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RESEARCH ARTICLE

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Unintended habitat loss on private land from grazing restrictions on public rangelands

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Abstract

1. Management of public lands, and who should have access to them, is often contentious. Most ranches in the western US rely upon seasonal grazing access to public lands, and conflict over biodiversity management has led to proposals to restrict grazing access on public lands. We evaluate whether grazing restrictions on public rangelands could have the unintended effect of increasing the conversion of private rangeland to cropland, causing habitat loss for sage-grouse, a species of conservation concern.
2. Using a model parameterized with empirical observations of land use change and ranch versus cropland profitability, we explore how changes to public lands grazing policy could affect ranch profitability and consequently land use on private lands across the western US.
3. We predict that restricting grazing of public lands by 50% would result in the loss of an additional 171,400 ha of sage-grouse habitat on private lands by 2050, on top of the 842,000 ha predicted to be lost under business as usual. Most of this conversion would affect sage-grouse mesic habitat, 75% of which occurs on private land and is vital to the species during brood rearing. Under such policy changes, we estimate that an additional 105,700 ha (3.24%) of sage-grouse mesic habitat held on private land in the study region would be directly lost by 2050, and the cumulative area affected by fragmentation would be much higher.
4. By considering the human and ecological links between public and private land, we show that attempts to improve habitat on public lands via grazing restrictions

could result in greater system-wide fragmentation of sage-grouse habitat from unintended habitat loss on private lands.

5. *Synthesis and applications*. Policy interventions on public lands can affect private landholders. Landholders' responses can result in unintended consequences, both for habitat on private land and community support for conservation. Restricting grazing on US public lands is likely to increase habitat loss on private lands and reduce community support for sage grouse conservation. Policy that manages resources on public lands while also supporting sustainable, economically viable ranching operations on private lands is a promising approach to maximizing sage grouse habitat.

KEYWORDS

cropland, econometrics, grazing restrictions, land use, landholders, perverse outcomes, public lands, rangeland management

1 | INTRODUCTION

Policies intended to provide conservation benefits can have unintended consequences for conservation if such policies result in perverse incentives for landowners (Polasky, 2006). In particular, conservation actions can produce feedback effects through markets, with positive or negative consequences (Lim, Carrasco, McHardy, & Edwards, 2017). The conservation community is becoming more mindful of unintended feedbacks that can undermine conservation goals (Larrosa, Carrasco, & Milner-Gulland, 2016); though consideration of the linkages between people, markets, and ecosystems is far from common practice in analyses of conservation policies (Milner-Gulland, 2012).

In the western US, ranches typically make use of both public and private rangeland. Public land is grazed via long-term leases that are granted to private ranchers and these rights are transferred with ownership of the private land. Private lands are generally lower in elevation, higher in productivity and have more surface water compared to nearby public lands due to land disposal laws and resulting settlement patterns in the 19th century (Scott et al., 2001; Talbert, Knight, & Mitchell, 2007). However, ranches across much of the region are heavily dependent on public grazing leases, especially where the majority of the land base is publicly-owned (Torell et al., 2002). Ranching is economically marginal in many cases, and where rangeland can be converted to cropland or other uses, there is often an economic incentive to do so. Grazing on public land has come under increased scrutiny in recent decades as public attitudes have shifted to favour the scenic, recreational, and biodiversity conservation values of public lands. Partly in response to these pressures, grazing on public lands has gradually declined since the mid-20th century (Yahdjian, Sala, & Havstad, 2015).

The western US is home to many endemic and rare species that rely on intact sagebrush and prairie grasslands, including the greater sage-grouse (*Centrocercus urophasianus*). Sage-grouse have suffered widespread population declines and conservationists and ranchers share a goal of avoiding continued declines that would trigger listing under the U.S. Endangered Species Act (Duvall, Metcalf, & Coates, 2017). Sage-grouse require habitat on both public and private lands. These birds spend much of the winter and early breeding season on upland sagebrush, which is primarily on public lands, moving to mesic areas in late summer to raise their chicks. Most of this brood rearing habitat is on private land (75%; Donnelly, Naugle, Hagen, & Maestas, 2016). Thus, maintaining habitat across both public and private land is crucial for the persistence of the species.

Though tightly regulated today, grazing on public lands in the U.S. historically caused widespread ecological damage and remains controversial (Pool, 2009). Concern over perceived impacts of livestock grazing to sage-grouse recently prompted federal land management agencies to adopt guidelines specifying desired habitat conditions for sage-grouse to be used when evaluating management of public grazing leases. Failure to meet these conditions would likely trigger reduction or seasonal restriction of grazing access to a leased area of public land. However, the benefits of these guidelines are unproven and the relationships between livestock grazing and sage-grouse persistence remain obscure. Few studies have directly examined the role of grazing management on sage-grouse demography and these have failed to produce unambiguous evidence that a reduction in grazing provides benefits (Monroe et al., 2017; Smith, Tack, Berkeley, Szczypinski, & Naugle, 2018).

Here, we explore the unintended consequences of restricting grazing on public lands in the western US. Specifically, we apply a predictive econometric model to evaluate whether grazing restrictions on public lands could reduce ranch profitability, thereby increasing rates of conversion of rangeland to cropland on private lands. In particular, we examine the potential loss of sage-grouse mesic habitat, a critically important resource for sage-grouse that occurs disproportionately on private lands. Our study is an example of a broader need to consider the unintended consequences of conservation actions through economically driven models of land-use change (Alix-Garcia, Shapiro, & Sims, 2012; Sohngen, Mendelsohn, & Sedjo, 1999; Wu, 2000).

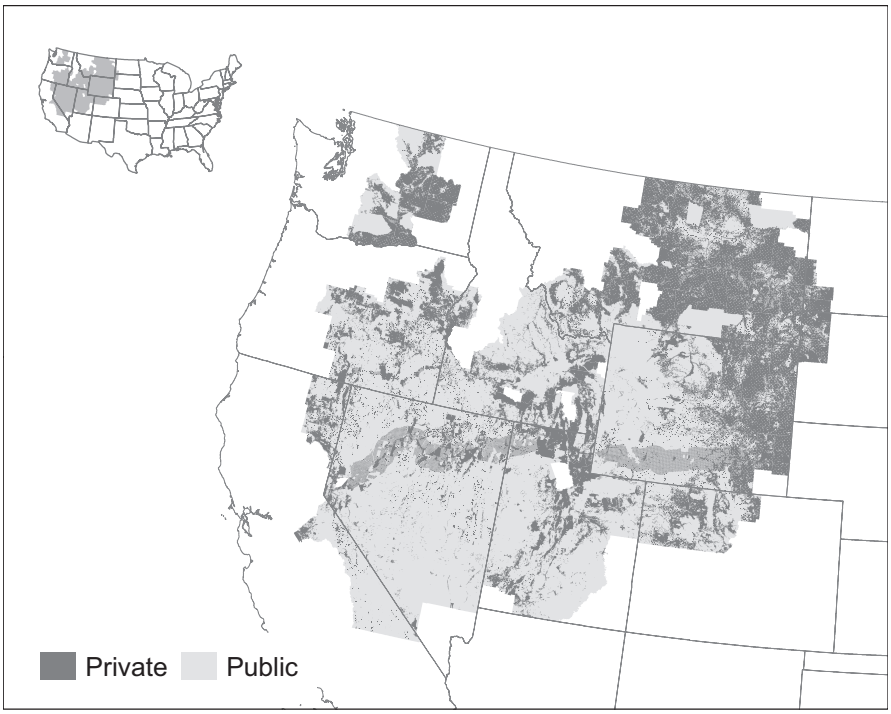

FIGURE 1 Map showing public (light grey) and private (dark grey) land on counties overlapping sage-grouse habitat and included in the study

2 | MATERIALS AND METHODS

2.1 | Study region

Our study area covered counties within 10 states (Colorado, Idaho, Nevada, Montana, South Dakota, North Dakota, Oregon, Utah, Washington, Wyoming) that contain sage-grouse habitat (Figure 1). We excluded coastal counties, highly urbanized counties and counties that do not overlap with sage-grouse management zones leaving 151 counties in the study region. Boundaries for private and public lands are drawn from the PAD-US CBI Version 2 (The Conservation Biology Institute (CBI), 2012), and we assume conversion can only occur on lands classified as "Private land".

2.2 | Econometric model

We predict land use change based on changes in ranch profitability using an econometric model. We parametrized this model using historical empirical data on land use changes and profits on rangeland and cropland. To estimate the effect of public land grazing restrictions on profits at the ranch level (recalling that ranchers graze livestock on both public and private lands), we estimated the proportion of forage that comes from public versus private land in each county and assumed that profits were reduced in proportion to decreased access to overall forage. A key assumption underpinning our model is that rangeland is more likely to be converted to cropland when the benefit obtained by having land in cropland outweighs all the other benefits that a rancher might obtain from having land in rangeland, whether they be financial, social or cultural benefits. Rather than assuming that all ranchers behave the same way and would convert their properties to cropland at a certain economic threshold, we use empirical data to evaluate the change in the rate of conversion as profit changes. This is further explained in Supporting Information Appendix S1, and the implication of this assumption more fully discussed in Section 4.

The econometric model follows earlier studies (e.g. Lawler et al., 2014) by expressing the probability of land-use change in each county as a function of net economic returns to different land-uses, with the other benefits obtained by ranchers from the different land uses included within the error term. Specifically, we empirically modelled the relationship between land rents and the probability of land use change (hereafter conversion rate) in each county for the period 2008–2012, using annual data at the county scale. We controlled for year, sage-grouse management zone, human population, proportion of cropland, area of urban land and proportion of irrigated cropland in each county (Equation 1).

To predict the effects of policy change, we first assume that rent from grazing is a function of the number of livestock that can be produced in a given county, itself a function of the productivity of that land and the area of land available for grazing. We then adjust the 2012 rent for each county by the expected proportional change in livestock production after policy implementation (Equation 3). From these adjusted rents, we predict the conversion rate after policy change from the empirically derived model coefficients (Equation 2). Finally, we use this conversion rate to estimate the area of sage grouse habitat and sage brush affected by the year 2050 under reduction of grazing access to public lands (Equation 4).

To analyse the effect of potential grazing restrictions, we made two assumptions that allow us to quantify the impact of grazing restrictions on net returns. First, we assume that ranch-level net returns are directly proportional to the forage available to a rancher on both public and private land, such that a loss of access to one-half of

TABLE 1 Model coefficients and heteroskedacisity–robust *SE*s

Model variable	Estimate	*SE*	*t*-Value	*p*
(Intercept)	−6.53	0.388	−16.844	<0.0001**
Cropland rent	6.64×10^{-3}	1.98×10^{-3}	3.355	0.0008**
Rangeland rent	-1.80×10^{-2}	6.77×10^{-3}	2.660	0.0078**
Proportion cropland	-6.05×10^{-1}	6.97×10^{-1}	0.867	0.3860
Percent cropland irrigated	-9.59×10^{-3}	3.69×10^{-3}	−2.601	0.0093**
Urban area (ha)	-2.32×10^{-4}	5.24×10^{-5}	−4.438	<0.0001**
Road density	-1.54×10^{-3}	3.65×10^{-4}	−4.232	<0.0001**
Population	2.40×10^{-5}	5.27×10^{-6}	4.551	<0.0001**
Proportion cropland: Percent cropland irrigated	3.31×10^{-2}	9.64×10^{-3}	3.434	0.0006**
Region Rocky Mountain	9.23×10^{-2}	2.09×10^{-1}	0.442	0.6588
Region Washington	6.90×10^{-1}	2.96×10^{-1}	2.335	0.0195*
Year 2010	1.89×10^{-2}	1.97×10^{-1}	0.096	0.9233
Year 2011	-1.13×10^{-1}	1.95×10^{-1}	−0.577	0.5639
Year 2012	-1.30×10^{-1}	2.01×10^{-1}	−0.646	0.5186

Note. Significant at *$p < 0.05$, **$p < 0.001$.

total forage would cut ranch-level net returns in half. Loss of grazing access to public lands has been shown to have a corresponding ranch profit loss (1% loss of animal-unit-months (AUM) ~0.95% profit loss; Taylor, Coupal, & Foulke, 2005; ignoring the effects of debt or off-farm income). Second, we assume changes in ranch-level net returns have effects on land-use change that are equivalent to the effects of the per-acre net returns that we used to parametrize our economic/land-use change model.

The model takes the form:

$$\ln(y_{it}^{RC}/y_{it}^{RR}) = \alpha_i + \delta_t + w_{it}^{C}\beta^{C} - w_{it}^{R}\beta^{R} + \tau_{it}, \quad (1)$$

where w_{it}^{C} is the crop rent in county i at time t and β^{C} is the estimated coefficient (parameters with superscript R are equivalently defined for rangeland), α_i represents a county fixed effect, δ_t is year of conversion (combined, the set of independent variables is denoted X), and τ_{it} is a random disturbance. Changes in crop and range rents affect $\ln(y^{RC}{}_{it}/y^{RR}{}_{it})$, the natural log of the area of rangeland in time t that converts to cropland by time $t + 1$, divided by the area of rangeland in time t that stays rangeland by time $t + 1$. The model is grounded in economic theory, as described in Supporting Information Appendix S1. We obtain the probability that rangeland converts to cropland p_{it}^{RC} (hereafter referred to as conversion rate) from the estimated model coefficients ($\hat{\theta}$, Table 1) and independent variables (X) according to Equation 2:

$$p_{it}^{RC} = (e^{X\hat{\theta}})/(1 + e^{X\hat{\theta}}) \quad (2)$$

With this equation, we predict the additional area converted into cropland with losses of grazing access on public lands. We modelled the loss of grazing access for a range (0%–100%) of decrease in public lands AUM, a measure of the number of livestock able to be fed on a given parcel of land, excluding strictly protected areas. We consider county-specific losses, where changes to w_{it}^{R} are a function of the proportional relationship between the forage available on public land and private land for grazing in a county, such that:

$$w_{it}^{R\prime} = W_{it}^{R}(P \times \text{public AUM}_i + \text{private AUM}_i)/(\text{public AUM}_i + \text{private AUM}_i), \quad (3)$$

where P is the proportion of forage on public land available for grazing after implementation of policy.

We parameterized this model with remotely sensed annual land-use change data (56 m resolution, Lark, Mueller, Johnson, & Gibbs, 2017; Lark, Salmon, & Gibbs, 2015) and annual data on pasture and cropland rental rates (NASS, 2015) adjusted to 2015 real US dollars, which reflect per-acre net returns for 2008–2012. Due to the use of aggregate data, it is customary to include additional explanatory variables to control for county characteristics, such as the amount of land available for conversion and urbanization pressures (Hardie & Parks, 1997). We include controls for year, sage-grouse management zone, human population, proportion of cropland, area of urban land, and proportion of irrigated cropland in each county (data sources for these variables are detailed below). We assumed conversion to cropland can only happen on private land and on land classified as "suitable for cropping" (Land Capability Class 1-6 gSSURGO; Soil Survey Staff, 2016), excluding water, forest, and developed land (USDA, 2015). We defined cropland as including all row crops, closely grown crops, or horticultural/tree crops, but not fallow, hay pasture or alfalfa, consistent with Lark et al. (2015). The most common crops in the region are alfalfa, dryland wheat, barley, and corn.

We obtained the area of urban land (urbanized plus urban clusters) in each county from 2010 US Census estimates (US Census Bureau, 2010). We averaged estimates of county population and population change across 2008–2014 values from the US Census

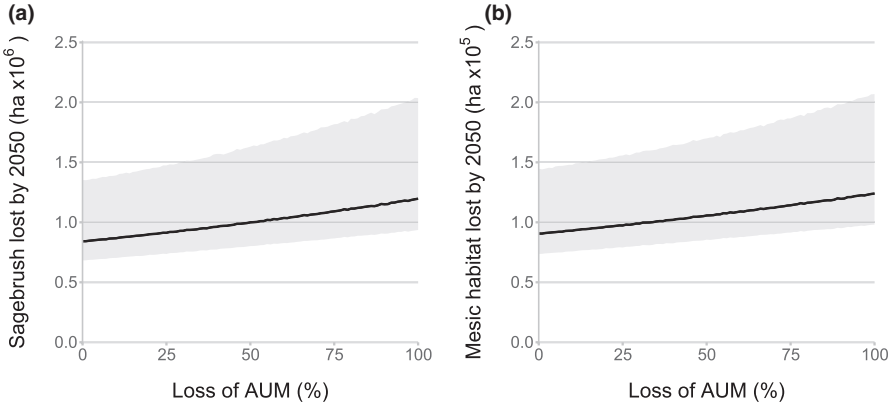

FIGURE 2 Sage-grouse (a) sagebrush ($\times 10^6$ ha) and (b) mesic habitat ($\times 10^5$ ha) predicted to be lost from private land by 2050 with restriction on grazing of lands (0% equates to projections under current grazing rates), cumulative across 151 counties in study region. 95% confidence intervals are represented by grey shading

Bureau (2010). We obtained data on the area of irrigated land in each county from the 2012 USDA Census of Agriculture (USDA, 2012). We estimated the proportion of cropland in a county from Lark et al. (2015) as 1 minus the proportion of stable noncropped land across the time period 2008–2012, to capture both actively cropped areas and fallow or rotational grazing in this control variable. We included an interaction term between proportion of cropland in a county and percent cropland that is irrigated. We calculated road density in each county from the 2000 US Census TIGER/Line Roads dataset. We included sage grouse management zones in the model, aggregated into three regions ("Great Basin", MZ III, IV, V; "Rocky Mountain", MZ I, II, VII; "Washington" MZ VI). We use standard industry equations (Butler et al., 2003) to calculate AUM_i (animal-unit-months in county *i*) from the "normal forage productivity" field of gSSURGO (Soil Survey Staff (gSSURGO), 2016; we use the 2016 release though could not find information on the temporal providence of the "normal forage productivity" field. Our model thus reflects expected productivity that averages inter-year variability in forage productivity). We assume a grazing efficiency of 25%, a requirement of 30lb air-dried feed per day per cow–calf pair, and 30.5 days per month. We assume grazing does not occur on lands under strict protection (IUCN I-VI or GAP Status Code 1 or 2; The Conservation Biology Institute (CBI), 2012). Forage estimates were unavailable for California. The proportion of forage that occurred on public land versus private land in each county ranged from 0% to 100% and averaged 64.1%.

We trained the model on 397 county–year combinations across 139 counties, and predicted across 151 counties. We accounted for zeros (counties with no conversion) by log-transforming the data prior to modelling, $\ln(y_{it}^{RC}/y_{it}^{RR}+0.0001)$. We treated all variables as fixed effects.

2.3 | Predictions of cropland conversion on sage-grouse habitat

We estimated future impacts of cropland conversion on sage-grouse mesic habitat on private land (Donnelly et al., 2016) by allocating the predicted expansion of cropland according to the county-specific proportion of privately owned mesic and nonmesic habitat that was affected by cropland expansion between 2008 to 2011 (Supporting Information Figure S1 and Figure 2). Spatial patterns of how and whether cropland expansion affects habitat differ across this landscape (Supporting Information Figure S2). For instance, in places where groundwater is available for irrigation, cropland expansion often occurs proximate to, but not overlapping mesic habitat. In others, such as where irrigation uses surface water, expansion occurs predominantly on mesic habitat. We make the simplifying assumption that these patterns continue, though changes to water policy and crop technology could shift these patterns. We quantify losses of sage-grouse habitat to direct cropland conversion only, as our models do not allow us to predict the spatial pattern of future cropland at sufficient resolution to map habitat fragmentation, though effects on sage-grouse populations are known to be substantial (Smith et al., 2016). We estimated county predictions of the conversion rate under the assumption that forage productivity and rents remain constant at 2012 values, and areas of habitat converted (mean and confidence intervals) from 10,000 multivariate normal draws (also known as Krinsky–Robb method, a Monte–Carlo simulation) from the heteroskedacity–robust variance–covariance matrix of model coefficients using county data for 2012 (*X*) and the model coefficients $\hat{\theta}$ of each of the 10,000 draws. Counties missing rent data for 2012 were filled by averaging rents across 2008–2014 (37 counties), or where rent estimates for a county were unavailable for any year, by averaging rents across neighbouring counties (six counties). We calculated the cumulative area of rangeland converted to cropland in each county by substituting the 2012 estimates of p^{rc} into

$$y_N^{rc} = \{1-(1-p^{rc})^N\} y_0^r, \quad (4)$$

where N = 38 years (2012–2050). We excluded counties in California from model predictions as forage estimates were not available for this state. We made predictions to 2050. Reported values are for habitat on private land only, we assume habitat on public land cannot be converted to cropland.

Analyses were conducted in R version 3.4.0 (R Core Team, 2017). ArcGIS 10.3 (www.esri.com), Python version 2.7.12 (www.python.org), the GDAL package for Python (http://gdal.org/python), and the

"raster" (Hijmans, 2016), "plyr" (Wickham, 2011) and "rgdal" (Bivand, Keitt, & Rowlingson, 2017) packages for R.

Supporting information includes theoretical foundations for model (Supporting Information Appendix S1), patterns of historical cropland conversion on mesic habitat (Supporting Information Appendix S2), sensitivity analyses of model coefficients (Supporting Information Appendix S3), alternate transformations (Supporting Information Appendix S4), marginal effects (Supporting Information Appendix S5), model predictions (Supporting Information Appendix S6), and mesic predictions (Supporting Information Appendix S7). Code and predictions for each county are available at https://doi.org/10.5063/f13776x1 (Runge et al., 2018).

3 | RESULTS

Rangeland rent had a small but statistically significant effect on the rate of conversion of rangeland to cropland (coef −0.0180 ± 0.0068; $p = 0.0078$; *df* 383; Table 1). The negative coefficient is consistent with expectations, as a higher rangeland rent should decrease range-to-crop transitions and increase the land remaining in range. This estimate was robust to inclusion of different sets of variables in the model (Supporting Information Figure S3 and Supporting Information Table S1). Many of the other variables included in the model had statistically significant coefficient estimates, but are harder to interpret. For example, urbanization variables could be associated with lower range-to-crop and range-to-range transitions, thus having an ambiguous effect on the dependent variable. We investigated constants for the log-transformation in the range 1–1 × 10^{-8}, and found 1 × 10^{-4} gave the least skewed error structure in the residuals (Supporting Information Figure S4). County marginal effects of rangeland rent, percent cropland, and population for 0%–100% loss of AUM are included in Supporting Information Figures S5–S7. We found no evidence of spatial autocorrelation in the model residuals (Supporting Information Figures S8 and S9).

Removing access to public grazing land increased the rate at which natural vegetation (sagebrush and prairie grassland) converted

TABLE 2 Summary of model predictions of conversion of sage-grouse habitat to cropland with restriction on grazing of public lands, averaged across counties. Confidence intervals are shown in parentheses

	Background rate	10% restriction on public land	50% restriction on public land	100% restriction on public land
Annual conversion probability, averaged across counties	0.00116 (0.00091–0.00199)	0.00120 (0.00094–0.00204)	0.00138 (0.00107–0.00239)	0.00165 (0.00125–0.00302)
Increase on background conversion (%)	0 (0–0)	3.06 (2.21–5.68)	16.99 (11.73–33.11)	38.14 (25.18–80.33)
Conversion probability, max of counties in study region	0.00623 (0.0039–0.01476)	0.00643 (0.00417–0.01541)	0.00717 (0.00537–0.01764)	0.00984 (0.00721–0.01959)
Sagebrush converted to cropland, cumulative to year 2050 (km^2)	8,420 (6,824–13,527)	8,671 (7,031–13,905)	9,997 (8,021–16,307)	11,968 (9,363–20,355)
Additional area of sagebrush converted to cropland, cumulative to year 2050 (km^2)		308 (203–664)	1,714 (1,103–3,771)	3,856 (2,362–8,795)
Sagebrush converted to cropland by 2050 (as % of total area)	0.77 (0.62–1.23)	0.79 (0.64–1.27)	0.91 (0.73–1.48)	1.09 (0.85–1.85)
Sagebrush converted to cropland by 2050 (as % of sagebrush on private land)	1.96 (1.59–3.15)	2.02 (1.64–3.24)	2.33 (1.87–3.8)	2.79 (2.18–4.75)
Remaining area of sage-grouse mesic habitat by 2050 (km^2)	37,353 (37,523–36,813)	37,329 (37,502–36,780)	37,202 (37,403–36,560)	37,019 (37,278–36,190)
Sage-grouse mesic habitat lost by 2050 (km^2)	906 (737–1,446)	930 (757–1,479)	1,057 (856–1,699)	1,240 (981–2,069)
Sage-grouse mesic habitat lost by 2050 (%)	2.37 (1.93–3.78)	2.43 (1.98–3.87)	2.76 (2.24–4.44)	3.24 (2.56–5.41)

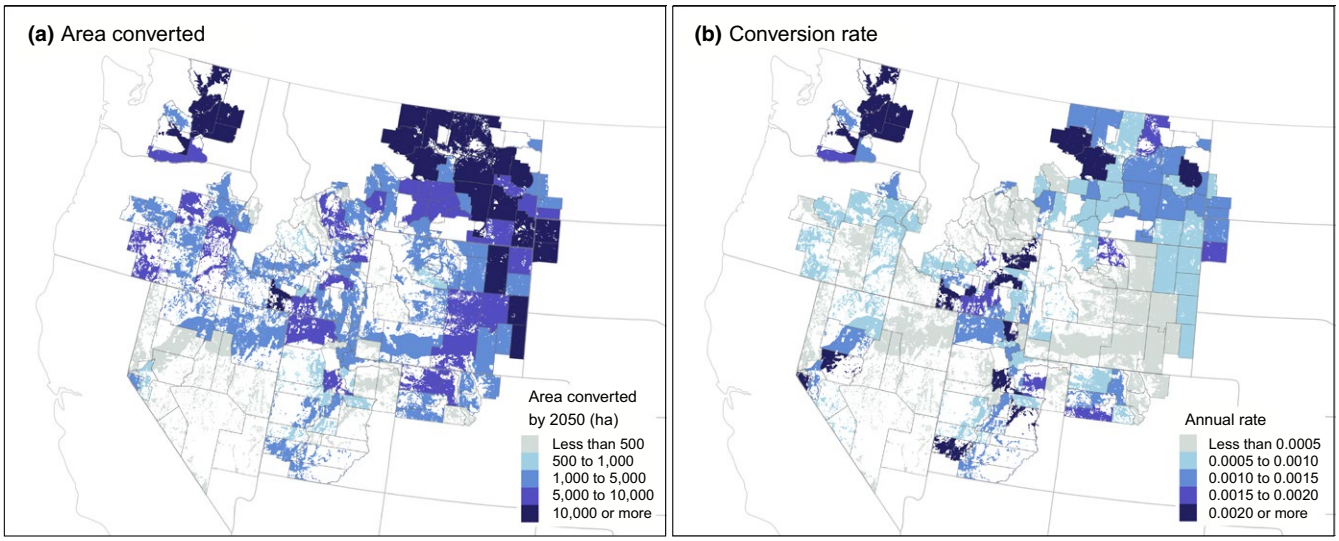

FIGURE 3 Background rates of conversion under current public lands grazing policy (a) Area (ha) of rangeland predicted to be converted to cropland by 2050, no policy (b) annual predicted conversion rate (ha converted to cropland/ha rangeland in county), no policy change

to cropland by 3.06% for a 10% restriction on grazing, up to 38.14% for 100% restriction on grazing (Table 2). This equates to an additional 30,800 ha (10% restriction on grazing), 171,400 ha (50%) or 385,600 ha (100% restriction on grazing) of natural vegetation lost from this landscape by 2050 (Figure 2a). The majority of this conversion would affect sage-grouse mesic habitat, 75% of which occurs on private land and is vital to the species during brood rearing (Donnelly et al., 2016). Under such policy changes, we estimate that between 93,000 (2.43%) and 124,000 ha (3.24%) of sage-grouse mesic habitat held on private land in the study region would be directly lost by 2050 (10% and 100% restriction on grazing respectively, these numbers include background conversion of 90,600 ha; Table 2, Figure 2b).

Under business as usual, 842,000 ha of native vegetation is predicted to be converted to cropland in this study region by 2050 (Figure 3). Thus, our results suggest that a 100% restriction on grazing would result in a 42% increase in area converted to cropland on top of this background conversion. This equates to increasing conversion from 1.96% to 2.79% of remaining sagebrush on private land.

Including baseline conversion, we predict counties in Washington would lose 0%–32% of their mesic habitat with 100% restriction on grazing access. In Montana, we predict that counties would lose 0%–20.4% of their mesic habitat with 100% grazing restriction (Figure 4).

When compared with actual conversion in 2012 the model under-predicted conversion in counties with high conversion rates (Supporting Information Figures S8 and S9 and Supporting Information Table S2). The data we used on conversion rates included areas where alfalfa, which sage-grouse sometimes use as brood-rearing habitat, is occasionally rotated with intensive cropping. Excluding conversion in counties where alfalfa predominates halved the estimated impacts (Supporting Information Table S3 and Supporting Information Figures S10–S13).

4 | DISCUSSION

Conservation actions can have unintended effects on other species and ecosystems (due to unintended ecological interactions; Hansen & DeFries, 2007) and on communities (whether mediated by ecological change, or change in rules around access to resources; Milner-Gulland, 2012). In addition, the responses of individuals or communities to economic opportunities associated with ecosystem-based resources can, as we document here, indirectly affect the species or ecosystem meant to benefit from the conservation action (Fauchald, Hausner, Schmidt, & Clark, 2017; Hausner et al., 2011; Lambin & Meyfroidt, 2010).

Our findings demonstrate that reduced access to forage on western US public land can be expected to increase the conversion of sagebrush rangelands to cropland on private lands, resulting in unintended loss and fragmentation of sage-grouse habitat. Whether benefits of such grazing restrictions would outweigh these losses is speculative. The scientific evidence for the effects of grazing on sage grouse is, perhaps surprisingly, currently obscure. Though chronic overgrazing has multiple detrimental effects on sage-grouse habitat quality, studies suggests only a small fraction of BLM grazing allotments currently fail to meet existing standards due to livestock grazing (Manier et al., 2013; Veblen et al., 2014). There is currently a lack of evidence that grazing levels permitted on public land under existing standards are broadly harmful to sage-grouse populations, or that further reduction or elimination of grazing provides benefits. A recent 10-year experimental study on the effects by Smith et al. (2018) indicated that removal of grazing had no significant effect on sage grouse nest success, when compared with low to moderate levels of rotational grazing. In part, some of the concern over the impacts of grazing on ground-nesting birds such as sage grouse has arisen from studies using statistical methodologies that have since been

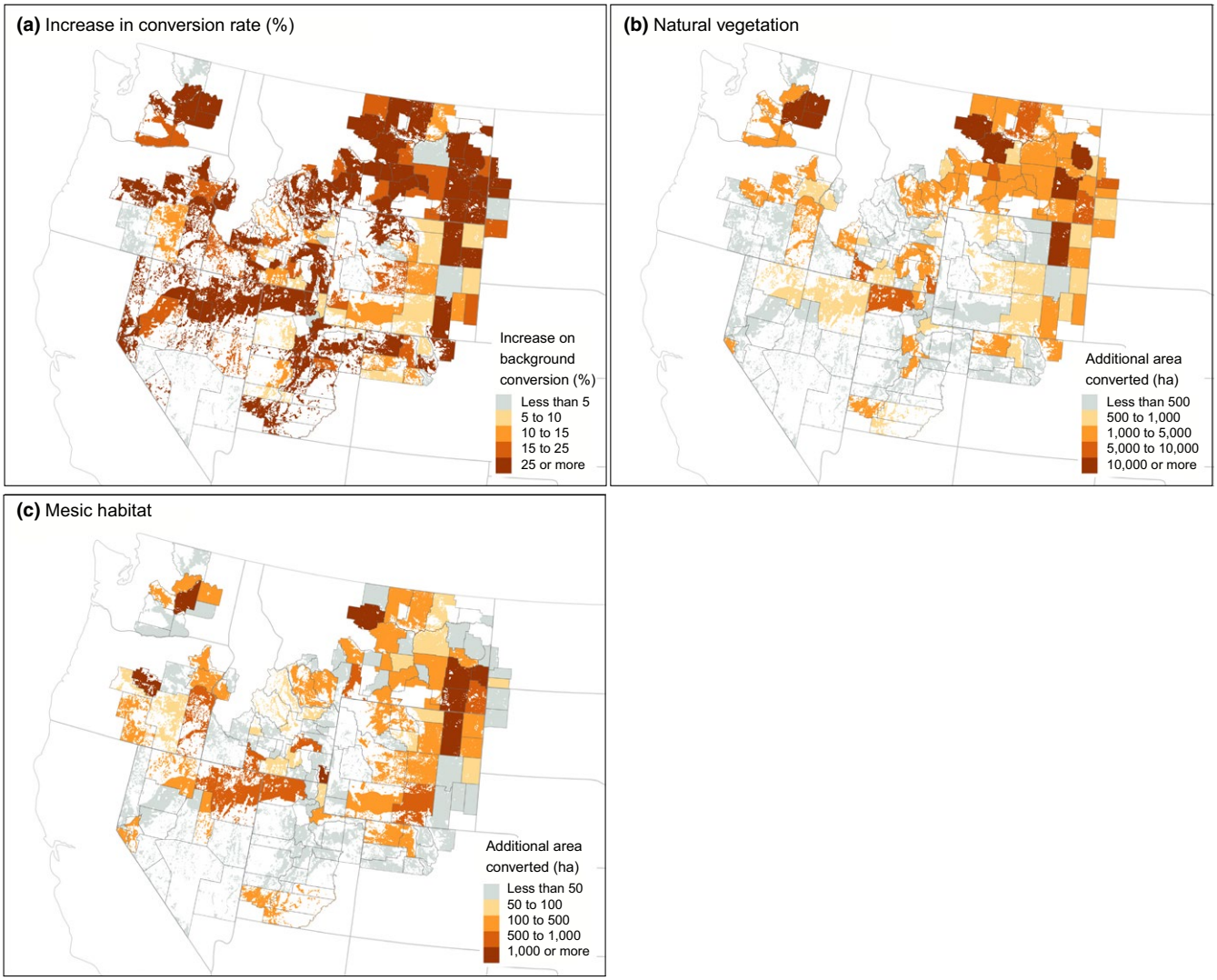

FIGURE 4 Counties predicted to be most affected by changes to public lands grazing policy. Additional area lost to cropland from private land across the western US by 2050 under 100% restriction on grazing of public lands as (a) percent increase on background rate of conversion to public land (b) natural vegetation lost (ha) (c) sage-grouse mesic habitat lost (ha)

questioned (see Gibson, Blomberg, & Sedinger, 2016). As there is, at present, no clear indication of the direction or magnitude of grazing effects on sage grouse, we are unable to determine the net effects of grazing restrictions on sage grouse populations (that is, the difference between any hypothetical population increases arising from grazing restrictions, and the population decreases arising from cropland conversion).

Restrictions on grazing access to public lands could have wide implications for ranching communities and conservation initiatives in the region. Private land owners are an integral part of managing this landscape for biodiversity; being active participants in mesic restoration, fire management and conservation easements (Walker & Janssen, 2002). Conservation policy that acts against the interests and values of ranchers is likely to reduce social and political support for sage-grouse conservation initiatives (Duvall et al., 2017). Work in other landscapes shows that strict restrictions on access to common pool resources are more likely to be revoked or ignored in areas where the economic pressures driving land-use change are great (Mascia & Pailler, 2011), or where communities resist top–down control (Fauchald et al., 2017). In this landscape, previous changes to rules governing access to public lands were met with considerable opposition, propelled by perceptions that such decisions were driven by outside parties, and by opposition to Federal government influence on local land-use decisions (Durrant & Shumway, 2004).

The empirical estimates for the rangeland conversion model support our main hypothesis that rancher decisions are determined, in part, by profits. We note that this is a probabilistic relationship, as there is no fixed opportunity cost at which ranches convert to cropland. This is consistent with the finding of previous research that ranching is as much a cultural identity as it is an economic activity (Gentner & Tanaka, 2002), and ranchers tend to resist switching to farming. Faced with restrictions on access to public rangelands, however, ranchers have a set of choices. In the short term, many may choose to maintain herds on their private lands year round

(Peterson & Coppock, 2001). The increased pressure will likely lead to the loss of or reduction in quality of sage-grouse mesic habitat on private land whether due to increased grazing intensity, or conversion of native vegetation to hay or alfalfa to offset lost forage. In the longer term, the loss of grazing access to public land will increase the likelihood of a ranch "going broke", particularly in the absence of off-farm income or with farm-related debt (Taylor et al., 2005; Torell et al., 2002). More than 30% of farmers and ranchers are 65 or older (USDA, 2012). Decreasing farm income is likely to push many to retire sooner, increasing the rate at which land converts to alternate land uses (Peterson & Coppock, 2001). In addition, while we considered only transitions to cropland, urbanization is also a growing threat in this landscape (Copeland et al., 2013; Hansen et al., 2005) and one-third of ranchers show willingness to sell to developers (Peterson & Coppock, 2001). These changes to ranching communities would be exacerbated in areas where there is a lot of public land, smaller and more isolated communities, and fewer opportunities to supplement economic losses with off-farm income.

Though changes to grazing policy had a relatively small impact on the total amount of sage-grouse habitat, the area of habitat converted to crops understates impacts to sage-grouse populations for several reasons. First, absolute area of habitat loss underestimates population impacts for interior habitat specialists such as sage-grouse (Bender, Contreras, & Fahrig, 1998), and our analysis does not explicitly consider effects of fragmentation on quality of remaining habitat. Smith et al. (2016) estimated that sage-grouse populations are highly sensitive to presence of cropland within a distance of 3.2 km. Using this estimate, we would expect a 1 km^2 crop field to impact habitat quality over a 45 km^2 area and a 16 km^2 crop field to impact approximately 99 km^2. The cumulative area affected would therefore be many times larger than our estimated footprint when these landscape effects are considered.

We find that cropland conversion would disproportionally affect counties in Washington, Montana and Wyoming. Sage-grouse populations in Washington are already low, and further expansion of cropland could jeopardise these populations, though the effect on overall sage-grouse population size would be small. In Montana and Wyoming, where sage-grouse populations are high, expansion could have a higher overall impact on sage-grouse numbers. While lack of suitable soil will limit cropland expansion, such as in Nevada and western Wyoming, cropland is only one of several land-use transitions possible under grazing restrictions. Others land uses not considered our analysis, such as low-density residential development, may have equal or greater impacts on population persistence (Copeland et al., 2013), further adding to the cumulative stressors on this species.

Although we have focussed on sage-grouse, other sagebrush-associated species could be affected by grazing restrictions on public lands or habitat loss on private lands (Lipsey et al., 2015; Rowland, Wisdom, Suring, & Meinke, 2006). For example, some species could benefit from grazing restrictions (e.g. Brennan & Kuvlesky, 2005). Some of these species are valuable game species (e.g. pronghorn, elk) and actual or perceived negative impacts of policy change on populations of these species has potential for negative perception of, and conflict with, conservation initiatives.

In this analysis, we make the implicit assumption that public & private AUMs are perfect substitutes, implying that ranchers faced by a decline in public AUM lose profit from their public lands but that profits from private lands remain unaffected. In fact, public and private AUMs are complementary to some degree, with public AUM used for summer grazing, and private AUM often used to grow hay that is stored to feed wintering stock. Consequently, a loss of AUMs from public lands could result in a loss in profit from private lands. Thus, our estimates of the impact of reductions in public AUM on ranch economics are likely conservative (Torell, Rimbey, Tanaka, Taylor, & Wulfhorst, 2014). Gentner and Tanaka (2002) surveyed 2,000 ranchers with BLM grazing permits and found that less than 20% of surveyed ranchers stated they would continue grazing at current herd sizes if their AUM allocation on public land was reduced by 50%. Loss of summer grazing on public land was particularly influential on the stated likelihood of reducing herd size, intensifying use of private rangeland or conversion to cropland. We assume that AUMs are currently at capacity, though demand for forage has decreased in recent decades (Yahdjian et al., 2015).

Decisions on how much access to grant to resource users of public lands are part of a wider discourse on when, where and how to deliver conservation outcomes that have political and social longevity while succeeding in their objective of maintaining biodiversity (Brooks, Waylen, & Mulder, 2013). We show that restricting grazing on public lands would increase the rate and magnitude of habitat loss to cropland, with negative impacts on a species that is highly sensitive to cropland expansion. Such changes would be likely to have negative effects on ranching communities and could jeopardize efforts to engage these communities in conservation initiatives. Consequently, policy interventions to address threats to their habitat on public land should be constructed to avoid unintended consequences that exacerbate threats to their habitat on private land and reduce community support for sage-grouse conservation.

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Estimates of the area predicted to be converted and conversion rate for each county are available at https://doi.org/doi:10.5063/f13776x1 (Runge et al., 2018).

AUTHORS' CONTRIBUTIONS

C.A.R. collected and analysed the data and led the writing of the manuscript. C.A.R., J.F., A.J.P., A.L. and K.H. designed the methodology. All authors conceived the ideas, contributed critically to the drafts and gave final approval for publication.

DATA ACCESSIBILITY

Data and code are available via the Knowledge Network for Biocomplexity https://doi.org/10.5063/f13776x1 (Runge et al., 2018).

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SUPPORTING INFORMATION

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