# Human population growth and accessibility from cities shape rangeland condition in the American West

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**ABSTRACT**

Rangelands from the western United States are drylands threatened by multiple factors including climate change, increasing wildfires, and invasive plants. The region also includes several of the fastest growing cities in the United States. How rapid human population growth will impact rangeland condition remains unclear. We quantified impacts of human population growth and accessibility (i.e., travel time from population centers) on rangeland condition relative to other well-established drivers, including wildfire, climate, and land ownership. Rangeland condition was represented as the relative dominance of ecologically relevant vegetation components. We modeled rangeland condition across 111 million hectares over three decades (1989, 1998, 2008, and 2018) using a Bayesian spatio-temporal regression. Finally, we applied our modeling framework to the three counties with the highest population growth rates to test whether different outcomes had emerged at smaller spatial extents. We found that rangeland condition decreased by a total of almost 13% from 1989 to 2018. Climate and wildfire were the strongest drivers of rangeland degradation. Our findings highlight human population growth as a potential threat to rangeland condition, although accessibility from major cities was associated with increased rangeland quality. This result suggests that while areas in close proximity to cities receive increased management effort, remote areas located in fast-growing counties experience the cost of increased human use without commensurate management. Our results were consistent at both the regional and county-level scales. An integrated policy platform that addresses both biophysical and social drivers of rangeland degradation will be necessary to protect these dryland ecosystems.

**KEYWORDS:** *Landsat, rangeland degradation, landscape planning, American West, sagebrush ecosystems*

## 1.0 INTRODUCTION

Drylands cover 40% of the global land surface and more than 3 billion people worldwide are dependent on the ecosystem services (ES) that they provide (Hoover *et al* 2020). Dryland ecosystems are subject to multiple forces of change, including climate change, invasive plants, and increasing wildfires (Maestre *et al* 2016a). For example, human pressure and land use intensification can quickly induce changes in healthy drylands that lead to degradation, thus reducing their capacity to recover from extreme climate events (Gunderson 2000; Sun *et al* 2021). We define degradation as a transition from a functional native ecosystem to an undesired state that may include lower biodiversity and provide fewer ecosystem services (Maestre *et al* 2016b). For many drylands ecosystems, reversing degradation is challenging, due to hysteresis (Scheffer *et al* 2001; Suding *et al* 2004). Potential cascading feedbacks between multiple drivers of change emphasize an urgent need for research on socio-ecological dynamics in drylands to inform sustainable land management (Nkonya *et al* 2011).

Sagebrush steppe of the western United States is a dryland ecosystem that exemplifies numerous threats

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to ecosystem condition (Chambers *et al* 2017). Once extensive in cold deserts across the western US, the sagebrush ecosystem now occupies less than half of its former extent (Davies *et al* 2011; Requena-Mullor *et al* 2019). The loss and degradation of sagebrush steppe jeopardizes ES, including forage for cattle (Davies *et al* 2014), and threatens endemic biodiversity (e.g., greater sage grouse and Columbia Basin pygmy rabbit [Coates *et al* 2016]). Considerable research efforts have focused on monitoring the condition of sagebrush-steppe rangelands, and quantifying the relative impacts of different drivers. Healthy sagebrush steppe ecosystems are characterized by the presence of native shrubs and perennial herbaceous plants (Rigge *et al* 2019; Rigge *et al* 2020). Wildfire is one of the most important drivers of perturbation, leading the ecosystem to an alternative stable but degraded state from which sagebrush and other native shrubs cannot easily recover naturally after large fires, but invasive annual grasses can (Balch *et al* 2013; Bradley *et al* 2018). While fire is a natural part of the sagebrush-steppe ecosystem, climate change and invasive species increase the prevalence and frequency of wildfire, driving sagebrush steppe ecosystems to a permanently degraded state (Barbero *et al* 2015; Abatzoglou and Williams 2016; Smith *et al* 2021).

One potential driver of rangeland degradation is human population growth, which is affecting many formerly rural regions of the western U.S. While human population growth has been previously explored at the state-level (see for example Cameron *et al* 2014), how rapid human population growth may impact sagebrush-steppe rangelands across the western U.S. remains poorly understood. The region is also home to the fastest-growing metropolitan areas in the country, including several cities that have experienced an order of magnitude increase in population in the past two decades (Jones *et al* 2019). The immediate impact of population growth on rangelands is land conversion for development (Freedgood *et al* 2020). However, the ecological footprint of rangeland conversion is relatively small because the majority of rangelands in the West is under public ownership and cannot be developed (Maestas *et al* 2001). Despite the small footprint of developed land, amenity migration has resulted in increased recreational use on surrounding rangelands, including camping, off-road vehicles, and recreational shooting. Recreational uses of rangeland ecosystems represent important stressors that are widespread and understudied (Winkler *et al* 2007; Narducci *et al* 2019). Increased recreational use brings the potential for invasive species spread, unauthorized trail development, and additional ignition of fire.

Evaluating the impacts of human population growth on unconverted rangeland ecosystems relative to other drivers is complicated because the spatial patterns associated with human population growth are likely different from those of fire, grazing, and climate change. Human population growth is typically localized in private lands around a few cities, but the majority of rangeland is located in rural areas with very low population density. Although urban and rural areas are linked through multiple and complex processes such as the flows of people, economic goods, and land use change (Seto *et al* 2012; Güneralp *et al* 2013; Tan and Li 2013), to our knowledge, the link between human population growth and the diffuse impacts on rangeland ecosystem condition has not been empirically measured on a large scale.

Land tenure is an essential consideration when modeling broad-scale dynamics of rangeland condition (Cumming and Epstein 2020). The history of colonialism in the United States has generated spatial variability in both ecosystem productivity and land management practices (e.g., livestock grazing, fire, invasive plant control, and recreational use) across public, Tribal, and private lands. For example, while private owners owned desirable areas to homesteaders, such as valley bottoms with access to water, Native American communities, originally the largest landholders in the West, were left a tiny portion of the landscape in the form of Reservations wherein Tribal governments retain sovereignty and land management authority with each Tribe implementing its unique management according to its objectives (Kimmerer and Lake 2001). Public lands typically have a multiple-use mandate, and thus are required to allow recreation and resource extraction by a wide-range of stakeholders. The general result of this history is that the highest quality ecosystems were allocated to the private domain, while less productive, and/or previously degraded ecosystems, became part of the public or tribal domain.

The overarching objective of this paper is to measure the impacts of human population growth on rangeland ecosystem condition across the western U.S., relative to other biophysical and social factors. We formulated several alternative hypotheses about the effect of localized population density on rangeland conditions. First, human population growth in localized areas may lead to degradation of rangeland conditions on nearby public lands with a multi-use mandate, but not on private or tribal lands where access is typically more restrictive than public lands (Maestas *et al* 2001; Donnelly *et al* 2016). Second, increased demand for outdoor recreation could lead to increased pressure and therefore degradation, even on remote rangelands that are open for recreation (Wittman and Bennett 2021). On the other hand, human population growth may have positive impacts on rangelands. For example, increased allocation of management resources to areas close to cities could lead to improved rangeland ecosystems through changes in grazing management, increased emphasis on fire control, reduced fuel loads, and expanded restoration efforts. These contrasting possibilities would suggest different policy pathways, from maintaining focus on land degradation in rural areas to shifting focus towards public lands around recreation centers.

To test those hypotheses, we use a satellite-derived time-series dataset of ecosystem condition from 1989 to 2018 (Rigge *et al* 2019; Rigge *et al* 2021a,b), several hypothesized drivers of ecosystem condition (human population growth, land ownership, accessibility, wildfire, climate and topography), and a Bayesian spatio-temporal mixed model approach. We supplement our region-wide analysis with a more in-depth examination of our predictor variables for the three counties undergoing the highest human population growth rate. Finally, we discuss our results in the context of relevant policy solutions to address cross-scale threats to rangelands.

## 2.0 Methods

### *2.1 Study area*

The study area comprises approximately 111 million hectares across 121 counties, in nine states of the western U.S. (figure 1(a)), including most of the area historically covered by sagebrush. The mean annual temperature averaged across the study region is 10.4 °C, varying from -3.8 to 24.6 °C based on 1981–2019 Daymet climate data (Thornton *et al* 2014), and the cumulative annual precipitation ranges from 5.4 cm to 174.5 cm, with a mean of 38.9 cm. The region is dominated by a heterogeneous topography (elevation range is 4,219 m). The central and northern portion of our study region encompasses an array of native shrub and perennial forb and grassland communities, where big sagebrush (*Artemisia tridentata* spp. Nutt.) is the dominant species in healthy communities. In the south, aridity-tolerant non-sagebrush shrub species are more abundant, while big sagebrush becomes sparser and occupies wetter locations. In burned areas, the composition and structure of plant communities are simplified by the invasion of exotic species such as cheatgrass (*Bromus tectorum* L.).

Regarding land tenure in the counties included in our study area, a majority (61.85%) of rangelands are managed by various agencies of the federal government in the public trust. Private owners own 31.25% of the land, while Native American communities own ~2.33% of the landscape. The remainder of the study region is managed by state or local agencies (4.57%) for a variety of objectives.

### *2.2 Datasets*

***Rangeland condition:*** We used a subset of the National Land Cover Dataset (NLCD) Back in Time dataset, a dataset of fractional components of sagebrush steppe ecosystems (i.e., sagebrush, non-sagebrush shrub, perennial herbaceous, and annual herbaceous) derived from Landsat imagery from 1985 to 2018 (Xian *et al* 2015, Rigge *et al* 2021a; Rigge *et al* 2021b). The NLCD is a well-validated and commonly used satellite-derived dataset that has been extensively validated with field data across 16 states from the western U.S. (Xian *et al* 2015; Rigge *et al* 2020a). Remote sensing has enabled measurements of vegetation cover with relevance to ecosystem services (Hudson *et al* 2021).

We integrated the four fractional components into a single measure of rangeland condition, the Actual Score Index (ASI) (Rigge *et al* 2019 provide a complete description of the ASI). The ASI represents community composition/structure by considering both the type and amount of cover based on the weighted linear combination of the cover of each fractional component. Weight values consider the histograms of component cover, giving high weights to those components that are more ecologically relevant in non-degraded rangeland sites, e.g., sagebrush and non-sagebrush shrub (see Table 1 in Rigge *et al* 2019 for details). Overall, higher ASI values represent sites with abundant sagebrush and/or non-sagebrush shrub cover and perennial grasses (both associated with lack of disturbance), while low values are linked to sites where annual grasses are dominant. To make the analysis computationally feasible, we chose four different years, one in each decade, to analyze change over three 10-year timesteps (1989-1998, 1998-2008, and 2008-2018). Thus, we chose 1989, 1998, 2008, and 2018 with the aim of 1) maximizing the length of our study period by starting in 2018 (i.e., 30 years), and 2) being able to split this period into shorter, consecutive, and same-length intervals (i.e., 10 years each) that allowed the model to converge while accounting for the computational cost. We then computed the ASI using the four fractional components on a scale of 30 x 30 m following Rigge *et al* (2019). Then, we generated 10 000 random points over the study area (figure 1(a)) and filtered the points that did not match non-rangeland areas, e.g., urban areas, agriculture, water bodies, and forests. After filtering, we collected the ASI from 7766 points over four years (1989, 1998, 2008, and 2018), resulting in 31064 observations.

***Covariates:*** We included several hypothesized socio-economic and biophysical predictors as covariates in our model. Human population was collected per county and year (figure 1(b)). Accessibility measures the distance in minutes to major cities (i.e., cities of 50 000 or more people in 2000) and represents how easily a location can be reached from cities (figure 1(c); see Appendix A for more details). In addition, we considered two fire attributes (fire occurrence and the number of fires; see Appendix A for more details), two topographic variables (slope (degrees) and elevation (meters)), and two climatic variables (annual cumulative precipitation (mm) and mean annual temperature (°C)). These biophysical covariates have a demonstrated impact on sagebrush cover, a key component of ASI (Requena-Mullor *et al* 2019; see Appendix A for a complete description of the covariates).

All covariates (except fire occurrence) were standardized to have a mean of zero and standard deviation of 1. Finally, we included land ownership as a contextual variable to determine whether our hypothesized drivers of ecosystem condition varied depending on the land ownership. We included four broad categories (federal, private, state-local, and tribal) (see Appendix A for a description).

### *2.3 Broad scale trends*

Prior to testing our hypothesized predictors, we described the change in conditions of sagebrush steppe over time. We computed temporal trends for rangeland condition across landowner types using ordinary least squares linear regression (Appendix B). We also calculated descriptive statistics per pixel to measure percentage of change (POC) in rangeland condition per decade with respect to the entire period.

$$POC = (ASI_{t2} - ASI_{t1}) / |ASI_{2018} - ASI_{1989}| \bullet 100 \qquad [1]$$

where $t_1$ and $t_2$ represent the first and second year of each decade.

### *2.4 Modeling of sagebrush steppe condition*

We applied a Bayesian spatio-temporal mixed model approach to test the relative impact of hypothesized drivers of rangeland condition for three 10-year time steps. We considered the condition of rangeland ecosystems to be a gamma process ($y_{it} \sim Gamma(a_{it}, b_{it})$) with mean $E(y_{it}) = \mu_{it} = a_{it}/b_{it}$, at pixel *i* and year *t*. Our model was defined on $\log(\mu_{it})$ as a combination of linear functions of covariates, varying intercepts for the county, varying intercepts and slopes for land tenure, and a spatio-temporal random effect:

$$\log(\mu_{it}) = \Sigma_{m=1}^{2}(\beta_0+\delta_{0m}) + \Sigma_{p=1}^{8}(\beta_{1p}+\delta_{1p})\ x_{itp} + \xi_{it} \qquad [2]$$

where *i* = 1, …, 7766, and *t* = 1989, 1999, 2009, and 2018 (N = 31064). Additive parameters include an intercept for rangeland condition ($\beta_0$), and slope terms ($\beta_{1p}$) for the effect of human population, accessibility, annual cumulative precipitation, mean annual temperature, slope, elevation, fire occurrence and number of fires. The intercept was allowed to vary by county and land tenure types ($\delta_{0m}$) to control for "contextual" dependency in the observations belonging to the same category. Likewise, the effects of the covariates ($x_{itp}$) were allowed to vary through land tenure types ($\delta_{1p}$). Effect sizes were calculated as percentages representing the rate of change in the ASI when the predictor increases by 1 SD.

To avoid biased model estimates due to autocorrelation, we included a random effect ($\xi_{it}$) to account for residual spatio-temporal structure in rangeland condition due to unmeasured covariates (e.g., soil type, grazing history). We use a stochastic partial differential equation approach to model spatio-temporal random effects (Lindgren *et al* 2011). This approach translates a continuous spatial process, modeled using the Matèrn covariance function (Cressie 2015), into a discretely indexed spatial random process. The spatial random process can then be extended to the spatio-temporal case by including a time dimension. This representation allowed us to account for the residual structured variance that was not explained by the covariates included in the model. See Appendix C for further details on the modeling approach.

### *2.4.1 County-specific model*

We explored the relationship between rangeland condition and drivers by fitting new data points from the three counties that experienced the highest human population growth rate from 1989 to 2018. These counties were Ada County, Idaho, with a human population growth rate of 133.9%; Tooele County (162.4%), Utah; and Lyon County (191.4%), Nevada (Appendix D). We randomly sampled a large number of data points (10970, 15000, and 24364, respectively) in each county and extracted the ASI and same covariates as we used for the region-wide model. In Ada county only, human population growth and temperature were correlated (Spearman's rank correlation rho = 0.5). This correlation reveals that in time periods with high human population growth, the temperature in Ada County also tended to increase. As our goal here was to estimate the effect of human population on ecosystem condition, we discarded temperature in the model for Ada County. We found minimal correlation between temperature and human population growth for Lyon and Tooele county, so retained both covariates in these models. The limited spatial extent of an individual county enabled us to model rangeland condition with near-continuous coverage across the county. Moreover, modeling each county separately enables more direct comparisons between county-level data fit with independent models.

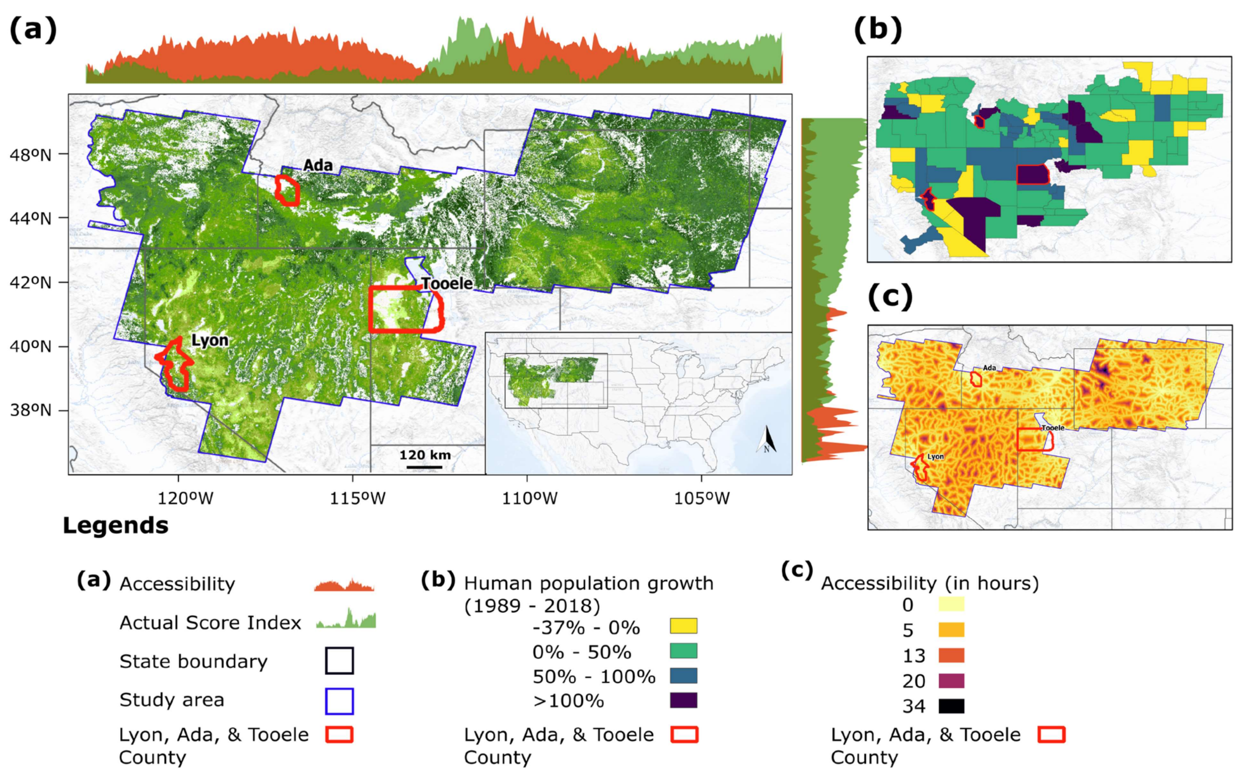


**Figure 1.** (a) ASI 2018 image of the study area (blue line). The study area boundaries were defined by the Worldwide Reference System path-rows. The marginal graphics represent the median accessibility and the actual score index (ASI) computed by rows and columns (pixel size: 30 x 30m). For example, rangelands in the northeastern part of the study area have lower accessibility (i.e. higher travel time) and higher ASI (i.e., better rangeland condition) than southeastern areas. Base map source: USGS National Map Services. (b) Human population growth measured as the magnitude of change occurred from 1989 to 2018 per county. (c) Accessibility measured as the distance in hours to major cities (i.e., cities of 50,000 or more people in 2000).

## 3. Results

### *3.1 Broad scale trends*

Over the sampled pixels, rangeland condition, measured as ASI, decreased by 44.6% (±6% SE) and 70.3% (±7.5% SE) in the first two decades respectively, and increased by 61.4% (±7 SE) from 2008 to 2018. We note that our estimates of decline are conservative as developed pixels (i.e., the most dramatic change in rangeland condition) are masked out of subsequent years. The total decrease in ASI from 1989 to 2018 was almost 15% (±0.44 SE). Linear trends in rangeland condition are shown in Appendix B, with an overall decrease in the four land ownership types. Federal and Tribal lands achieved the highest and lowest percentage of pixels with significant negative trends, respectively.

### *3.2 Model Performance and Regional Results*

Our results demonstrate negative impacts of human population growth on rangeland condition at regional scales. The ASI decreased by 3.79% with a population increase of 59,602 people in a county/year (95% CI: -5.88%, -1.63%) (figure 2 and 3(a)). In contrast, accessibility from a population center had a positive effect on rangeland condition, which decreased by 1.88% (95% CI: -3.67%, -0.02%) when the travel time from a population center increased by ~3.5 hours (figure 3(b)). In addition to socioeconomic variables, biophysical variables also impacted rangeland condition, including temperature, by far the strongest predictor. A temperature increase of 2.6°C was associated with a 29.85% decrease in rangeland condition (95% CI: -32.45%, -27.05%). The occurrence of a wildfire was the next strongest predictor, and was associated with a 9.34% decline in rangeland condition (95% CI: -12.9%, -5.65%). Relative to wildfire occurrence and temperature, human population, the number of wildfires, and accessibility were the next most important predictors. The slope was the predictor with the weakest influence on the sagebrush condition with a predicted increase of 1.4% when the slope increased by 7.6° (95% CI: -0.78%, 3.4%).

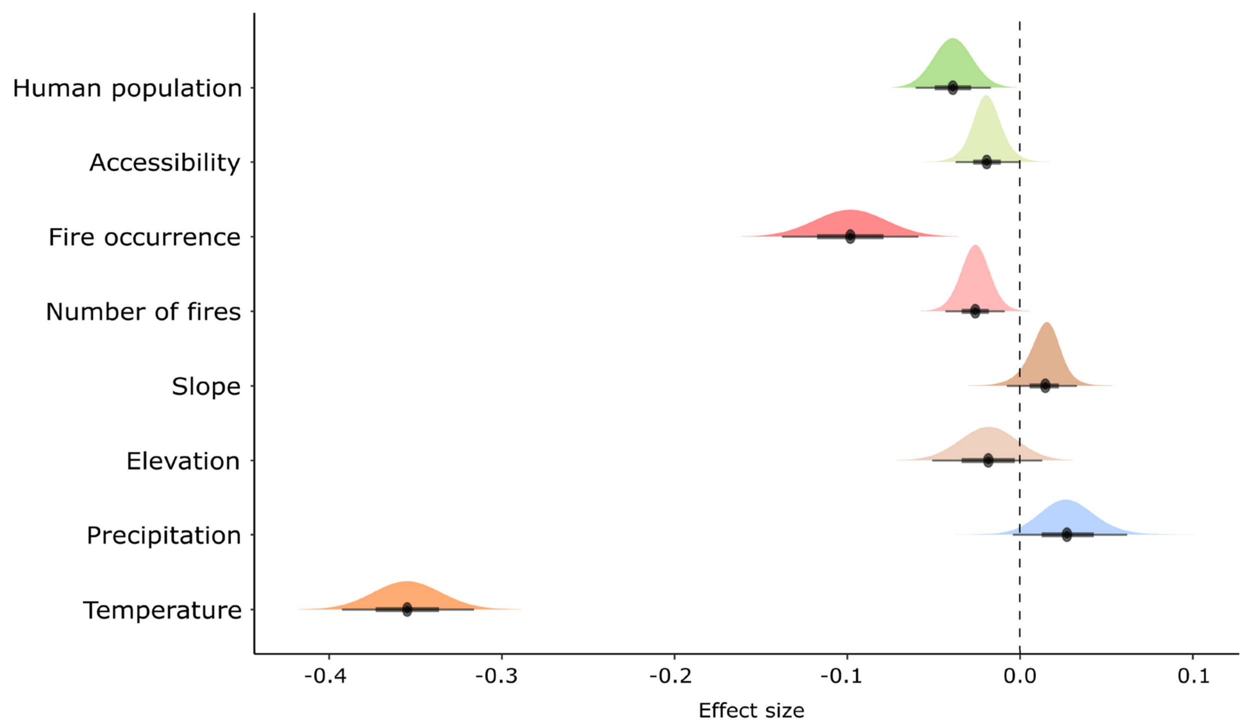


**Figure 2.** Region-wide model showing the relationship between rangeland condition and covariates. Effect sizes are shown on a log scale. Posterior distributions of effects are colored by covariate. Posterior mean (black dots), and 95% and 50% credible intervals (thin and thick lines, respectively).

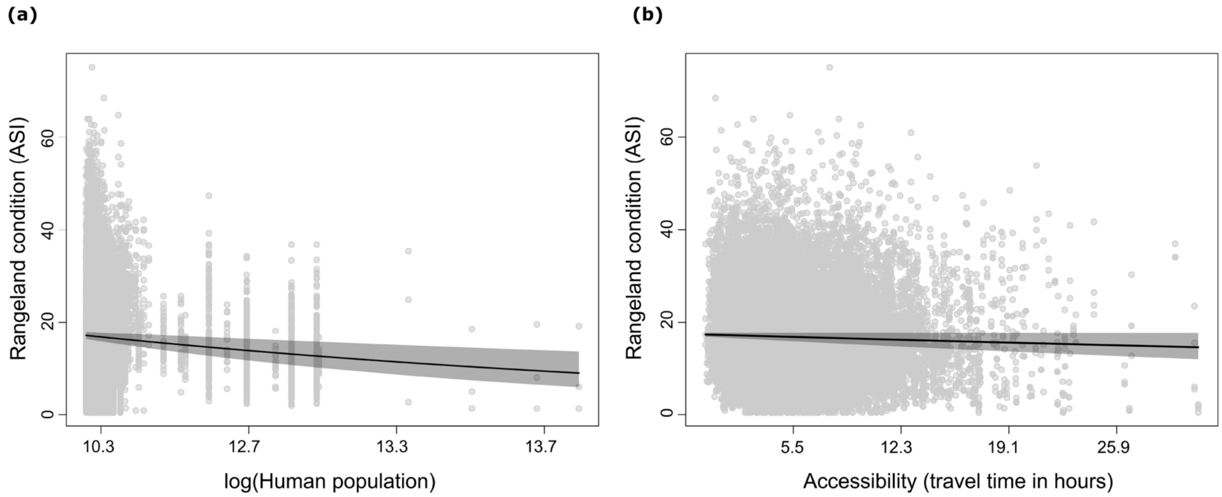


**Figure 3.** The predicted marginal effect of human population growth (a) and accessibility (b) on rangeland condition. Dark shadow represents the 95% credible interval.

Out-of-sample predictions indicated that both covariates and the spatio-temporal term achieved an averaged performance improvement of ~44.7% (±0.67% SE) with respect to the model that included both the intercept and spatio-temporal term, and an improvement ~35% (±0.57% SE) when the spatio-temporal term was removed from the model (Appendix E). Thus, the spatio-temporal random effect term captured ~9.7% of the variation from the structured errors (i.e., 44.7% — 35%). The Variance Inflation Factor (VIF) was lower than 5 for all the covariates and below 3 for most of them (Appendix F). These values suggest that interactions between covariates were minimal.

We found slight differences in rangeland condition across the four different land ownership categories (figure 4). Federal lands and tribal lands had a rangeland condition that was 2.94% (95% CI: -5.94%, -0.03%) and 0.78% (95% CI: -4.82%, 2.9%) lower than average, respectively, and state/local and privately-owned lands had a ~2% greater than average rangeland condition, but the range of these differences strongly overlap with zero. We also found little evidence that land ownership influences the other hypothesized drivers of rangeland condition (Appendix G).

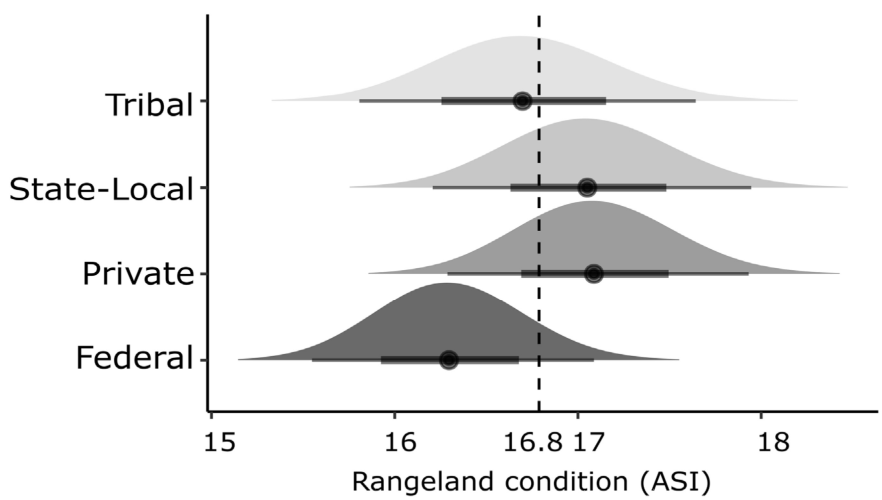


**Figure 4.** Variation in rangeland condition across land ownership types. Posterior distribution of rangeland condition estimated for each land ownership type. Posterior mean (black dots) ±95% and 50% credible intervals (thin and thick lines, respectively). Posterior mean intercept estimated across land ownership types (vertical dashed line).

*3.3 County-level analysis*

We found a consistent negative effect of human population growth in all three counties (figure 5). The county-level models that included all covariates and the spatio-temporal term achieved averaged performance improvements of 45.5% (±0.4% SE), 52% (±0.7% SE), and 54.5% (±0.4% SE) with respect to the models that included both the intercept and spatio-temporal term for the Ada county, Tooele county, and Lyon county, respectively (Appendix H).

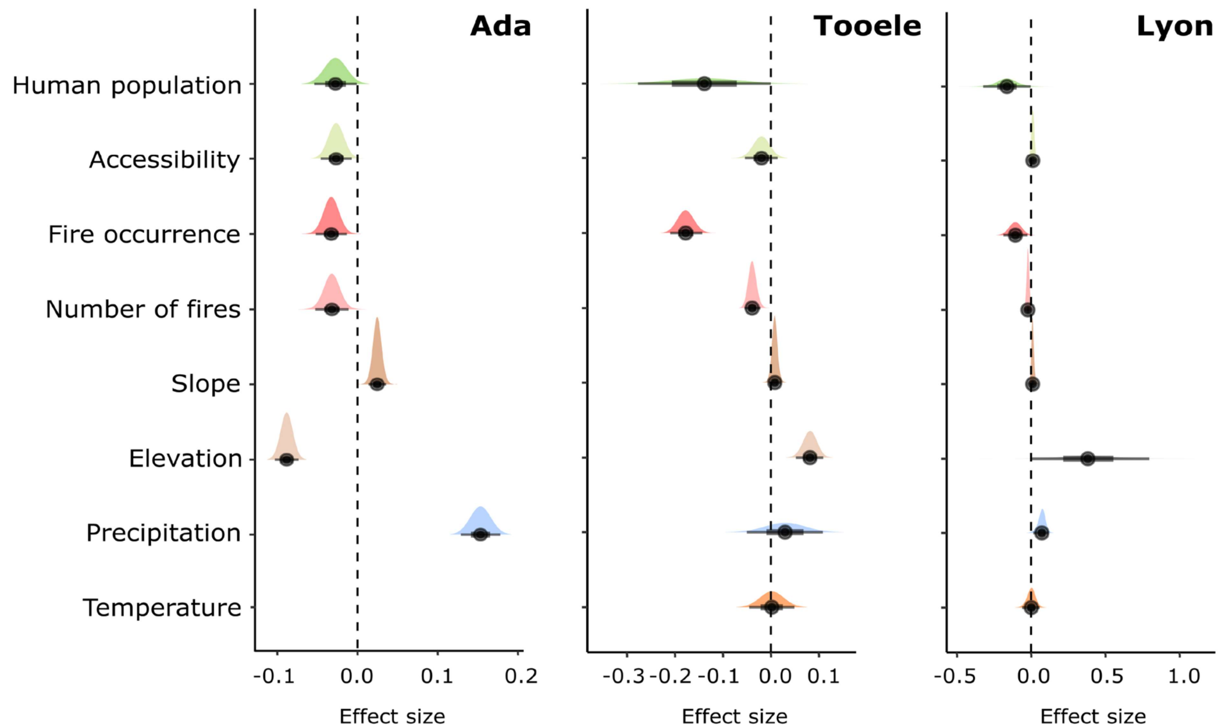


**Figure 5.** County-level models showing the relationship between rangeland condition and covariates in Ada county, Tooele county, and Lyon county. Ada county model did not include temperature because of its correlation with human population growth. Effect sizes are shown on a log scale. Posterior distributions of the estimated effects are colored by covariate. Posterior mean (black dots) ±95% and 50% credible intervals (thin and thick lines, respectively).

## 4.0 Discussion

### *4.1 Human population growth and impacts on rangeland condition*

Our first suite of hypotheses assessed the directionality of the relationship between human population growth and rangeland condition. We found strong evidence for the hypothesis that human population growth has contributed to the decline of rangeland condition in western rangelands. Our regional model indicated that population growth had a similar impact to fire frequency and precipitation, two widely recognized biophysical drivers of rangeland condition. Nevertheless, impacts of population growth were small relative to the effect of temperature (Balch 2013; Williamson *et al* 2020; Rigge *et al* 2021a). The results of our county-level analysis are consistent with that of our full sample, and provide further evidence that human population growth was strongly associated with lower rangeland condition.

We were unable to causally test the impacts of land tenure because of the complicated land use legacy of the public-private-tribal allocation of land (Sheridan 2007). A confounding factor is that ecosystem condition was a prerequisite for determining land ownership. Highest quality ecosystems were allocated to the private domain, while less productive, and/or previously degraded ecosystems became part of the public or tribal domain. Because of this land use legacy, it is difficult to disentangle causal impacts of recent land management activities on rangeland condition versus historical legacies, and furthermore, these impacts are highly variable depending on the local social-ecological context. However, we included land ownership as a contextual variable in our regional model to gain some insights as to general patterns between different land tenure types. Our reasoning was that if recreational use is one of the main stressors associated with population growth, then we would see a higher impact on public versus private lands. As expected, our model results indicate that private lands tend to have more suitable rangeland conditions than public and tribal lands, but these differences were minimal, and we did not see that population growth had a more negative impact on public than private lands. This result parallels a recent analysis of remotely sensed net primary productivity across the continental USA, which also found minimal differences in trajectories of productivity across public, private, and tribal land ownership (Robinson *et al* 2019). Within our study area, these results suggest that ongoing declines in rangeland condition are not limited to a single jurisdiction. Given that private land owners typically also manage grazing allotments on public land near their property (Runge *et al* 2019), spatial impacts of climate change and human population growth may transcend land boundaries.

Our second suite of hypotheses assessed the impacts of accessibility on rangeland condition, we hypothesized that impacts of population growth would also be detectable in rural countries located near to urban areas. We used travel time from a major city as a measure of accessibility, and found that shorter travel time to urban areas was correlated with increased rangeland quality. A spatial interpretation of this result is that population centers are surrounded by an area where rangeland condition is buffered from negative effects of population growth. These results also support our alternative hypothesis about accessibility, which is that higher accessibility may translate into improved management (e.g. wildfire fighting, restoration) and reduced fuel load compared to relatively inaccessible areas. For example, firefighting crews and aircraft are more likely to be dispatched to protect homes in general and higher valued homes in particular, than comparable uninhabited areas (Bayham and Yoder 2020). Furthermore, roads may reduce load fuel and serve as fire breaks. Large fires mainly occur in remote areas where road density is usually low (Narayanaraj and Wimberly 2012). In contrast, a decrease in fire occurrence has been observed in more accessible areas because of landscape fragmentation, fuel insufficiency, or higher fire suppression efforts (Zumbrunnen *et al* 2012). Another potential explanation is that post-fire restoration efforts can be greater near population centers because citizen interest in these ecosystems has greatly increased over the past half-century (Havstad *et al* 2009). Therefore, land management agencies are interested in increasing the level of trust in their capacity to implement management actions (Shindler *et al* 2011). Urban areas may also have access to more resources for habitat conservation relative to rural areas. For example, the City of Boise in Ada County manages 5000 acres of open space near the city, funded in part through multi-million dollar property tax levies (City of Boise 2005). The City of Bend in Deschutes County includes around 4000 acres of non-developed open space lands and establishes an array of non-regulatory measures to preserve natural areas within its urban growth boundary (City of Bend 2016). Moreover, population movement from urban into rural areas for lifestyle reasons, i.e., the so-called "New West" rural communities, are higher-income amenity migrants who support sustainable landscape management through funding resources, community programs, and conservation practices (Paveglio *et al* 2015). All these conservation efforts represent independent strategies in response to different human population growth. Finally, appreciative recreational activities with minimal environmental impact (e.g. hiking) may be more common close to urban areas, while off-road recreation with greater potential for environmental damage (Switalski 2018) may take place further from population centers (Krannich *et al* 2011).

### *4.2 Benefits and limitations of our modeling approach*

Our analytical approach used data for ecosystem conditions and hypothesized drivers covering broad spatial and temporal scales. Taking a broad scale approach is important because ecosystems are increasingly stressed by multiple, interacting biophysical and social factors that induce subtle, but widespread, changes in ecosystem condition over long time periods (Maestre *et al* 2016a; Jones *et al* 2019). We found that while rangeland condition declined over the entire study period, it increased in the last decade (i.e., 2008-2018). Rigge *et al* (2021) showed that the cover of the main sagebrush-steppe fractional components, i.e., sagebrush and non-sagebrush shrub, increased from 2004/5 to 2018, although there was an overall decrease over the period of 1985-2018. Our findings were consistent with this result, indicating that the rangeland condition in the last decade belonged to a longer increase period. Measuring the relative impacts of the driving forces is essential in order to enact appropriate policy solutions. Ever-increasing amounts of time-series data on ecosystem conditions from satellites, including the multi-decadal, globally available Landsat archive, now enable inference at spatial and temporal scales that match land management challenges (Yang *et al* 2018). Ultimately, spatially-explicit models of ecosystem dynamics could inform spatially-targeted interventions with potential to lower cost and improve land management outcomes (Strassburg *et al* 2019; Chambers *et al* 2017).

While our model allows for hypothesis testing about the general relationship between rangeland condition and hypothesized drivers, we were not able to disentangle what specific processes may be responsible for the observed relationships. Human population growth in the region is associated with a suite of socio-ecological changes, many with the potential to impact vegetation cover and plant community composition. Generally speaking, economic shifts associated with the influx of new residents include a land use transition from resource production to recreational consumption in natural areas (Sheridan 2007). Higher recreational land use in counties with higher influx of new residents is also a plausible driver of declines in rangeland quality (Krannich *et al* 2011). Finally, feedbacks between human population growth and biophysical drivers of rangeland degradation, such as increased spread of invasive grasses near roads and powerlines (Bradley and Mustard 2006) or increased human ignition of wildfires (Balch *et al* 2017) could also be responsible for the observed relationships. While grazing can have a range of impacts on rangeland condition, depending on cattle density and frequency of grazing (Root *et al* 2020; Williamson *et al* 2020; Davies *et al* 2014), ranchers may be motivated to maintain rangeland condition in ways that real estate developers are not (Brunson and Hunsinger 2008).

Despite the likely importance of grazing intensity for rangeland quality, the lack of a consistent dataset that characterizes grazing intensity did not allow us to consider grazing influence on rangeland condition. Livestock grazing is the most common land-use in the sagebrush biome; however, data on the number of livestock, grazing intensity, and season of use is reported inconsistently on public lands and not at all on private lands making it difficult to adequately capture the role of livestock in the system at the grain and extent of our study. These challenges may explain why there are no regional studies that have modeled and projected changes in rangeland fractional component cover while explicitly including grazing intensity (Rigge *et al* 2021b). Still, our Bayesian spatio-temporal models are designed to account for missing covariates, including grazing intensity, to strengthen inference on the effects of our primary covariates. In agreement with previous studies, climatic and fire covariates showed a large influence on rangeland condition (Bates *et al* 2006; Davies *et al* 2012), which supports the robustness of our modeling approach. We found that the spatio-temporal random effect term in our models captured almost 10% of the variation from the structured errors in the broad-scale model. This result provides a quantitative estimate for the importance of unmeasured spatial variation due to grazing and other factors, and emphasizes the need for further studies with regional-scale covariates. Acquiring regional-scale data for grazing intensity will likely require significant effort, including manually scanning and extracting information from rangeland management records (Swette and Lambin 2021).

### *4.3 Relevant policy solutions to address cross-scale threats to the sagebrush ecosystem*

The growing list of compounding and ubiquitous drivers of sagebrush decline highlights the need for a more integrated approach to land management planning. For example, allotment management plans for federal grazing permittees are often focused on achieving livestock production targets that are acceptable to the permittee and mitigating impacts associated with livestock grazing (Swette and Lambin 2021). Recreation objectives and management are determined via a completely separate process (e.g., Travel Management Planning). Land use and development decisions made by multiple jurisdictions through a myriad of disconnected planning processes further compound this complexity. The general decline in ASI that we measured across the region and regardless of tenure indicates that mitigation efforts designed to reduce the impacts of these other land uses are failing to keep pace with the growing intensity of threats. This highlights a need for a more coordinated approach focused on achieving ecosystem integrity objectives across jurisdictions rather than simply mitigating the effects of myriad land uses.

Given the current state of sagebrush shrublands, it is unlikely that restoration of all shrublands to a high level of quality is ecologically or financially possible (Chambers *et al* 2017). Instead, we envision a portfolio approach where some portions of the landscape are targeted for intensive investment to maintain existing social and ecological processes, some are targeted for intentional transition from current land uses and vegetation communities to those that may be more compatible with an uncertain climatic future, and others are left unmanaged to provide managers and researchers with a baseline to evaluate the impacts of these decisions (Aplet and McKinley 2017; Maestas *et al* 2022). Such an approach may focus on restoring local ecosystems via seeding or restoring local economies via targeted investment in important industries in some areas and facilitating transitions to new ecosystems (e.g., fire prone annual grasslands) and economies (e.g., amenity industries) in others. Recent investments in open space preservation may also provide opportunities for cities and towns to mitigate regional environmental change. This approach can only be successful if decision-making and management is coordinated across jurisdictions and the metrics by which success is assessed are tied to explicit objectives for ecosystem integrity. We suggest such an intentional approach might provide an important alternative to the current approach which our analysis suggests is failing to meet ecosystem objectives across the bulk of the landscape.

## 5.0 Conclusions

Human population growth is often cited as a key factor in threats to ecosystem integrity. Since people live mostly concentrated in cities, whether or not the impact of human demographic change is detectable on a large scale remains unclear. Our study reveals the contribution of human population growth to the decline of rangeland condition across the western United States. Our results highlight the importance of accessibility when mitigating rangeland degradation and point out that an unequal allocation of management resources/efforts between areas surrounding population centers and remote rural areas may impact the rangeland condition on a broader scale. The increased demand for developed areas to cope with human population growth seems to be coupled with the higher availability of resources for rangeland conservation in areas close to cities compared to relatively inaccessible areas. Finally, an integrated policy platform that results in more coordinated management plans across jurisdictions will be necessary to protect these dryland ecosystems.

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