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Impacts of livestock grazing on the probability of burning in wildfires vary by region and vegetation type in California

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ABSTRACT

Wildfire activity has recently increased in California, impacting ecosystems and human well-being. California's rangelands are complex social-ecological systems composed of multiple ecosystems and the people who live and work in them. Livestock grazing has been proposed as a tool for reducing wildfire activity. Here, we explore how grazing affects wildfire at large spatial scales, assessing burn probability on rangelands with different grazing levels. We collected grazing data by surveying 140 large private landowners in three social-ecological regions: California's North Bay, Central Coast, and Central Valley and Foothills. Using pre-regression matching and mixed effects regression, we calculate the burn probability from 2001 to 2017 in points sampled from grazed and ungrazed properties in each region in grasslands, shrub/scrublands, and forests. We find that in the Central Coast and North Bay, annual burn probability decreases as stocking levels increase across all vegetation types, with reductions of 0.008–0.036. In the Central Valley and Foothills, the relationship is complex, with burn probability increasing over some grazing levels and variations in the effect of higher stocking densities. Our results indicate that livestock grazing may reduce annual burn probability in some regions and ecosystems in California, providing the first large-scale assessment of this relationship.

1. Introduction

Across the western United States, wildfire activity has increased over the past four decades, impacting ecosystems and human lives (Parks and Abatzoglou, 2020). In California, the annual area burned has increased four-fold since 1972, driven by interactions between increased anthropogenic ignitions and human development in wildland areas, climate change, and the legacy of fire suppression (Balch et al., 2017; Radeloff et al., 2018; Stephens and Ruth, 2005; Williams et al., 2019). While much of this increase in burned area has occurred in forests, large areas of California's rangeland grasslands and shrublands have also burned (Calhoun et al., 2021). For example, the 2017 Tubbs and Thomas fires collectively burned over 150,000 acres of shrubland and 55,000 acres of grassland (over 64% of the total area burned in the two fires). In recent decades, California's rangelands have also experienced rapid human population growth, with a corresponding increase in the amount of wildland-urban interface (Radeloff et al., 2018). Global trends reflect this pattern, as 45.8% of temperate grasslands, savannas, and shrublands have been converted for human use, along with 41.4% of Mediterranean forests, woodland, and scrublands (Hoekstra et al., 2005).

California's rangelands represent complex social-ecological systems, with ranches, residential developments, and other human land uses interacting with diverse, fire-adapted ecosystems including grasslands, chaparral and other shrublands, and woodlands (Cameron et al., 2014). Historically, these landscapes burned periodically in lightning-ignited fires, and Indigenous peoples used fire to manage the distribution and

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abundance of resources (Anderson, 2005; Stephens et al., 2007). Nineteenth century colonial policies prohibited Indigenous fire use, followed by widespread, intensive livestock grazing and other land use changes (Taylor et al., 2016). In the early twentieth century, California began practicing aggressive fire suppression, seeking to reduce loss of human lives and property (Stephens and Sugihara, 2006). As a result of interactions between land management policies, land use change, and climate change, dry fuels have accumulated across California's landscapes, setting the stage for large and severe wildfires (Steel et al., 2015). These altered fire regimes can disrupt ecological processes and force type conversions in vegetation communities such as forests in the Sierra Nevada mountains (Coop et al., 2020) and southern California's coastal chaparral (Syphard et al., 2019).

In recent years, there have been calls in the academic literature and popular media to use livestock grazing as a tool to mitigate the risk of large, high-severity fires, both in the American West and elsewhere (Davies et al., 2015; Nelson, 2019; Williams et al., 2006). Grazing reduces the accumulation and connectivity of fuels, particularly fine (herbaceous) fuels: livestock directly consume potential fuels and trample vegetation, reducing fine fuels' flammability by mixing them with mineral soil while rearranging the their spatial structure (Nader et al., 2007; Tsiouvaras et al., 1989).

Small-scale experimental, observational, and modeling studies have shown that by reducing fuel accumulation and connectivity, grazing may reduce flame length, rate of spread, and fire intensity and severity (Diamond et al., 2009; Launchbaugh et al., 2008; Leonard et al., 2010; Starns et al., 2019). Grazing and fire may also interact to create spatial heterogeneity in vegetation structure and composition (Fuhlendorf et al., 2009). In tallgrass prairies in the Great Plains, ungulates preferentially graze recently-burned areas due to the higher nutritional value of regrowth forage. They thus create a heterogeneous landscape where recently-burned patches have low fuel accumulation due to grazing, while patches without recent fire have greater fuel accumulation, with implications for fire spread (Allred et al., 2011). This pattern is less clear but still present in arid shortgrass steppe (Augustine and Derner, 2014). Meanwhile, a study in southeastern Australia found no difference in burn probability between grazed and ungrazed grasslands and shrublands (Williams et al., 2006). Overall, findings relating grazing to subsequent fires vary depending on vegetation type and the timing of grazing relative to both plant phenology and fire seasons. Studies of this interaction in California's ecosystems are limited, and questions remain about the relationship between grazing and wildfire at broad spatial scales (Keeley et al., 2011).

While satellite imagery provides widely available information on trends in the extent and location of rangeland ecosystems (land cover), data on grazing levels (land use) are limited. The lack of spatially-explicit livestock grazing data across broad areas is a major barrier to research on landscape-scale relationships between grazing and wildfire. Spatial data on rangelands as a land cover do not typically include information on whether or not grazing is occurring, let alone stocking rates, which are key to grazing-fire interactions. As the impact of wildfire on rangeland social-ecological systems is mediated by land management decisions, a more complete understanding of the grazing-fire relationship requires data on land cover, land use, and grazing intensity.

Here, we assess the effect of grazing on burn probability in California's rangelands, using points sampled from grazed and ungrazed rangeland properties. Combining a time series of fires from 2001 to 2017, grazing data from ranches across seven counties, and environmental and socioeconomic covariates, we use pre-regression matching and logistic mixed effects models to analyze whether 1) livestock grazing impacts burn probability in California, and 2) whether the effect of livestock grazing on burn probability varies by region and dominant vegetation type. Our application of econometric methods to assess the impact of land management on burn probability is relatively novel (but see Starrs et al., 2018). Because we use pre-regression matching to control for systematic differences in covariates that may be related to both burn probability and the presence of grazing (*e.g.*, net primary productivity, topographic variables, etc.), we are able to better understand the effect of grazing on burn probability.

2. Methods

2.1. Study area

Our study area comprises three social-ecological regions in California, defined by environmental and administrative boundaries (Bailey ecoprovinces (Bailey, 1995) and county borders, respectively): the Central Coast, Central Valley and Foothills, and the North Bay (Fig. 1). Cattle production practices are largely similar across the three regions.

The Central Coast region contains the California Coastal Chaparral Forest and Shrub ("Coastal Chaparral"; Fig. 1) and California Coastal Range Open Woodland-Shrub-Coniferous Forest-Meadow ("Coastal Range") Provinces in Santa Barbara and Ventura counties. Historically, this region had frequent, small lightning-ignited fires and larger, wind-driven fires every 50–100 years (Keeley and Fotheringham, 2001). Before European colonization, Indigenous Californians used fire to maintain grasslands and other resource-abundant vegetation communities (Keeley, 2002). Large fire frequency has increased in recent years as the human population has grown and ignitions coincide with Santa Ana winds (Keeley and Zedler, 2009). The number of ranching operations and cattle declined from the 1960s to 2017 (Andersen et al., 2002; USDA-NASS, 2017, 2007).

The Central Valley and Foothills region, represented by San Joaquin, Merced, and Mariposa counties, is almost entirely in the California Dry Steppe Province, as well as the low-elevation western Sierran Steppe-Mixed Forest-Coniferous Forest-Alpine Meadow Province ("Sierran Steppe") and the eastern Coastal Range. Land conversion for intensive human uses has transformed this region (Cameron et al., 2014); by 2001, cultivated cropland covered nearly 37% of the land area in this region (compared to 5% in the Central Coast and the North Bay) (USGS, 2014). Little is known about the pre-colonization fire regime of the Central

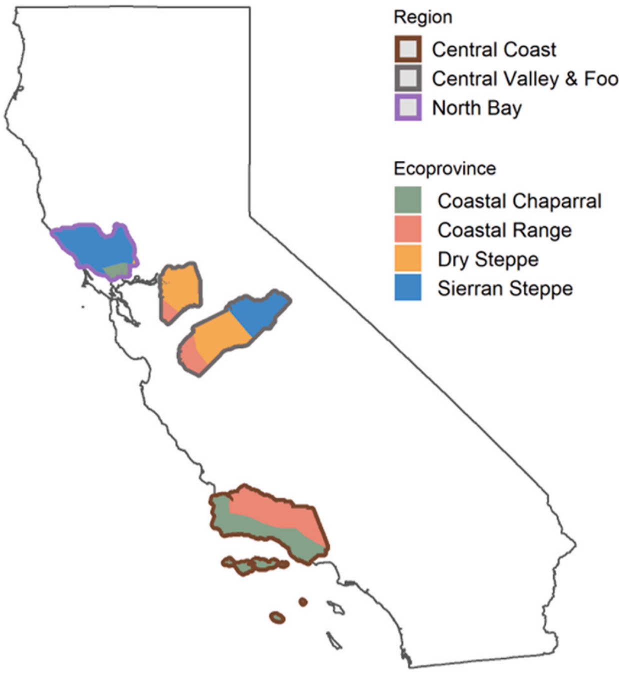

Fig. 1. Map of the three social-ecological regions of California studied, with their component ecoprovinces denoted by color. The regions were defined by Bailey ecoprovinces and county boundaries. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

Valley (Willis, 2018). In the latter half of the twentieth century, cattle numbers increased, with a relatively stable number of ranching operations and cattle since 2002 (Andersen et al., 2002; USDA-NASS, 2017, 2007).

The North Bay region includes Napa County and most of Sonoma County; the region falls within the southwestern Sierran Steppe ecoprovince and the northernmost portion of the Coastal Chaparral ecoprovince. The chaparral ecosystems in the North Bay are resilient to fire and historically had high-severity, stand-replacing fires every 30–100 years (Stephens et al., 2007). North Bay woodlands have experienced decades of fire suppression, leading to a greater fuel density, more vertical fuel connectivity, and increased risk of crown fires (Stephens et al., 2018). Since 2002, the number of cattle in the North Bay has remained fairly stable (USDA-NASS, 2017, 2007).

2.2. Data

We collected grazing intensity data through a telephone survey of large private landowners in our study regions. We randomly sampled private landowners with properties of at least 2 km^2 (500 acres), of which at least 1 km^2 were grassland or shrub/scrubland, until we obtained 20 responses per county. We asked each landowner whether they were actively grazing their land. For grazed properties, we then collected data on the number of animals grazed, months of active grazing, and acres grazed. The data represented long-term grazing levels, as property owners reported relatively static trends in land use intensity. For this reason, we used the same level of grazing intensity for each property across the entire time series (2001–2017). We then calculated the animal units per year (AUY) per acre grazed, a measure of grazing intensity. We collected data from 140 properties, 123 of which provided sufficient data for analysis. With few exceptions, these properties were not adjacent to each other.

Within the surveyed properties, we sampled points along a 200-m grid, using land cover data from the 2001 National Land Cover Database to restrict our samples to grasslands, shrub/scrublands, and forests. We did not sample land covers unlikely to experience grazing (*e.g.*, water, developed areas) (USGS, 2014). As our sample included very few irrigated pastures, distributed across land cover types, we do not believe that irrigation affected our results (Appendix 1). Because we observed minimal change in the dominant land cover types in the properties surveyed from 2001 to 2016 (USGS, 2019), we assumed constant land cover classes during our study period (Appendix 2). The 200 m sample combined with the non-adjacent nature of our properties reduces the chance for unobserved spatial autocorrelation in our model.

For all sample points, we compiled fire history data. We included all fires from 1996 to 2017 that overlapped with our sample points, using the California Department of Forestry and Fire Protection's Fire and Resource Assessment Program database (CAL FIRE, 2020), the most complete dataset available for California fire perimeters. We determined whether each point burned in wildfires in each year (2001–2017). To account for the legacy effects of past fires, we also determined whether each point had burned in previous years (Price et al., 2015).

For each sample point, we assembled climatic, biophysical, and human variables related to forage/fuel quantities and conditions, potential ignitions, fire behavior, and agencies' responses to fires: total precipitation from the previous water year and seasonal precipitation; mean seasonal maximum wind speed, maximum and minimum temperatures, soil moisture, and Palmer Drought Severity Index (PDSI) (Abatzoglou et al., 2018); annual net primary productivity (NPP) (Running and Zhao, 2019); population density in 2000 and 2010 (Radeloff et al., 2018); distance to roads (US Census Bureau, 2018); and elevation, slope, and solar radiation aspect index (USGS, 2013). We performed data compilation and calculations in R, using the *raster*, *sf*, and *lwgeom* packages (Hijmans, 2019; Pebesma, 2018, 2019). Appendix 3 provides for further details on data compilation.

2.3. Matching

There may be fundamental geographic, climatic, or environmental differences between sample points on grazed and ungrazed properties. For example, property owners with steep terrain with low NPP might be less likely to graze than property owners with gentle terrain and abundant forage. Differences between grazed and ungrazed properties may also impact their burn probability, through differences in fuel accumulation, ignition probability, and fire spread. To control for these potential differences and improve our ability to understand causal relationships, we used pre-regression nearest neighbor propensity score matching (Stuart, 2010). For each grazed sample point in our dataset, we identified the ungrazed sample point with the closest propensity score, then removed all unmatched grazed and ungrazed sites. Using the *MatchIt* package (Ho et al., 2011), with a caliper of 0.25 and a maximum ratio of grazed to ungrazed samples of 5:1, we matched along the climatic, biophysical, and human covariates described above, as well as latitude and longitude, which we included as linear variables. For time-varying variables (*e.g.*, climatic variables, NPP), we used mean values over the first five years of the study period (2001–2005) for the matching process. Grazed and ungrazed points were thus matched on their mean covariate values from the beginning of the time series. Because our matching ratio was 5:1, each ungrazed sample point could have up to 5 grazed matching points; this allowed us to expand our dataset despite the fact that we had more grazed sample points than ungrazed. We assessed match quality by comparing the standardized mean differences in variable values for grazed and ungrazed points in the matched and unmatched datasets. We removed unmatched points from our dataset. We developed two matched datasets, one with sample points on only grasslands and shrub/scrublands (Table A1) – the vegetation types where grazing occurs – and a second on points in grasslands, shrub/scrublands, or forests (Table A2). We used this second dataset to explore the potential impact of grazing in grassland and shrub/scrublands on the probability of wildfires spreading into forested land within the properties studied.

Of the 123 properties with grazing intensity data, we omitted five properties located in the California Coastal Steppe-Mixed Forest-Redwood Forest Province of western Sonoma County because its fire regime and climate are very different from those of the other regions in our study (Stephens et al., 2018). The remaining 118 properties cover 301,649 acres (1220.73 km^2) of land across the three regions (Table 1). Both matched datasets included sample points from all 118 properties (78 grazed and 40 ungrazed). The grassland and shrub/scrublands matching yielded 12,184 grazed and 5020 ungrazed sample points (out of an initial dataset of 17,171 grazed and 5020 ungrazed points); the matching process for grassland, shrub/scrublands, and forested points yielded a dataset with 13,252 grazed and 3992 ungrazed points (out of an initial dataset of 19,104 grazed and 6276 ungrazed sample points). The standardized mean differences between the covariate values for the grazed and ungrazed points in both datasets were sufficiently low, indicating effective reduction of bias (Appendix 4, Table A4a, Table A4b). Grazed properties' grazing levels ranged from 0.012 to 0.424 AUY per grazed acre. To assess whether our results were robust to the pre-regression matching process, we also ran our model for the full, unmatched dataset of grassland, shrubland, and forested points (Appendix 5).

2.4. Models

We used logistic mixed effects models with cluster-robust standard errors at the property-year level to estimate the effect of grazing on burn probability, where the response variable was whether or not a given sample point burned in each year in the time series. The mixed effects models captured the effects of unobserved factors that may influence burn probability, such as land use legacies or fine-scale environmental factors affecting the production of forage/fuel. The cluster-robust standard errors controlled for pseudoreplication and unobserved variables

Table 1

Summary of the sizes, fire histories, land cover types, and select covariates of the properties in the matched dataset. Where mean values are reported, standard deviations are presented in parentheses.

	Central Coast		Central Valley & Foothills		North Bay	
	Ungrazed	Grazed	Ungrazed	Grazed	Ungrazed	Grazed
Number of properties	22	26	9	39	9	13
Mean property size (km^2)	6.61 (7.10)	12.81 (24.33)	25.36 (27.79)	9.94 (8.82)	4.58 (3.08)	6.55 (3.44)
Mean annual area burned, 2001–2017 (km^2)	3.56 (7.77)	9.28 (19.42)	0.56 (1.29)	2.66 (6.15)	0.96 (3.97)	2.10 (5.69)
Forest area (km^2)	17.85	24.35	9.59	19.01	18.88	22.76
Grassland area (km^2)	8.71	71.04	83.19	306.13	3.83	19.12
Shrub/scrubland area (km^2)	98.16	209.33	10.33	32.09	17.86	41.33
Mean 2000 population density (people/km^2)	7.1 (39.9)	1.1 (14.2)	1.0 (6.4)	0.7 (2.3)	2.8 (14.7)	2.4 (5.9)
Mean annual precipitation, 2001–2017 (cm)	38.5 (17.4)	40.1 (17.1)	39.9 (18.9)	45.2 (20.1)	95.0 (28.0)	81.8 (26.0)
Mean NPP, 2001–2017 (kg*C/m^2)	0.66 (0.24)	0.64 (0.30)	0.45 (0.11)	0.44 (0.10)	0.99 (0.17)	0.79 (0.18)

on individual properties (Abadie et al., 2017; Cameron and Miller, 2015).

We fit a logistic regression model to the grasslands and shrub/scrublands matched dataset, dropping highly correlated explanatory variables ($|\text{Pearson's correlation coefficient}| \geq 0.66$, p-value < 0.05):

$$BN_{it} = B_0 + B_1{*}Grazed_i + B_2{*}YearFactor_t + B_3{*}AUYperGrazedAcre_i + B_4Region_i + B_5{*}Landcover_i + B_6{*}BPYone_{it} + B_7{*}BPYfive_{it} + B_{8\text{-}14}{*}StaticControls_i + B_{15\text{-}26}{*}DynamicControls_{it} + B_{27\text{-}31}{*}Interactions_i + u_i + e_{it}$$

where *BN* represented whether or not point *i* burned in year *t*; *Grazed* was a binary variable indicating whether the point was in a grazed or ungrazed property, allowing us to capture unobserved differences between grazed and ungrazed properties for which the matching process did not control; *Year* was an annual fixed effect (2001–2017); *AUYperGrazedAcre* represented the animal units per year per grazed acre for the property; *Region* was a fixed effect for the three social-ecological regions; *Landcover* was the dominant vegetation type present (grassland or shrub/scrub); *BPYone* and *BPYfive* were lag variables for whether the point burned in the previous one or five years, respectively; *StaticControls* were the socio-environmental covariates that may influence burn probability and did not vary over time (population density in 2000, distance to roads, elevation, slope, aspect, latitude and longitude); and *DynamicControls* were the time-varying climatic covariates (see Table A7 for the list of variables). *Interactions* included all interactions between *AUYperGrazedAcre*, *Region*, and *Landcover* to test for different responses to grazing across regions and vegetation types. We interacted latitude and longitude to control for spatial autocorrelation (Schleicher et al., 2017). u_i is the site-specific random effect and e_{it} is the error term for each point in each year. To interpret the interactions between region, land cover type, and grazing levels, we calculated the predicted burn probability across the range of grazing levels, along with 95% confidence intervals and p-values, in Stata (StataCorp, 2019).

To assess whether grazing in grasslands and shrub/scrublands had an impact on burn probability in forests, we used our second matched dataset to perform a logistic regression model using the same equation. Here, the AUY per grazed acre was still a function of the number of grassland and shrub/scrubland acres on the property, as we assumed minimal grazing occurred in forests. To further assess the potential impact of spatial autocorrelation, we subset the second matched dataset such that all points were at least 400 m distant from each other and performed the same logistic regression model (Appendix 6).

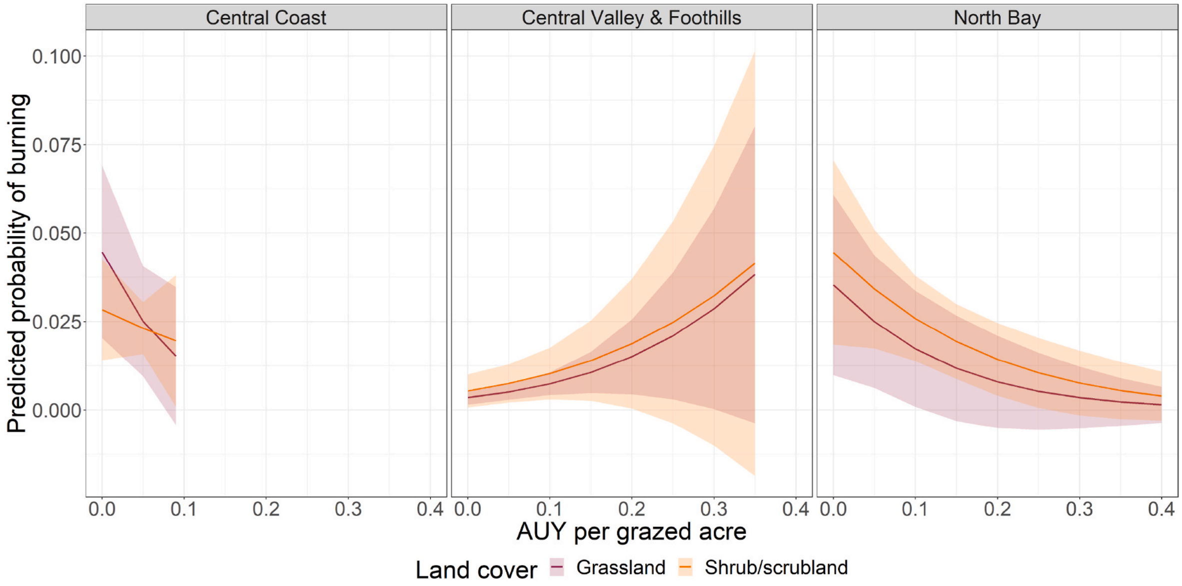

Fig. 2. The predicted probabilities of burning in the three regions based on data using only the two dominant vegetation types using for grazing (grassland and shrub/scrubland). The model is used to estimate changes in predicted probabilities of burning as AUY per grazed acre increases from 0 (ungrazed). The annual predicted probability values represent the likelihood of a given sample point in each region and vegetation type burning, across the gradient of grazing levels. The graphs extend across the range of AUY per grazed acre values observed in each region: the maximum grazing levels in each region are 0.090 AUYs per grazed acre (11.111 acres per AUY) in the Central Coast, 0.357 AUYs per grazed acre (2.800 acres per AUY) in the Central Valley and Foothills, 0.424 AUYs per grazed acre (2.357 acres per AUY) in the North Bay. The shaded regions represent the 95% confidence intervals.

3. Results

In the Central Coast, the annual predicted probability of burning decreased as grazing levels increased in the grassland and shrub/scrublands dataset, even though the maximum level of grazing intensity was relatively low (Fig. 2). In shrub/scrublands, this effect held across observed levels of grazing, with a decrease in burn probability from 0.028 ($p < 0.001$) to 0.020 ($p < 0.001$) as grazing increased from no grazing to 0.09 AUYs per grazed acre. In grasslands, burn probability decreased from 0.045 ($p < 0.001$) to 0.025 ($p = 0.002$) as grazing levels increased to 0.05 AUYs per grazed acre, without a significant continued decrease at higher grazing levels.

Similarly, in the North Bay, burn probability declined as grazing intensity increased: in grasslands, burn probability decreased from 0.035 ($p = 0.007$) to 0.017 ($p = 0.039$) as grazing levels increased to 0.10 AUY per grazed acre. In shrub/scrublands, burn probability decreased from 0.045 ($p = 0.001$) to 0.010 ($p = 0.037$) as grazing levels increased to 0.25 AUY per grazed acre (Fig. 2). At maximum grazing levels, the burn probabilities of grasslands in the North Bay and the Central Coast were comparable (0.015 vs. 0.017), even though grazing reached greater intensity levels in the North Bay.

In contrast to the other regions, grasslands and shrub/scrublands in the Central Valley and Foothills region showed an increase in burn probability when grazing levels increased from no grazing to 0.30 and 0.20 AUYs per grazed acre, for grasslands and shrub/scrublands, respectively (Fig. 2). From grazing levels of 0.20–0.30 AUY per grazed acres, grassland burn probability increased from 0.015 ($p = 0.005$) to 0.029 ($p = 0.048$), but there was considerable uncertainty around the predictions. Shrub/scrubland burn probability more than doubled as grazing levels increased to 0.20 AUY per grazed acre, reaching a similar value as Central Coast shrub/scrublands under the Central Coast's maximum grazing levels. At higher levels of grazing, there was no significant trend for either land cover type.

In the grassland and shrub/scrubland dataset, points on steeper slopes, with more winter, summer, and fall precipitation, less spring precipitation and less total rainfall in the previous year, higher summer wind speeds, and lower annual NPP were more likely to burn in a given year (Table 2). Sample points that had burned in the previous year were much less likely to burn. Several years in the time series had significant effects on burn probability (Table A7). The coefficient estimates for latitude, longitude, and their interaction had absolute values of less than 0.001, indicating that spatial autocorrelation had minimal impact on our model (Schleicher et al., 2017).

When we calculated the predicted probability of wildfire in the second matched dataset (grasslands, shrub/scrublands, and forests), we found that forests in the Central Coast and North Bay showed decreases in burn probability as grazing increased (Fig. 3). In the Central Coast, burn probability in forests declined from 0.075 ($p = 0.001$) to 0.039 ($p < 0.001$) as grazing levels increased from no grazing to 0.05 AUYs per grazed acre. In the North Bay, burn probability in forests decreased from 0.042 ($p < 0.001$) to 0.012 ($p < 0.001$) as grazing intensity increased from no livestock to 0.40 AUYs per grazed acre. Forests in the Central Valley and Foothills, on the other hand, had an increase in burn probability from 0.006 ($p = 0.033$) to 0.007 ($p = 0.033$) when grazing intensity increased from 0.05 AUYs per grazed acre to 0.10 AUYs per grazed acre, but there was no significant trend in burn probability at higher levels of grazing. In the second dataset, points that were farther from roads, on steeper slopes, and with more winter and fall precipitation were more likely to burn (Table 2). Points with higher annual NPP and that burned in the previous year were less likely to burn, and several years had positive effects on the probability of burning relative to the reference year (Table A7). The patterns observed for the matched grassland, shrub/scrubland, and forest dataset were the same as those that resulted when we used the full, unmatched dataset to model burn probability (Appendix 5).

Table 2

Coefficient estimates for the logistic regression models of burn probability in the matched dataset of grasslands and shrub/scrublands and the matched dataset of grasslands, shrub/scrublands, and forests. Cluster-robust standard errors are in parentheses below each coefficient estimate. The Central Coast serves as the reference region. For the datasets with grasslands and shrub/scrublands, grasslands are the reference land cover type. For datasets with grasslands, shrub/scrublands, and forests, forests are the reference. See Table A7 for complete table of coefficients, including the interactions terms and the effects of each year's fixed effect, latitude, longitude, and the covariates that were not significant in either model.

Variable	Coefficient estimates	
	Grassland and shrub/ scrubland dataset	Grassland, shrub/scrubland, and forest dataset
Intercept	149.826 (70.041)	142.643* (69.307)
Grazed	0.978* (0.378)	0.802* (0.381)
AUY per grazed acre	−14.470 (10.431)	−18.406 (11.361)
Central Valley & Foothills	7.907* (3.653)	8.198* (3.507)
North Bay	11.186* (5.177)	11.936* (4.868)
Grassland		−0.958* (0.407)
Shrub/scrub	−0.577* (0.245)	−1.434*** (0.315)
Distance to roads (m)	$1.506 * 10^{-4}$ (0.000)	$2.770*10^{-4}$* (0.000)
Slope (°)	0.033*** (0.007)	0.035*** (0.008)
Total precipitation, fall (cm)	0.263*** (0.059)	0.210*** (0.059)
Total precipitation, spring (cm)	−0.133* (0.052)	−0.095 (0.057)
Total precipitation, summer (cm)	0.824* (0.415)	0.615 (0.435)
Total precipitation, winter (cm)	0.181** (0.058)	0.117** (0.041)
Max wind speed, summer (m/s)	0.568* (1.248)	0.959 (1.168)
Previous year precipitation (cm)	−0.097* (0.047)	−0.049 (0.037)
NPP ($kg*C/m^2$)	−3.776*** (0.663)	−4.239*** (0.647)
Burned in previous year	−4.225*** (1.184)	−4.772*** (0.945)

*$p < 0.05$, **$p < 0.01$, ***$p < 0.001$.

4. Discussion

In California rangelands, the impact of livestock grazing on burn probability varies across regions and vegetation types. We find that an increase in grazing levels is related to reduced burn probability in the forests, grasslands, and shrub/scrublands in the North Bay and Central Coast regions (with Central Coast shrub/scrublands showing the smallest relative decline). The decreases in burn probability in these Central Coast ecosystems occur even with a small change in AUY per grazed acre (from no grazing to 0.05 AUYs per grazed acre). In contrast, the three land cover types in the Central Valley and Foothills region showed increased burn probability over some of the range of observed grazing intensities, but as grazing intensity increased, the trends were not significant.

The similar responses to increased grazing intensity in forests and grasslands in the North Bay and the Central Coast suggest that the relationship between grazing levels and burn probability may be generalizable across some ecological communities in California; in these systems, grazing may effectively manage fires by reducing fuel availability and/or connectivity. The decline in grassland burn probability as grazing increased was 0.020 in the Central Coast and 0.022 in the North Bay. Non-native annual species dominate grasslands in both regions (Keeley and Syphard, 2018; Stephens et al., 2018), perhaps accounting for the similar observed responses to grazing. Notably, we also observed a decline in burn probability in forested areas (where we assume minimal grazing occurs). In both regions, decades of fire suppression and spread of nonnative species have led to increased densities of flammable forest understories (Keeley and Syphard, 2018; Stephens et al., 2018). Livestock grazing may reduce burn probability by disrupting fuel

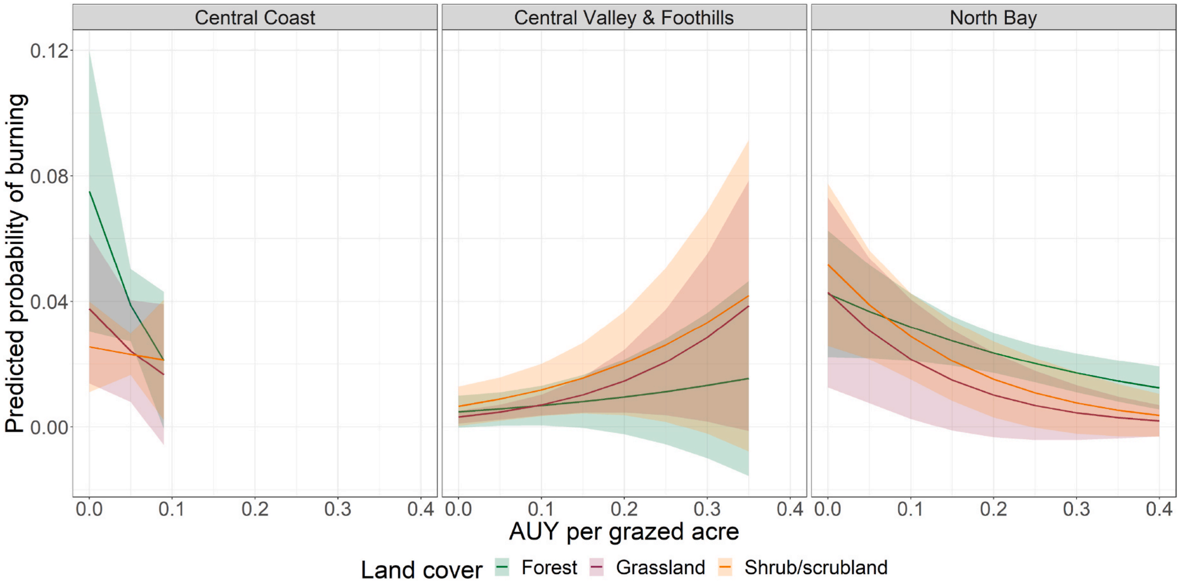

Fig. 3. The predicted probabilities of burning in the three regions with data from forest, grassland, and shrub/scrubland as AUY per grazed acre increases from 0. The graphs extend across the range of AUY per grazed acre values observed in each region. The shaded regions represent the 95% confidence intervals.

connectivity to adjacent grasslands and shrub/scrublands or directly reducing understory biomass accumulation. Policies that reduce barriers to grazing on private and public lands adjacent to forests in both these regions may reduce the probability of high-severity wildfires (Sulak and Huntsinger, 2007; Wolf et al., 2017).

The different strengths of the shrubland response to grazing across the North Bay and Central Coast may reflect different climatic conditions. In the Central Coast, autumn foëhn winds spread fires over large areas, potentially overcoming any effect of fuel quantity and connectivity on fire extent (Keeley and Fotheringham, 2001). While our model includes mean maximum fall wind speeds, there may be effects at finer spatial scales than our data could capture. The Central Coast sample points were also drier on average than the North Bay points, which may correspond to greater fuel aridity and a dampened effect of grazing.

It is more difficult to draw conclusions about the grazing-fire link in the Central Valley and Foothills. At lower stocking rates, the burn probability increases with grazing intensity, contradicting our hypotheses about the relationship between grazing levels and burn probability. When stocking rates are moderate or high, however, there is greater variation in wildfire responses. Our dataset contained few sample points from shrub/scrublands or forests in this region (Table 1), and overall it had a lower fire frequency in the matched dataset than the other two regions, complicating comparisons. Large portions of this region have been converted to intensive human land uses. Because some of the properties we surveyed are close to intensive agriculture, these landscapes likely experience altered fire regimes, possibly changing the grazing-fire relationship. In this highly-modified landscape, the location of rangelands and their proximity to either wildlands or intensive agriculture—factors that we did not control for—may be more important predictors of burn probability than grazing levels. The Central Valley and Foothills region also had lower mean human population density and net primary productivity than the other two regions (Table 1), which may contribute to the different patterns observed there.

The relative abundance of native and non-native grassland species in coastal California as compared to inland regions may also contribute to the different effect of grazing on wildfire in the Central Valley and Foothills. While non-native species dominate almost all California grasslands (Seabloom et al., 2003), the abundance of non-native annuals is higher in inland regions (Hatch et al., 1999; Rayburn et al., 2016). It is possible, then, that in the coastal regions where grazing reduced burn probability, livestock grazing more effectively decreases the amount of fine fuel composed of non-native, flammable annuals, without the same effect occurring in the more heavily-invaded Central Valley and Foothills (Keeley, 2001). Forests in the North Bay and Central Coast were generally intermixed with grasslands and shrub/scrublands in the properties we surveyed, while in the Central Valley and Foothills region, forests were located at higher elevations than grasslands and shrub/scrublands, in the foothills of the Southern Coastal Range and the Sierra Nevadas. At these higher elevations, the forested portions of grazed properties in the Central Valley and Foothills may be more isolated from the effects of grazing, thus dampening any potential effect of grazing on their burn probability. In socio-environmentally diverse regions such as the Central Valley and Foothills, our regional-scale of analysis may obscure finer-scale relationships between grazing and burn probability.

While our data on grazing levels, collected through surveys that reached ranchers directly, provided us with an unusually detailed breakdown of grazing intensity on private lands, we still made several key assumptions about the data. First, the theoretical link between grazing levels and burn probability is based on variation in fuel levels. We did not have data on end-of-season residual dry matter (RDM) quantities; we used AUY per grazed acre as a proxy, assuming that as the grazing intensity increased, the unconsumed forage (RDM) would decrease. Second, we could not account for the seasonality of grazing relative to the phenology of dominant plant species, which is relevant for RDM quantities (Diamond et al., 2009; Launchbaugh et al., 2008; Nader et al., 2007). Third, we assumed uniform grazing levels across all grassland and shrub/scrubland areas within each property. In reality, cattle preferentially graze close to water, along fences, and in recently burned areas, while avoiding steep slopes (Allred et al., 2011; Augustine and Derner, 2014), and some shrubland may be too dense for cattle. However, we did not have data on grazing intensity at spatial scales finer than the property-level. Our grazing data also represented long-term, relatively static levels of grazing and may not capture fine-scale variation in grazing intensity over space and time within each property. The impact of grazing on burn probability is also related to the livestock species used and the individual animals' previous grazing experiences (Nader et al., 2007). Our study does not account for this level of interaction, although the majority of the surveyed properties graze cattle exclusively. Finally, while the matching procedure used can help control for observable differences between grazed and ungrazed properties, it cannot control for other forms of endogeneity that may limit our analysis.

Interestingly, although our use of pre-regression matching reduced the standardized mean differences in the covariate values for grazed and ungrazed properties (Appendix 4), the results derived from the matched dataset did not differ meaningfully from those derived from using the full, unmatched dataset (Appendix 5). This indicates that while there were systematic differences between grazed and ungrazed properties in our study area, these differences did not affect the relationship between grazing levels and burn probability. While in this particular case, pre-matching regression may not be absolutely necessary to reduce biases stemming from systematic differences between the treatment groups, we cannot assume this finding would apply beyond our study area.

Along with its potential impacts on burn probability, livestock grazing can have undesirable ecological effects, particularly at higher stocking levels. These impacts include reductions in water quality, soil compaction, degradation of riparian ecosystems, weed transmission, and disease interactions with wildlife (Nader et al., 2007). Land managers must balance these tradeoffs, along with the vegetation community- and region-specific effects of grazing on burn probability, when deciding if and where to use grazing as a fire management tool. Targeted grazing, which focuses on patches that have not burned recently, may be particularly effective in reducing fuel availability (Diamond et al., 2009), potentially shifting the balance of ecological tradeoffs.

Interannual variation in the environmental factors controlling forage production poses a challenge for ranchers seeking to maximize the fuels reduction potential of grazing: the ideal stocking level for minimizing RDM likely varies annually (Bartolome et al., 2006). Variation in precipitation may lead ranchers to use conservative stocking levels that are insufficient for reducing fuel levels. This may hold especially true in the Central Valley and Foothills region, where weather variability drives vegetation dynamics due to the nonequilibrium dynamics of the system (Spiegal et al., 2016). Decisions about stocking levels can be further complicated under stocker operation agreements that limit flexibility in both livestock number and the dates of the grazing season. This highlights the uncertainties of land management in complex social-ecological systems like California's rangelands, where local ecology, varying climatic factors, and socioeconomic forces interact to shape patterns of livestock grazing and wildfires.

4.1. Conclusions

Our findings indicate that livestock grazing may play a role as one component of a multi-pronged approach to reducing burn probability as California confronts the legacies of a century of fire suppression and the increasing impacts of climate change. With grazing data from over 100 ranchers, we demonstrate that the negative effect of grazing on burn probability, previously demonstrated through small-scale studies, holds true across broad spatial and moderate temporal scales in some regions of California's fire-prone landscapes. Nevertheless, there was important variation in our results: in the Central Valley and Foothills region, we found that burn probability increased with grazing pressure, although the mechanism behind this relationship is unclear. These results have implications for land managers seeking to reduce burn probability, providing insights into which locations are most likely to benefit from fuel reduction via grazing and the stocking levels required to achieve these benefits.

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Authors contributions

All authors conceived the project; LM, MS, TB, SL, FM, and LW collected data; KS and VB analyzed the data; KS led the writing. All authors contributed to the manuscript and gave final approval for publication.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

All data used are publicly available except for the private property shapefiles and stocking levels, which the authors do not have permission to share.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2022.116092.

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