyses, there were several other supporting analyses that are reported below. Unless otherwise noted, all analyses were performed in the program R, version 4.3.1. The methods for these analyses are presented by category below.

Ecological Site analysis

Ecological sites were based on a classification of environmental variables at each of the study plots. The purpose of the ecological site classification is to try to find "types" of plots that have similar environmental characteristics and are therefore likely to have similar potential vegetation states, vegetation dynamics, ecosystem processes, and responses to management. One challenge is that it is difficult to know which environmental variables to include in the data used in the classification. To address this, we selected variables using Random Forest analysis based on whether they were important predictors of individual species occurrence on the 54 study plots.

Random Forest analysis was performed in R using the 'randomForest' package. The analysis was performed separately for each of the 138 species occurring on the 54 plots in the relevé dataset as response variables. The predictor variables in the Random Forest model were a set of soil nutrient, soil texture, geomorphology, and land use history variables. The model was run 1000 times to account for different results in each model run resulting from random sub-setting of the dataset performed by the Random Forest algorithm. Predictor variables that were deemed important for a high number of species were selected to be included in the ecological site classification.

The ecological site classification was based on a cluster analysis performed on plot data, using the values of the environmental variables selected by the Random Forest analysis as the basis for clustering. The cluster analysis determined which of the study plots had the most similar overall environmental factors. A combination of a Mantel test (Borcard et al. 2011), Indicator Species Analysis (Dufrene and Legendre 1997), and visual inspection of the cluster dendrogram were used to "prune" the resulting dendrogram to the optimal number of clusters.

Vegetation classification (defining vegetation states)

Vegetation classification was based on a cluster analysis performed on plot data, using the vegetation data in each plot (sampled in each year) as the basis for clustering. The vegetation data used for this analysis was the presence/absence data from the 10x10-meter relevé. We opted to use this binomial data for several reasons: 1) using percent cover data (instead of binomial) often over-emphasizes the importance of common species, 2) it is difficult to find an objective reason for specific data transformations to increase the importance of less-common species, 3) line-point sampling often misses small-statured or uncommon species that may be important indicators of different vegetation states, 4) high levels of presence for a given species in a class often is correlated to high levels of percent cover for that species in that class.

The vegetation classification was performed on data from all plots together, and also individually on data from the groups of plots in each ecological site. The two approaches were performed so that they could be compared to one another. A combination of a Mantel Test (Borcard et al. 2011), Indicator Species Analysis (Dufrene and Legendre 1997) and visual inspection of the cluster dendrogram were used to "prune" the resulting dendrograms to the optimal number of clusters.

Phytolith analysis

Phytolith samples were analyzed by Dr. Rand Evett and followed the methods of Evett and Bartolome (2013). These data were analyzed for differences in proportional mass and counts of phytolith morphotypes between plots in each of the ecological sites. Means and standard errors of morphotype counts and masses were graphed and these metrics were tested for each ESD against known thresholds representative of prehistoric grass states using t-tests.

Grazing Effects Analysis
We analyzed the effect of grazing on vegetation structure and composition with several methods. We created graphs to show differences in vegetation attributes, including percent cover of functional groups, percent cover of individual species, obstruction height, bare ground, and RDM. In these comparisons, grazing was generally treated as a categorical (yes/no) variable and results from grazed and ungrazed plots were compared.

We used negative binomial models to examine the relationship between grazing and percent cover of functional groups and individual species. In these analyses, 'grazing' was the number of animal unit days per acre of grazing during the growing season preceding spring sampling in the pasture that each study plot occurred in. The response variable was the species or functional group cover on a given plot in a given year. Year and ecological site were also included as predictor variables. These preliminary analyses can identify significant relationships between grazing use and species or functional groups, however more sophisticated models which account for repeated measures sampling at a subset of the study plots (the 29 plots sampled in 2022 and 2023) will be used in future analyses to better represent error in the sample. Similarly, cause and effect is difficult to determine since grazing use was not randomly applied to the study plots in a formal experiment.

Appendix C. Phytolith Methods and Results

Phytoliths are microscopic silica deposits plants develop in cells. Grasses form and deposit abundant phytoliths, which they leave behind in soils (Evett et al. 2006). They are unique to taxa, although often hard to distinguish from soil samples. A high abundance of phytoliths in soils can be an indicator of prehistoric grassland vegetation states (Evett and Bartolome 2013). Short cell phytoliths are a morphotype formed by grasses and of these, bilobate or "dumbbell" shaped phytoliths are characteristic of the California genera *Stipa (Nassella*), *Melica*, *Danthonia*, and other *Panicoid* grasses. Characteristic of the Mediterranean annual grass species that are the typical non-native grasses in California, are rondel (trapezoidal) and crenate (scalloped edges) phytoliths (Evett and Bartolome 2013). Evett and colleagues (2006) suggest that 0.3% phytolith dry weight in soils is a useful threshold to determine prehistoric grass dominance at a site. Based on research in California, 0.3% phytolith dry weight equates to 200,000 short cell phytoliths per gram of soil (Evett and Bartolome 2013). Research suggests that a bilobate/total phytolith ratio exceeding a threshold of 0.10 indicates a grassland dominated by the previously listed native species genera while 0.10 bilobates/total phytolith equates to 20,000 bilobates per gram of soil (Evett et al. 2006). Given annual rates of phytolith production and bioturbation in California, phytolith signatures in soil are indicative of conditions over the past 1,000 years or more.

Appendix C. Table 1. Phytolith results for each study plot.

Site	Soil Phytoliths (%)	Rondels	Bilobates	Crenates	Saddles	Total Short cells	Elongates	Appendages	Total Phytoliths	Diatoms	Short cells/ Elongates	Bilobates /Total
F1	0.57	71	65	65	0	201	195	18	414	6	1.03	0.16
F10	0.22	83	35	71	0	189	284	24	497	118	0.67	0.07
F11	1.98	755	552	639	29	1975	2149	116	4240	290	0.92	0.13
F12	0.35	247	75	67	15	404	441	22	868	7	0.92	0.09
F13	1.21	364	461	378	0	1202	1147	56	2405	112	1.05	0.19
F14	1.12	293	86	207	0	585	654	34	1273	86	0.89	0.07
F15	1.92	637	620	585	0	1841	1686	34	3562	155	1.09	0.17
F16	1.13	308	405	392	0	1105	1147	56	2307	98	0.96	0.18
F19	0.10	64	18	64	9	155	201	27	384	41	0.77	0.05
F2	0.78	381	77	136	13	607	516	71	1194	19	1.18	0.06
F21	0.55	250	177	137	8	573	476	24	1073	16	1.20	0.17
F3	0.23	132	48	91	3	274	220	16	511	11	1.24	0.09
F4	0.82	331	429	316	8	1084	753	23	1860	30	1.44	0.23
F6	0.42	166	120	75	0	361	497	30	888	105	0.73	0.14
F7	0.85	350	436	312	0	1097	769	27	1893	11	1.43	0.23
F9	0.20	77	47	53	0	177	142	12	331	12	1.25	0.14
HC1	0.25	26	32	69	0	127	191	17	335	79	0.67	0.09
HC2	0.23	26	49	30	0	105	139	11	256	23	0.76	0.19
HC3	0.17	55	13	33	0	100	153	11	264	15	0.66	0.05
HC4	0.38	107	70	86	0	263	267	33	563	25	0.98	0.12
HC5	0.11	34	21	36	0	91	130	7	228	69	0.70	0.09
HC6	0.48	118	98	67	5	288	453	46	788	62	0.64	0.12
HC7	0.11	34	65	33	0	132	237	0	369	68	0.56	0.18
N10	0.43	280	226	169	0	675	532	38	1245	8	1.27	0.18
N12	0.26	112	18	59	0	189	308	0	497	12	0.62	0.04
N2	1.38	565	312	536	30	1443	1874	30	3347	104	0.77	0.09
N3	0.79	243	448	269	0	959	1637	64	2661	294	0.59	0.17
N9	0.30	239	129	119	0	487	381	39	907	13	1.28	0.14

Site	Soil Phytoliths (%)	Rondels	Bilobates	Crenates	Saddles	Total Short cells	Elongates	Appendages	Total Phytoliths	Diatoms	Short cells/ Elongates	Bilobates /Total
RJ1	0.05	17	0	0	0	17	93	0	110	31	0.18	0.00
RJ10	4.68	705	684	725	0	2114	1711	40	3865	43	1.24	0.18
RJ11	3.59	535	529	430	0	1494	1439	83	3015	165	1.04	0.18
RJ12	0.31	64	25	59	0	148	162	26	336	15	0.91	0.07
RJ13	0.50	97	21	89	8	214	306	24	544	22	0.70	0.04
RJ14	1.64	280	310	499	0	1090	779	83	1952	106	1.40	0.16
RJ15	0.11	34	31	9	0	74	128	26	227	36	0.58	0.14
RJ16	1.17	265	653	530	47	1495	1315	66	2877	132	1.14	0.23
RJ17	0.65	294	364	280	0	938	700	28	1666	70	1.34	0.22
RJ18	0.32	93	52	72	0	217	258	57	531	93	0.84	0.10
RJ19	2.14	395	349	230	9	984	1149	156	2289	303	0.86	0.15
RJ2	0.80	456	58	167	0	680	725	61	1467	45	0.94	0.04
RJ20	0.76	246	142	159	0	547	579	38	1164	22	0.94	0.12
RJ21	0.08	25	0	51	0	76	304	0	380	152	0.25	0.00
RJ22	0.16	31	21	15	0	67	133	20	220	31	0.50	0.10
RJ3	0.74	246	56	175	0	477	604	71	1152	58	0.79	0.05
RJ4	0.51	152	66	113	0	331	444	20	795	7	0.75	0.08
RJ5	1.51	443	267	325	0	1035	677	46	1757	26	1.53	0.15
RJ6	1.01	262	253	163	0	678	572	16	1267	41	1.19	0.20
RJ7	3.82	666	468	616	25	1775	1972	49	3797	99	0.90	0.12
RJ8	4.98	620	413	574	23	1630	1791	69	3491	597	0.91	0.12
RJ9	3.43	648	759	641	0	2048	1897	190	4135	63	1.08	0.18
RJOtay	0.14	132	13	61	0	206	215	18	439	66	0.96	0.03
UG1	0.91	267	146	224	0	637	490	52	1179	0	1.30	0.12
UG2	0.17	108	43	86	4	241	237	26	503	4	1.02	0.09
UG3	0.10	62	38	70	3	172	226	5	403	3	0.76	0.09