Avian species occupancy and richness in montane shrublands subject to grazing:

Final Report

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Bird Conservancy of the Rockies

Connecting people, birds and land

Mission: Conserving birds and their habitats through science, education and land stewardship

Vision: Native bird populations are sustained in healthy ecosystems

Bird Conservancy of the Rockies conserves birds and their habitats through an integrated approach of science, education, and land stewardship. Our work radiates from the Rockies to the Great Plains, Mexico and beyond. Our mission is advanced through sound science, achieved through empowering people, realized through stewardship, and sustained through partnerships. Together, we are improving native bird populations, the land, and the lives of people.

Core Values:

1. **Science** provides the foundation for effective bird conservation.
2. **Education** is critical to the success of bird conservation.
3. **Stewardship** of birds and their habitats is a shared responsibility.

Goals:

1. Guide conservation action where it is needed most by conducting scientifically rigorous monitoring and research on birds and their habitats within the context of their full annual cycle.
2. Inspire conservation action in people by developing relationships through community outreach and science-based, experiential education programs.
3. Contribute to bird population viability and help sustain working lands by partnering with landowners and managers to enhance wildlife habitat.
4. Promote conservation and inform land management decisions by disseminating scientific knowledge and developing tools and recommendations.

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Executive Summary

The U.S. Forest Service is a multi-use land management agency tasked with managing federal lands for various interests, including providing opportunities for livestock grazing while also maintaining ecological integrity and biodiversity. In particular, the Bridger-Teton National Forest (BTNF) allows cattle grazing in montane shrublands within conifer forest landscapes, while also managing impacts to accommodate biodiversity. The Intermountain Region and the BTNF (hereafter Forest Service or FS) partnered with Bird Conservancy of the Rockies (hereafter Bird Conservancy) to monitor birds in two montane shrubland sites differing in cattle grazing: the Upper Gros Ventre (no grazing) and the Upper Green River (some grazing). Site-specific differences in productivity and moisture along with a lack of replication precluded informing grazing impacts on birds by comparing sites directly. We therefore instead focused on analyzing relevant vegetation relationships while controlling for differences between sites.

The FS built upon the Integrated Monitoring in Bird Conservation Regions (IMBCR) program to implement focused bird monitoring within montane shrublands of the Upper Green and Gros Ventre management areas in 2017–2020. Our objectives were to 1) estimate relationships of species occupancy and richness with vegetation attributes potentially impacted by grazing while controlling for site-level differences in productivity and moisture, 2) estimate relationships of avian species occupancy and richness with cattle stocking levels within Upper Green, and 3) assess vegetation relationships with cattle stocking levels within Upper Green to supplement inference of potential grazing effects on birds. For objective 1, we implemented a hierarchical community occupancy model relating avian species occupancy and richness with metrics quantifying herbaceous vegetation, shrub structure, and shrub composition. For objective 2, we implemented a separate model relating bird species occupancy and richness with pasture-specific cattle stocking levels. For objective 3, we related cattle stocking levels with vegetation metrics using Pearson's correlation coefficients.

We found numerous statistically supported species occupancy relationships with vegetation metrics, suggesting potential grazing impacts on birds. Reductions in grass cover and herbaceous height could negatively impact habitat quality for savannah sparrow, vesper sparrow, and Lincoln's sparrow, which showed positive relationships with these attributes. We also found a negative correlation of grass cover with cattle stocking, further suggesting potential impacts on bird species via reductions in grass cover. Considering positive occupancy relationships observed for mountain chickadee and dark-eyed junco with forb cover and herbaceous height and lower values for these metrics at Upper Green (the grazed site), grazing could also contribute to site level differences for these species. We found even stronger bird species relationships with shrub structure and composition, largely reflecting broad species associations with vegetation types, such as American three-toed woodpecker, Steller's jay, and red crossbill with conifer forest and spotted sandpiper, warbling vireo, and yellow warbler with riparian vegetation. Grazing severe enough to affect shrubs could therefore negatively impact various bird species. Because we found more limited covariate relationships with species richness, we suspect any potential grazing impacts would primarily pertain to species composition, although homogenization of the landscape could impact richness and diversity at larger scales. We found no species occupancy relationships with cattle stocking levels, but we suspect limited power and strength of inference with this relatively coarse grazing metric. Further study that includes greater replication and measurements of fine-scale variation in grazing intensity would complement this study to strengthen inference of grazing implications for montane shrubland birds.

Acknowledgements

The analysis detailed in this report represents a collaborative effort between Bird Conservancy of the Rockies and the U.S. Forest Service (Intermountain Region and Bridger-Teton National Forest). We thank Randall Griebel for providing stocking level data and spatial files of pasture boundaries. We thank Randall Griebel, Rema Sadak, and Don DeLong for contributing to the development of this project and feedback on supplemental vegetation data collection and analysis covariates. We thank Nick Van Lanen and Alex Van Boer for overseeing data collection and data entry, and we thank numerous field technicians for collecting the data.

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Introduction

As a multi-use land management agency, the U.S. Forest Service (hereafter USFS) must accommodate various stakeholders and interests (USDA 2012). In particular, the USFS must balance human land uses like logging and grazing with maintenance of ecological integrity and biodiversity. Montane sagebrush and wet meadows (hereafter montane shrublands) embedded within national forests of western North America provide opportunities for cattle grazing, but grazing can impact key components of biodiversity by altering habitat for wildlife. Biodiversity monitoring therefore provides critical information for managers to assess cattle stocking levels in montane shrublands.

Birds represent both a key component of montane shrubland biodiversity and a useful focus for monitoring. Because of their mobility, birds respond quickly to environmental change, including anthropogenic impacts and land management activities. Additionally, bird surveys do not require specialized equipment and can feasibly generate spatially extensive data representing a wide range of species with various life histories, allowing assessment of various components of ecological systems. Montane shrublands represent the confluence of multiple plant communities, including riparian vegetation, open grassy meadows, arid shrub communities, and coniferous forest edges, each of which provides habitat for different bird assemblages. Cattle primarily eat grass, but intensive grazing can also impact forbs and woody vegetation, compromising habitat quality for numerous bird species (Scasta 2014). Monitoring birds therefore complements vegetation monitoring where grazing occurs to assess ecological impacts holistically. Additionally, monitoring can inform avian relationships with vegetation to inform grazing management on USFS lands.

For this study, we conducted bird and vegetation surveys in two management areas: the Upper Green River (hereafter Upper Green) and the Upper Gros Ventre (hereafter Gros Ventre). The Bridger-Teton National Forest (BTNF) allows some grazing in the Upper Green at various stocking levels whereas no grazing was allowed in the Gros Ventre management area contemporaneously with this study. To evaluate potential grazing effects, the Intermountain Region and the BTNF (hereafter Forest Service or FS) worked with Bird Conservancy of the Rockies (hereafter Bird Conservancy) to extend Integrated Monitoring in Bird Conservation Regions (IMBCR) to survey birds at sites representing montane shrublands within these two management areas. Due to a lack of replication and confounding site-specific differences in productivity and moisture, a simple comparison of birds across sites would not have meaningfully informed grazing effects. We therefore evaluated avian relationships with vegetation attributes to inform potential grazing impacts, and examined avian and vegetation relationships with cattle stocking levels within the Upper Green management area.

In consultation with FS partners, Bird Conservancy staff have identified three objectives for this analysis and report:

1. Estimate relationships of avian species occupancy and richness with vegetation attributes potentially impacted by grazing while controlling for site-level differences in productivity and moisture between management areas.

2. Estimate relationships of avian species occupancy and richness with cattle stocking levels within the Upper Green site.

3. Examine vegetation relationships with cattle stocking levels at Upper Green to supplement inference of potential grazing effects on birds.

Methods

Sampling design for avian monitoring

The FS and Bird Conservancy implemented a spatially balanced sampling design built on IMBCR (Stevens and Olsen 2004, Pavlacky et al. 2017) within the Upper Green and Gros Ventre sites. We initially identified at each site an equal number of 1-km^2 grid cell units, each of which consisted of 16 survey points separated by 250 m and located ≥125 m from the grid cell boundary (Pavlacky et al. 2017). At each point within a grid cell, surveyors recorded all bird species seen or heard during six minute morning surveys (Hanni et al. 2018). Due to logistical constraints, surveyors missed surveying either birds or measuring vegetation at some points in some years. We excluded points in years with incomplete data, after which remaining data represented 843 surveys of 266 points within 17 grid cells across both sites, and 448 surveys of 141 points within 9 Upper Green grid cells for analyzing relationships with cattle stocking levels (Table 1, Figure 1).

Surveyors categorized points each year into vegetation types (see "primary habitats" in Hanni et al. 2018), allowing us to examine vegetation composition of the sampling frame. Surveyors categorized a minority of points differently across years, but assuming these cases represent points that straddle >1 vegetation type or do not clearly fit within a single category, we considered these data informative for representing vegetation composition at a coarse level. Sampling units across both management areas primarily consisted of sagebrush dominated shrublands, followed by conifer forest, riparian, meadows, wetlands, and aspen groves (Table 2). Upper Green and Gros Ventre sites shared this overall composition, but vegetation composition at the Upper Green site represented moister and more productive conditions compared to Gros Ventre. Specifically, surveyors classified Upper Green points as shrubland and conifer forest (i.e., relatively dry vegetation types) less frequently (65% of surveys) than did surveyors at Gros Ventre points (74% of surveys). Additionally, surveyors only ever classified Upper Green points as wetlands. In supplemental vegetation surveys, relatively mesic sedge and rush species dominated potential cattle forage (defined as sedges, rushes, or grasses) in 24% of 100-cm^2 quadrats in the Upper Green, whereas sedges or rushes dominated only 6% of Gros Ventre quadrats (see Appendix A). In contrast with Upper Green, the Gros Ventre had not been subject to grazing by domestic livestock for ≥14 years at the time of this study.

Table 1. Sampling units for analysis of avian species occupancy and richness patterns in montane shrublands of the Bridger-Teton National Forest. Survey points were nested within grid cells, which contained 16 points, of which a minimum of six were surveyed in any given year. Point × year occasions represent the total number of point surveys conducted, with each point surveyed no more than once per year. Values in parentheses represent sample sizes for analyzing avian relationships with cattle stocking levels implemented only in the Upper Green management area where grazing occurred.

Sampling unit	***n* (grazing analysis)**	
	Gros Ventre	**Upper Green**
Grid cell	8	9 (9)
Points	125	141 (142)
point × years	389	445 (448)

Table 2. Distribution of point surveys across vegetation types defined by Hanni et al. (2018). Surveyors categorized each point during each survey. Surveyors categorized 32% of survey points differently in different years suggesting these points straddled >1 vegetation type or did not clearly fit into a single vegetation type.

Vegetation type	**% composition (*n* point surveys)**		
	Upper Green	**Gros Ventre**	**Total**
Shrubland[a]	54 (240)	40 (154)	47 (394)
Confier forest[b]	11 (50)	34 (130)	22 (180)
Riparian	18 (78)	23 (89)	20 (167)
Meadow[c]	6 (27)	3 (13)	5 (40)
Wetland	7 (30)	0 (0)	4 (30)
Aspen	4 (20)	1 (2)	3 (22)

[a]Includes sage shrubland (94%) and shrubland (6%) primary habitats defined by Hanni et al. (2018).
[b]Includes lodgepole pine (22%), mixed conifer (13%), spruce-fir (60%), and insect infested (5%) primary habitats defined by Hanni et al. (2018).
[c]Includes primary habitats of grassland (70%) and montane meadow (30%) primary habitats defined by Hanni et al. (2018).

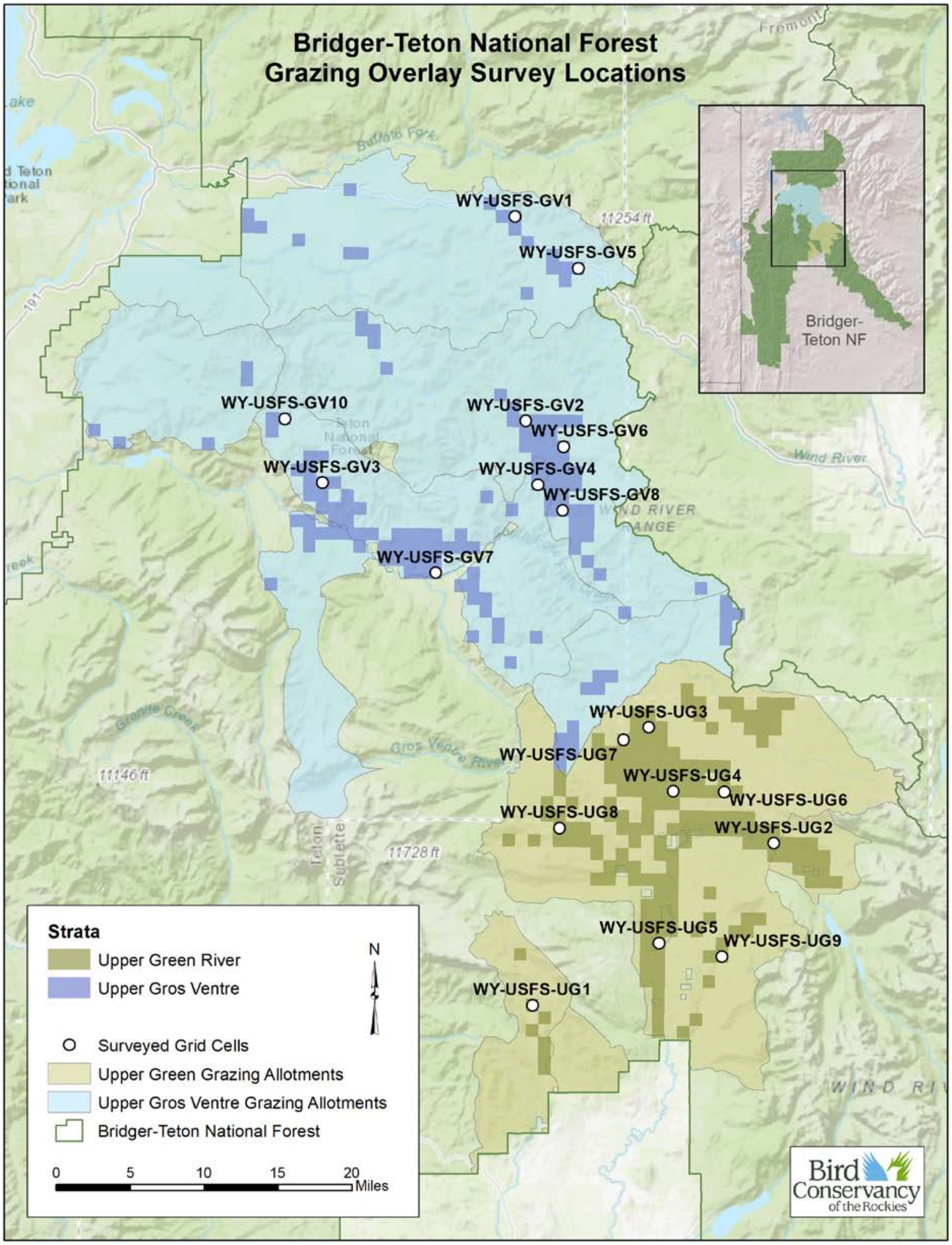

Figure 1. Distribution of surveyed grid cells, the montane shrubland sampling frame from which these cells were drawn following spatially balanced sampling (darker colors), and management areas comprising the study area.

Vegetation and grazing metrics

We compiled vegetation metrics for 50-m radius circular plots centered on survey points following the IMBCR field protocol (Hanni et al. 2018) to relate with bird population and community parameters. We compiled four metrics quantifying herbaceous vegetation, two metrics quantifying shrub structure, and four metrics quantifying shrub species composition (Table 3). All shrub metrics describe woody vegetation < 3 m in height. Canopy (woody vegetation > 3 m height) was not a major vegetation component at our survey points (canopy cover < 10% in 73% of vegetation observations), and we expected minimal grazing impacts on canopy vegetation, so we did not consider canopy metrics in this study.

Shrub composition metrics represented the relative cover of four species groups: arid shrubs, riparian shrubs, berry-producing shrubs, and conifer saplings. These four groups encompassed all species recorded at 98% of survey points, so the relative cover of any one shrub species group complemented (almost) exactly the sum of relative cover values for the other three groups. Multicollinearity would therefore have potentially obscured interpretation of our analysis model had we included all four shrub composition metrics as covariates (Neter et al. 1996:285-295). We therefore only included three of the four shrub composition variables in our analysis of vegetation relationships, treating the fourth as the reference group. We chose to treat arid shrubs as the reference group because excluding arid shrubs reduced variance inflation factors (*sensu* Neter et al. 1996:385-388) for the remaining three shrub composition covariates to VIF < 2. Although not represented explicitly in our analysis of bird monitoring data, we included arid shrubs in all covariate summaries for reference.

IMBCR employs a rapid assessment approach to measure vegetation consisting primarily of ocular estimation to measure vegetation. Considering the susceptibility of ocular estimation to measurement error, we supplemented IMBCR measurements of herbaceous vegetation (i.e., vegetation most impacted by grazing) with additional standardized metrics based on the Daubenmire method (Coulloudon et al. 1999) implemented in 2020. We found strong correspondence between IMBCR metrics and equivalent Daubenmire metrics (Appendix A). Considering this correspondence and having only recorded Daubenmire metrics in 2020, we proceeded with using IMBCR herbaceous vegetation metrics as covariates in our analysis.

The IMBCR vegetation assessment protocol changed slightly starting in 2018 from measuring one metric representing combined coverage of bare ground and litter to measuring these two attributes as separate metrics. Consequently, we lacked values for bare ground in 2017. We fitted a Random Forest model (Cutler et al. 2007) to 2018–2020 data relating bare ground at each point in each year with 1) all other herbaceous vegetation metrics (Table 3), 2) the sum of bare ground and litter cover, and 3) mean bare ground across years. We then applied this model to impute missing 2017 bare ground values.

In addition to vegetation measurements, we obtained cattle stocking data for pastures containing our survey points from BTNF. From these data, we compiled a single metric representing cattle density (i.e., the number of cattle divided by pasture area) multiplied by the number of days grazing was allowed prior to the bird survey date at each point (cattle days; Table 3). As such, cattle days quantified the level of cattle stocking prior to bird surveys in each year. We recognize that cattle grazing can affect vegetation and birds across multiple years, but our pre-survey cattle stocking metric was correlated with an equivalent metric of season-long cattle stocking (r = 0.87) and mean cattle stocking across years (r = 0.91). Thus, we interpreted our metric as a general representation of cattle stocking levels for pastures containing survey points. Nevertheless, we lacked high resolution information on the extent and intensity to which cattle grazed individual survey points within pastures, so our metric of cattle stocking was unavoidably coarse and limited for informing grazing effects on birds.

Table 3. Covariates used in models analyzing avian species occupancy and richness in montane meadows shrublands of the Bridger-Teton National Forest.

Habitat feature	**Covariates (abbrev.; units)**	**Description**
Herbaceous vegetation	Forb cover (ForbCov; %)	Percent cover of forbs
	Grass cover (GrassCov; %)	Percent cover of live grasses
	Herb height (HerbHt; cm)	Average height of live grasses and forbs
	Bare ground (Bare; %)	Percent bare ground
Shrub structure	Shrub cover (ShrubCov; %)	Percent cover of woody vegetation 0.25–<3.0 m in height
	Mean shrub height (ShrubHt; m)	Mean height of woody vegetation 0.25–<3.0 m in height (only defined where shrub cover > 0)
Shrub composition	Arid shrubs (Arid; %)[a]	Percentage of shrub layer composed of arid shrubs (*Artemisia* sp., *Purshia* sp., *Cercocarpus* sp., *Chrysothamnus* sp., *Sarcobatus* sp., *Rhus trilobata*, *Gutierrezia sarothrae*)
	Riparian shrubs (Riparian; %)	Percentage of shrub layer composed of riparian shrubs (*Salix* spp., *Pentaphylloides floribunda*, *Populus tremuloides*, *Rosa* sp., *Betula* sp., *Alnus* sp.)
	Berry shrubs (Berry; %)	Percentage of shrub layer composed of berry-producing shrubs (*Ribes* spp., *Symphoricarpos* sp., *Shepherdia canadensis*, *Rubus* sp., *Prunus virginiana*, *Amelanchier* sp., *Sambucus* sp., *Vaccinium* sp., *Lonicera* sp.)
	Conifer saplings (Conifer; %)	Percentage of shrub layer composed of conifer saplings (*Pinus* spp., *Abies lasiocarpa*, *Picea* spp., *Pseudotsuga menziesii*, *Juniperus communis*)
Grazing	Cattle days (Cattle; [cattle × days] / acre)	Density of cattle (number per acre) occupying the pasture containing the survey point on each day summed across active grazing days prior to the bird survey within the same year
Site	Site (UpperGreen)	Binary factor distinguishing Upper Gros Ventre (0) from Upper Green (1) sampling units
Survey[b]	Day of year (DOY)	Days elapsed since January 1
	Time since sunrise (Time)	Minutes elapsed since sunrise

[a]Arid shrubs represents the reference group for shrub composition covariates, and so was excluded from bird occupancy analysis to avoid problems arising from multicollinearity.
[b]These metrics served as covariates of detection probability.

Data Analysis

For objectives 1 and 2, we estimated relationships of avian species occupancy and richness with vegetation and grazing metrics using multi-species occupancy models (model structure detailed in Appendix B). Mirroring our hierarchically nested sampling units, we extended community models (Dorazio et al. 2006, Iknayan et al. 2014) to estimate occupancy at multiple scales (Mordecai et al. 2011, Pavlacky et al. 2012), but we focused inference at the point level where we had greater sample sizes (see also Latif et al. 2020). We used removal sampling to estimate detection probability within 6-min surveys, so we modeled occupancy and richness at survey points during a snapshot in time (*sensu* Latif et al. 2016). The resulting model included individual occupancy models for each species with information sharing across species to inform estimates for sparsely detected species and ultimately to facilitate species richness estimation. We fitted two models, one that estimated avian relationships with vegetation metrics (hereafter vegetation model) and a separate model that estimated relationships with cattle stocking levels within Upper Green (hereafter grazing model). The vegetation model also included a binary covariate differentiating the two management areas (Table 3) to control for site-specific differences when estimating vegetation relationships. We examined 95% Bayesian credible intervals (BCIs) to infer statistical support for vegetation and grazing relationships, whereby we considered relationships statistically supported if BCIs excluded zero. We estimated species richness by summing occupancy estimates across species, and we assessed statistical support for species richness relationships by examining 95% BCIs for community-level mean covariate relationships (Appendix B). We estimated species richness both for surveyed points (N_{FS}) and predicted richness for the general population of survey points represented by our sample (N_{pred}; described further in Appendix B).

For objective 3, we calculated Pearson's correlation coefficients relating vegetation metrics with our grazing metric quantifying cattle stocking levels. We interpreted negative correlations as potential grazing impacts, which we expected primarily for herbaceous vegetation.

Results

Surveyors recorded detections for 73 bird species in montane shrublands at Upper Green and Gros Ventre sites during the study period. The five most commonly detected species were white-crowned sparrow, Lincoln's sparrow, Brewer's sparrow, American robin, and dark-eyed junco (Appendix C). Survey points represented a range of vegetation conditions and cattle stocking levels that differed notably between sites (Table 4). Specifically, forb cover, herbaceous height, and bare ground were greater at Gros Ventre, whereas grass cover was greater at the Upper Green site. Shrubs were taller, consisted more so of berry-producing shrub species and conifer saplings, and consisted less so of arid shrub species at Gros Ventre compared to Upper Green points.

When analyzing relationships with vegetation, we found statistically supported occupancy relationships for 42 species and with all covariates except shrub height (Figures 2, 3). We found the strongest relationships supported for the most species with shrub composition covariates (34 species), whereas we found weaker relationships supported for fewer species with shrub cover (6 species) and herbaceous vegetation (17 species). These relationships controlled for site-level differences, which were statistically supported for 12 species (six favored Upper Green and six favored Gros Ventre; Figure 4). Posterior median detection probabilities for a 6-min survey ranged 0.38–1.00 (Appendix D).

Table 4. Summaries for covariate values recorded at Gros Ventre and Upper Green points (*n* = number of point surveys). Mann Whitney U *p*-values (*p*) indicate statistical significance of the difference between Gros Ventre and Upper Green for each covariate.

Covariate	Gros Ventre		Upper Green		*p*
	Mean (SD, range)	*n*	Mean (SD, range)	*n*	
ForbCov	16.63 (11.96, 1–70)	389	14.81 (10.45, 0–68)	445	0.04
GrassCov	10.53 (7.64, 1–50)	389	18.27 (10.75, 0–53)	445	<0.01
HerbHt	24.59 (10.13, 6–58)	389	20.06 (9.7, 0–59)	445	<0.01
Bare	30.49 (17.86, 0–86)	389	25.54 (16.71, 0–74)	445	<0.01
ShrubCov	22.73 (15.1, 0–100)	389	21.8 (14.78, 0–90)	445	0.36
ShrubHt	1.02 (0.53, 0–2.75)	389	0.88 (0.52, 0–2.5)	445	<0.01
Arid[a]	43.93 (37.57, 0–100)	389	59.66 (37.89, 0–100)	445	<0.01
Riparian	28.93 (34.32, 0–100)	389	28.48 (33.64, 0–100)	445	0.87
Berry	11.98 (21.86, 0–95)	389	2.45 (9.31, 0–70)	445	<0.01
Conifer	14.7 (27, 0–100)	389	7.07 (21.38, 0–100)	445	<0.01
Grazing	0 (0, 0–0)	389	4.56 (12.14, 0–59.14)	448	0.01

[a]Arid shrubs represent the reference group for shrub composition covariates, and so were excluded from bird occupancy analysis to avoid problems associated with multicollinearity.

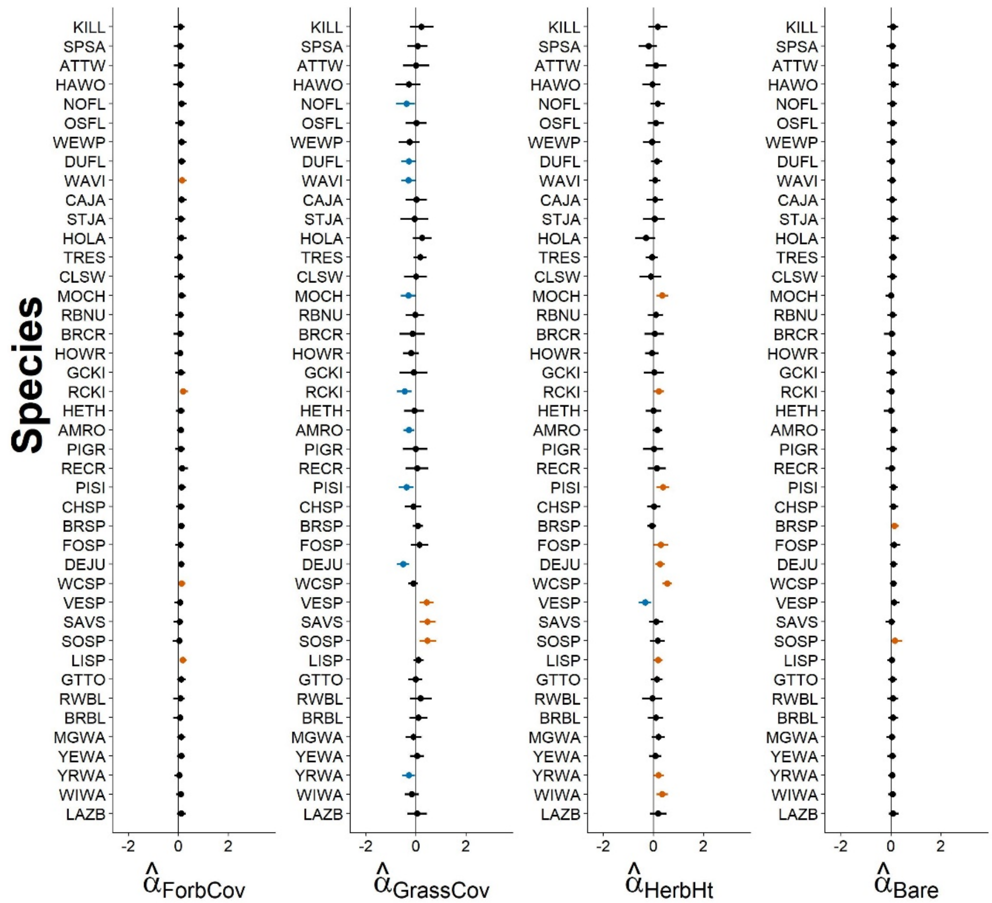

Figure 2. Estimated occupancy relationships (posterior medians and 95% Bayesian credible intervals) with herbaceous vegetation metrics for the 42 species with at least one supported relationship with herbaceous vegetation (here), shrubs (Figure 3), or site covariates (Figure 4; for full bird species names, see Appendix C; for full covariate names and descriptions, see Table 3). Statistically supported positive relationships are orange and supported negative relationships are blue.

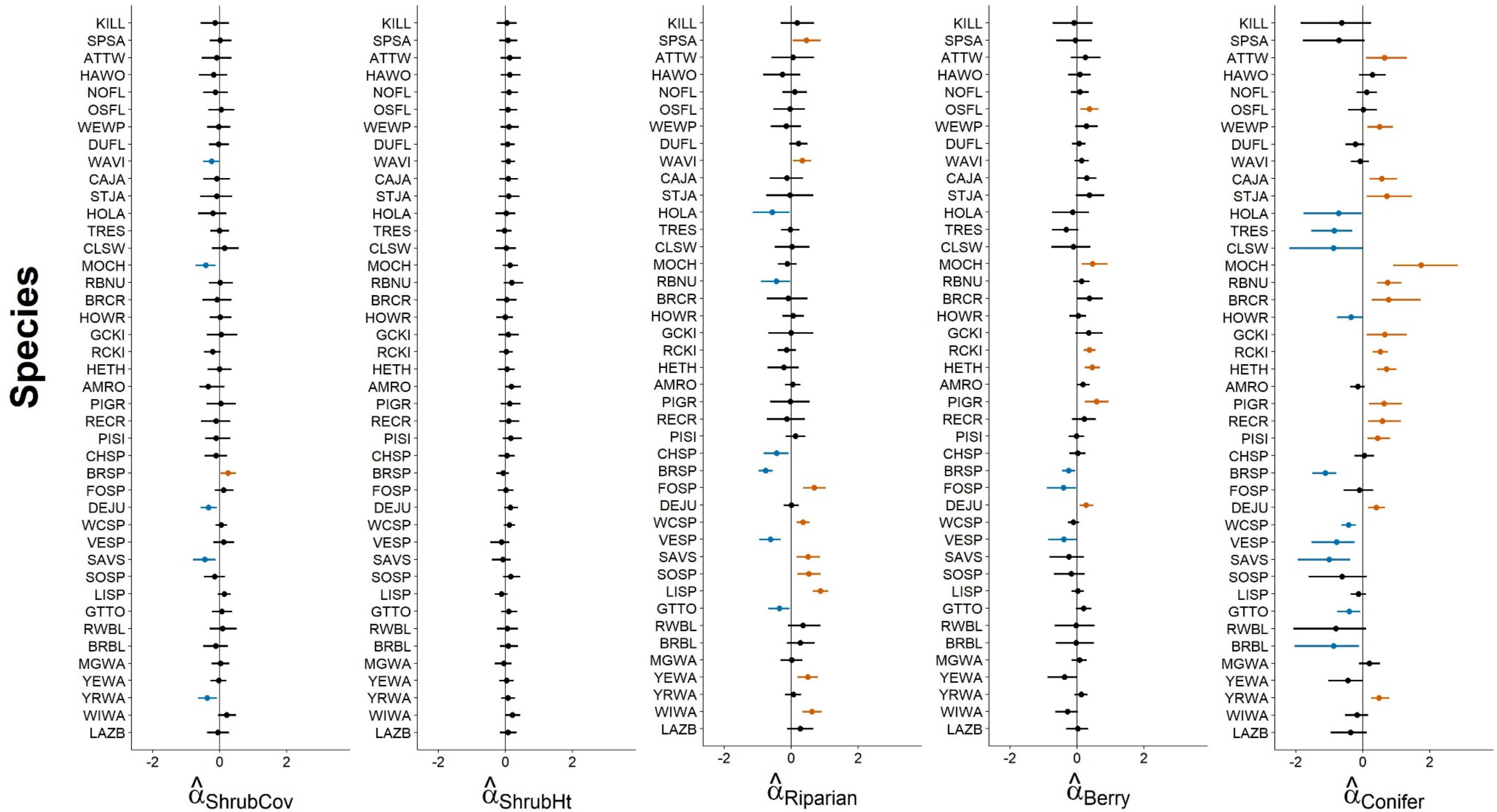

Figure 3. Estimated occupancy relationships (posterior medians and 95% Bayesian credible intervals) with shrub metrics for the 42 species with at least one supported relationship with herbaceous vegetation (Figure 2), shrubs (here), or site covariates (Figure 4; for full bird species names, see Appendix C; for full covariate names and descriptions, see Table 3). Statistically supported positive relationships are orange and supported negative relationships are blue.

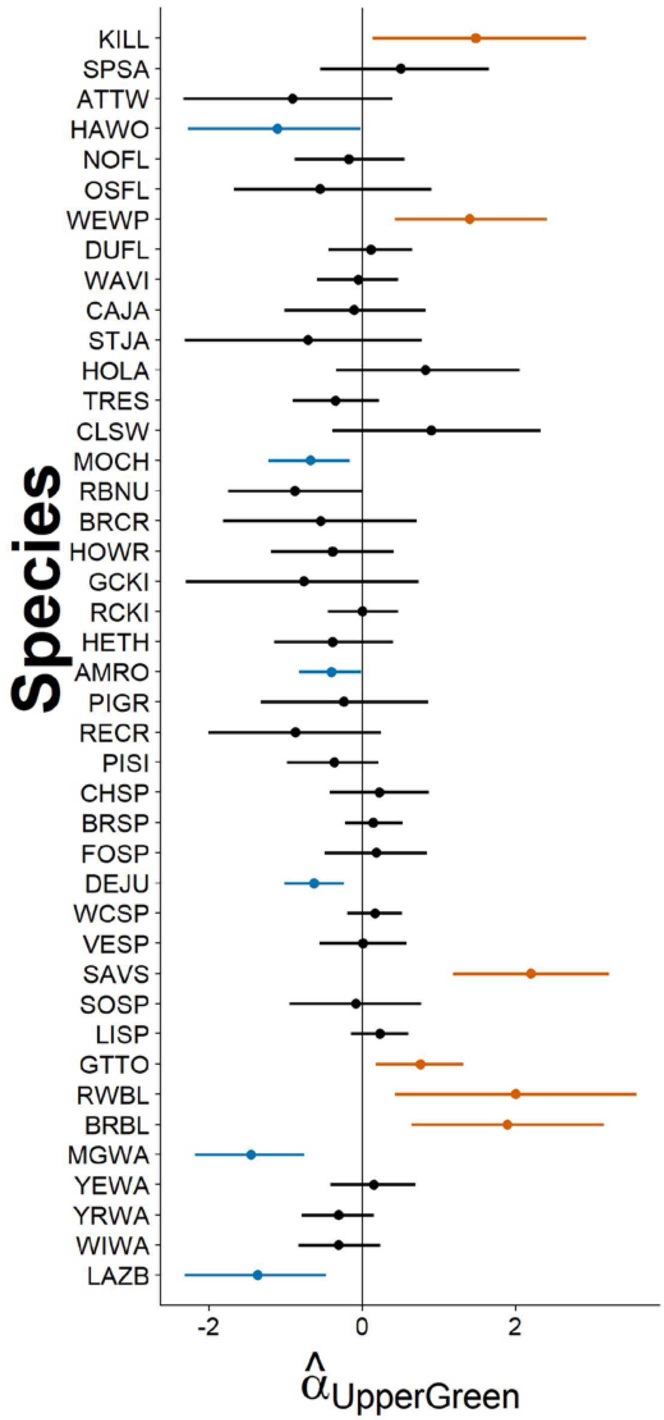

Figure 4. Estimated occupancy relationships (posterior medians and 95% Bayesian credible intervals) with the site covariate for the 42 species with at least one supported relationship with herbaceous vegetation (Figure 2), shrubs (Figure 3), or site (here; for full bird species names, see Appendix C; for full covariate names and descriptions, see Table 3). Statistically supported positive relationships are orange and indicate greater occupancy of Upper Green points, whereas supported negative relationships are blue and indicated greater occupancy of Gros Ventre points.

Of herbaceous vegetation metrics, we primarily found supported occupancy relationships with grass cover and herbaceous height (Figure 2). Vesper sparrow, savannah sparrow, and song sparrow occupancy related positively, whereas occupancy for northern flicker, warbling vireo, mountain chickadee, ruby-crowned kinglet, American robin, pine siskin, dark-eyed junco, and yellow-rumped warbler all related negatively with grass cover (Figure 2, Figure E1 in Appendix E). Occupancy for mountain chickadee, ruby-crowned kinglet, pine siskin, fox sparrow, dark-eyed junco, white-crowned sparrow, yellow-rumped warbler, and Wilson's warbler increased, whereas vesper sparrow occupancy declined with increasing herbaceous height (Figure 2, Figure E2 in Appendix E). We found only six statistically supported relationships for six species with forb cover and bare ground, all of which were positive but relatively weak compared to relationships with other covariates (Figure 2, Figures E3 and E4 in Appendix E).

Of shrub structure and composition metrics, we found the strongest occupancy relationships for the most bird species with relative cover of conifer saplings (Figure 3). Fifteen bird species exhibited positive occupancy relationships and ten species exhibited negative relationships with prominence of conifer sapling in the shrub layer (Figure 3, Figure E5 in Appendix E). Positive relationships with conifer saplings largely reflected species associations with conifer forest (e.g., American three-toed woodpecker, Steller's jay, and red crossbill). Similarly, we found nine positive relationships with riparian shrubs for species typically associated with riparian vegetation (e.g., spotted sandpiper, warbling vireo, and yellow warbler; Figure 3, Figure E6 in Appendix E). Olive-sided flycatcher, mountain chickadee, ruby-crowned kinglet, hermit thrush, pine grosbeak, and dark-eyed junco exhibited positive relationships with prominence of berry-producing shrub species (Figure 3, Figure E7 in Appendix E). Several species exhibited negative relationships with most or all three shrub composition variables, indicating associations with arid shrublands (e.g., Brewer's sparrow and green-tailed towhee) or grasslands (e.g., horned lark and vesper sparrow). Finally, only one species, Brewer's sparrow, exhibited a positive occupancy relationships with shrub cover, whereas five species (warbling vireo, mountain chickadee, dark-eyed junco, savannah sparrow, and yellow-rumped warbler) exhibited negative relationships (Figure 3, Figure E8 in Appendix E).

We estimated mean species richness at surveyed points of N_{FS} = 6.33 (95% BCI: 6.14, 6.59), and we found little evidence for species richness relationships with covariates. The data supported community-wide mean relationships with forb cover and bare ground that deviated from zero (Appendix F), but these relationships translated into limited variation in species richness (Figure 5) reflecting few species-level relationships (Figure 3). The data did not support relationships with vegetation covariates, nor did they support a difference in richness between sites.

Within Upper Green, we found no statistically supported species occupancy relationships with cattle days (Appendix G). Additionally, the 95% credible interval for the average (mean) relationship with cattle days across bird species included zero (Appendix F). We did find a statistically supported negative correlation between grass cover and cattle days, as well as negative and positive correlations of cattle days with arid and conifer shrubs, respectively (Table 5). These correlations were not particularly strong in magnitude, however ($|r| \leq 0.25$). We provide all model estimates and R code for implementing analyses presented here in a data supplement accompanying this report (Appendix H).

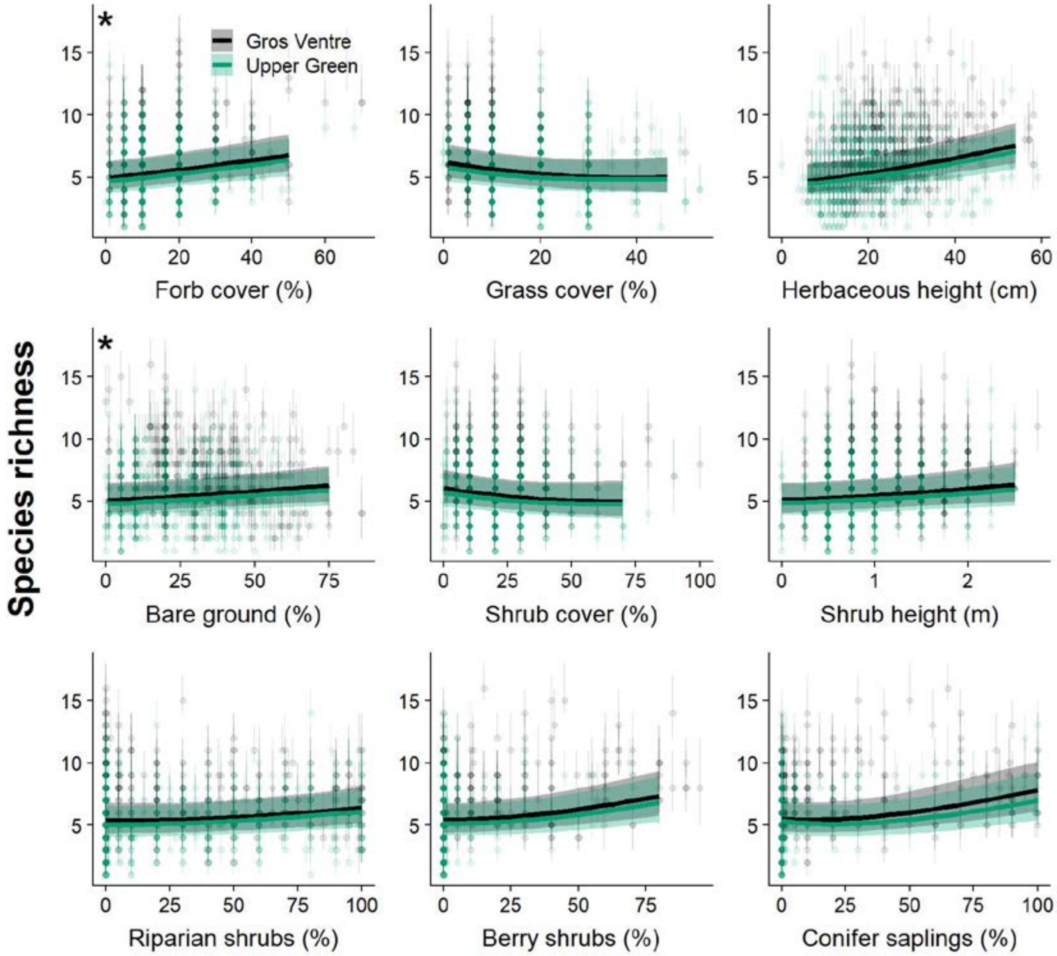

Figure 5. Bird species richness estimates (posterior medians and 95% Bayesian credible intervals) in relation to vegetation covariates. Points and error bars represent finite-sample estimates for surveyed points that account for observed detection histories ($\widehat{N}_{\mathrm{FS}}$), whereas lines and error bands represent predicted richness for hypothetical points ($\widehat{N}_{\mathrm{pred}}$; see Appendix B). Asterisks indicate relationships that were statistically supported (i.e., BCIs for community-level means excluded zero).

Table 5. Pearson's correlation coefficients relating cattle days (cattle stocking density × pre-survey grazing duration) with vegetation covariates.

Vegetation covariate	Pearson's *r*
ForbCov	-0.004
GrassCov	-0.129*
HerbHt	0.005
Bare	-0.036
ShrubCov	0.024
ShrubHt	0.088
Arid	-0.176*
Riparian	0.058
Berry	0.015
Conifer	0.248*

*$p < 0.05$

Discussion

We found numerous species occupancy relationships with vegetation relevant to evaluating potential grazing impacts on birds. Because cattle forage primarily on grass when available (Scosta 2014), we considered avian relationships with herbaceous vegetation as most immediately relevant to potential grazing impacts. Grazing could reduce grass cover and herbaceous height, which could negatively impact habitat quality for species associated with extensive grass cover or tall herbaceous vegetation (e.g., savannah sparrow, vesper sparrow, and Lincoln's sparrow; Ammon 2020, Jones and Cornely 2020, Wheelwright and Rising 2020). Indeed, somewhat lower grass cover in pastures with greater levels of cattle stocking was consistent with our expectation that grazing pressure would reduce grass cover. Occupancy relationships with grass cover and herbaceous height suggested grazing could potentially impact bird species both positively and negatively. Potential impacts via changes in herbaceous vegetation will depend, however, on the extent to which herbaceous vegetation represents key resources for species versus non-essential features coincidentally related with primary resources. For example, reduced grass cover may facilitate foraging by northern flicker for ants (Wiebe and Moore 2020), so grazing could benefit this species. In contrast, occupancy for canopy-associated species, such as mountain chickadee and ruby-crowned kinglet (McCallum et al. 2020, Swanson et al. 2020), may relate only indirectly with herbaceous vegetation. Habitat for such species may be relatively insensitive to grazing. Grazing could also increase bare ground extent or reduce forb cover via cattle foraging (where grass is not available) or trampling, but our results suggest limited potential for effects on birds via forbs and bare ground.

Although grazing primarily impacts herbaceous vegetation, intensive grazing can also impact shrub structure and composition especially when including grazing pressure by other livestock and wild ungulates (Scasta 2014). Considering the relatively strong and numerous species relationships with shrub composition found here, intensive grazing that alters shrub composition could impact a wide range of bird species. We observed relationships largely consistent with broad species-vegetation associations. Species associated with conifer forest related positively with relative cover of conifer saplings (e.g., red crossbill), riparian species related positively with riparian shrubs (e.g., yellow warbler), and arid shrubland species related negatively with non-arid shrubs (e.g., Brewer's sparrow). The extent to which particular attributes of shrub structure and composition measured here represent key foraging or nesting resources will inevitably depend on the species of interest. If grazing pressure from domestic and wild ungulates is severe enough to alter shrub structure and/or composition, however, our results suggest negative impacts to a wide range of bird species could follow.

Despite numerous individual species occupancy relationships, we found relatively limited relationships of bird species richness with herbaceous and woody vegetation. Thus, at a relatively fine scale (125m radius unit) and at grazing levels represented here, we might not expect cattle grazing to appreciably impact bird species richness. Nevertheless, we see substantial potential for grazing to alter bird species composition even if grazing only impacts herbaceous vegetation. Additionally, considering the numerous and varied species relationships with shrub composition, heterogeneity across landscapes appears central for supporting the full array of bird species associated with montane shrublands. Thus, any grazing impacts that result in homogenization of the landscape could negatively impact landscape-scale bird species richness even if local-scale richness remains unaffected.

We found substantial differences in both bird species composition and vegetation structure and composition between the Upper Green and Gros Ventre sites. Although only Upper Green was subject to grazing, we measured greater grass cover and less bare ground on average at Upper Green likely reflecting greater moisture and productivity rather than exposure to grazing. Grazing could contribute to lower herbaceous height and lower forb cover measured at Upper Green (although cattle forage primarily on grass rather than forbs; Scasta 2014). Grazing could in turn contribute to lower occupancy by mountain chickadee and dark-eyed junco at Upper Green if positive relationships with herbaceous height reflect direct relationships with key resources. Other species exhibiting relationships with herbaceous height or forb cover did not exhibit differences in occupancy between sites consistent with potential grazing effects. Potential contributions of either moisture/productivity or grazing to site-level differences in shrub structure and composition were not clear. Regardless, most differences in bird species composition between sites likely reflected differences in factors other than cattle grazing. For example, greater occupancy of Upper Green points by red-winged blackbirds likely reflects the presence of wetlands not found at Gros Ventre. We therefore needed the binary site effect in our analysis model to distinguish avian relationships with vegetation from broader site-specific differences.

Study limitations

Although we failed to document avian relationships with grazing, our ability to infer grazing effects was limited. Due to confounding differences in moisture and productivity, we could not fully leverage comparison across sites to evaluate grazing effects, and within the grazed Upper Green site, our grazing metric was restricted to coarse-level variation in cattle stocking. Controlled experimentation would be ideal for estimating grazing effects and isolating domestic versus wild ungulate grazing. Even without experimentation, however, finer resolution data on actual grazing intensity (i.e., utilization of herbaceous material) would support stronger inference of grazing effects on birds. Surveying more grazed and ungrazed sites across a broad gradient in moisture and productivity (i.e., replication) would also strengthen inference.

Literature Cited

Ammon, E. M. 2020. Lincoln's Sparrow (*Melospiza lincolnii*), version 2.0.*in* A. F. Poole and F. B. Gill, editors. The Birds of the World. Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bow.linspa.01.

Coulloudon, B., K. Eshelman, J. Gianola, N. Habich, L. Hughes, C. Johnson, M. Pellant, P. Podborny, A. Rasmussen, B. Robles, P. Shaver, J. Spehar, and J. Willoughby. 1999. Sampling vegetation attributes. Interagency Technical Reference 1734-4.

Cutler, D. R., T. C. Edwards Jr., K. H. Beard, A. Cutler, K. T. Hess, J. Gibson, and J. J. Lawler. 2007. Random forests for classification in ecology. Ecology **88**:2783-2792.

Dorazio, R. M., J. A. Royle, B. Söderström, and A. Glimskär. 2006. Estimating species richness and accumulation by modeling species occurrence and detectability. Ecology **87**:842-854.

Hanni, D. J., C. M. White, N. J. V. Lanen, J. J. Birek, J. M. Berven, and M. F. McLaren. 2018. Integrated Monitoring in Bird Conservation Regions (IMBCR): Field protocol for spatially-balanced sampling of land bird populations. Unpublished report. Bird Conservancy of the Rockies, Brighton, Colorado, USA.

Iknayan, K. J., M. W. Tingley, B. J. Furnas, and S. R. Beissinger. 2014. Detecting diversity: emerging methods to estimate species diversity. Trends in Ecology & Evolution **29**:97-106.

Jones, S. L., and J. E. Cornely. 2020. Vesper Sparrow (*Pooecetes gramineus*), version 1.0.*in* A. F. Poole and F. B. Gill, editors. Birds of the World. Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bow.vesspa.01.

Latif, Q. S., M. M. Ellis, and C. L. Amundson. 2016. A broader definition of occupancy: Comment on Hayes and Monfils. The Journal of Wildlife Management **80**:192-194.

Latif, Q. S., J. S. Ivan, A. E. Seglund, D. L. Pavlacky, and R. L. Truex. 2020. Avian relationships with bark beetle outbreaks and underlying mechanisms in lodgepole pine and spruce-fir forests of Colorado. Forest Ecology and Management **464**:118043.

McCallum, D. A., R. Grundel, and D. L. Dahlsten. 2020. Mountain Chickadee (*Poecile gambeli*), version 2.0.*in* A. F. Poole and F. B. Gill, editors. The Birds of the World. Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bow.mouchi.01.

Mordecai, R. S., B. J. Mattsson, C. J. Tzilkowski, and R. J. Cooper. 2011. Addressing challenges when studying mobile or episodic species: hierarchical Bayes estimation of occupancy and use. Journal of Applied Ecology **48**:56-66.

Neter, J., M. H. Kutner, C. J. Nachtsheim, and W. Wasserman. 1996. Applied linear statistical models. Times Mirror Higher Education Group, Inc.

Pavlacky, D. C., J. A. Blakesley, G. C. White, D. J. Hanni, and P. M. Lukacs. 2012. Hierarchical multi-scale occupancy estimation for monitoring wildlife populations. The Journal of Wildlife Management **76**:154-162.

Pavlacky, D. C., Jr., P. M. Lukacs, J. A. Blakesley, R. C. Skorkowsky, D. S. Klute, B. A. Hahn, V. J. Dreitz, T. L. George, and D. J. Hanni. 2017. A statistically rigorous sampling design to integrate avian monitoring and management within Bird Conservation Regions. PLoS ONE **12**:e0185924.

Scasta, D. 2014. Dietary composition and conflicts of livestock and wildlife on rangeland. University of Wyoming Extension Report B-1260.

Stevens, D. L., and A. R. Olsen. 2004. Spatially balanced sampling of natural resources. Journal of the American Statistical Association **99**:262-278.

Swanson, D. L., J. L. Ingold, and G. E. Wallace. 2020. Ruby-crowned Kinglet (*Regulus calendula*), version 1.0.*in* A. F. Poole, editor. The Birds of the World. Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bow.ruckin.01.

USDA. 2012. United States Department of Agriculture (USDA). 2012. National Forest System land management planning. Federal Register 77:21162 - 21276.

Wheelwright, N. T., and J. D. Rising. 2020. Savannah Sparrow (*Passerculus sandwichensis*), version 1.0.*in* A. F. Poole, editor. Birds of the World. Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bow.savspa.01.

Wiebe, K. L., and W. S. Moore. 2020. Northern Flicker (*Colaptes auratus*), version 2.1.*in* P. G. Rodewald, editor. The Birds of the World. Cornell Lab of Ornithology, Ithaca, NY, USA. https://doi.org/10.2173/bow.norfli.01.

Appendix A

To address concerns regarding the repeatability and accuracy of ocular ground cover metrics recorded as part of the IMBCR protocol, the FS commissioned supplemental measurements in 2020 following a protocol based on the Daubenmire method (Coulloudon et al. 1999). The supplemental protocol involved measuring ground cover within a series of 10 50×20cm quadrats placed systematically within 50 m of each survey point. Surveyors recorded three metrics in each quadrat: 1) percent bare ground, 2) percent herbaceous cover (forbs and grasses), and 3) mean grass height. We calculated mean values across quadrats for each survey point, and then compared these supplemental values with equivalent values recorded using ocular estimation as part of the standard IMBCR field protocol. We found moderate to strong correlations between supplemental metrics and their equivalent IMBCR counterparts (Figure A1). Bare ground was least correlated, so assuming greater repeatability with the supplemental protocol, IMBCR bare ground measurements may be least reliable of the three metrics considered here. Nevertheless, correlations of IMBCR with supplemental metrics suggest IMBCR ground cover metrics are generally informative. Additionally, IMBCR metrics represented the entire 50 m radius circle (*contra* a sample of quadrats with the supplemental protocol) and surveyors recorded IMBCR metrics with every bird survey across all four years of monitoring. Considering the strong potential for ground cover to vary among years, we decided to use the standard IMBCR vegetation metrics as covariates when analyzing bird species occupancy and richness.

In addition to the three metrics summarized above, surveyors also recorded height of the dominant forage species (defined as grasses, sedges, or rushes) for each quadrat as part of the supplemental vegetation protocol. Surveyors identified 38 species recorded as dominating at least one quadrat (Table A1).

Coulloudon, B., K. Eshelman, J. Gianola, N. Habich, L. Hughes, C. Johnson, M. Pellant, P. Podborny, A. Rasmussen, B. Robles, P. Shaver, J. Spehar, and J. Willoughby. 1999. Sampling vegetation attributes. Interagency Technical Reference 1734-4.

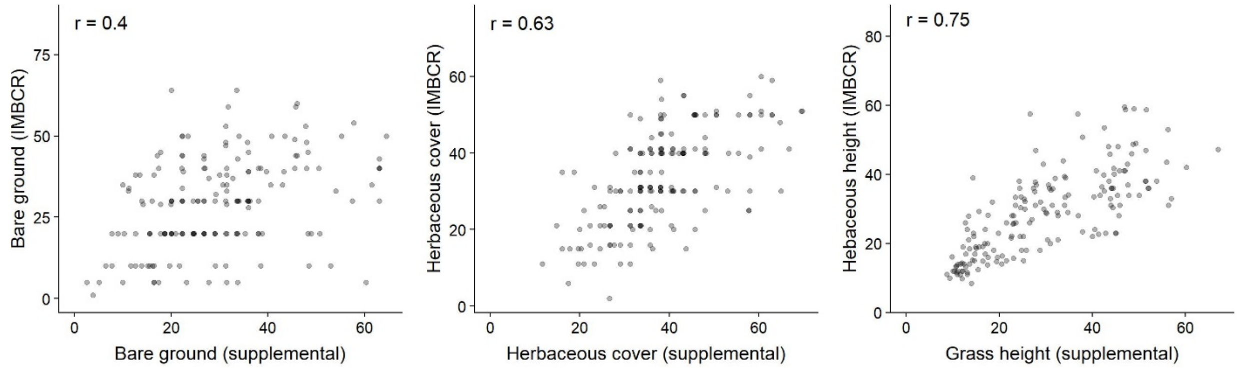

Figure A1. Scatterplots relating IMBCR ground cover metrics with equivalent metrics recorded using a supplemental Daubenmire-based protocol implemented in 2020. Pearson's correlation coefficients (*r*) are reported in the upper left of each plot. The right-hand panel compares herbaceous vegetation height overall (i.e., height of forbs and grasses) measured with IMBCR versus grass height measured via the supplemental protocol.

Table A1. Potential cattle forage species (grasses, forbs, or rushes) recorded as dominating quadrats in supplemental ground cover measurements, along with the percent of quadrats that each species dominated by site.

Common name	**Taxonomic name**	**Dominance (percent quadrats)**	
		Gros Ventre	**Upper Green**
Alpine Fescue	*Festuca brachyphylla*	0.1	0
Arrowgrass	*Triglochin maritima*	0.4	0
Baltic Rush	*Juncus arcticus*	0.7	1.8
Basin Wildrye	*Leymus cinereus*	0.5	0
Bluebunch Wheatgrass	*Pseudoroegneria spicata*	0.8	0
Bluegrass spp.	*Poa spp.*	2.1	0.8
Bottlebrush Squirreltail	*Elymus elymoides*	0.1	0
Cusick's Bluegrass	*Poa cusickii*	0.1	0.8
Foxtail spp.	*Alopecurus spp.*	0	0.1
Green Needlegrass	*Nassella viridula*	2	0
Idaho Fescue	*Festuca idahoensis*	13.5	41.4
Idaho Fescue	*Festuca idahoensis*	0.1	0
Indian Ricegrass	*Achnatherum hymenoides*	0.3	0
Interior Bluegrass	*Poa nemoralis interior*	0.2	0
Kentucky Bluegrass	*Poa pratensis*	7.7	3.8
Kentucky Bluegrass	*Poa pratensis*	0.2	0
King Spikefescue	*Leucopoa kingii*	1.1	0.8
Mountain Brome	*Bromus marginatus*	0.3	3.9
Mutton Bluegrass	*Poa fendleriana*	0.4	4.2
Nebraska Sedge	*Carex nebrascensis*	1.6	8.2

Needle-and-thread	*Hesperostipa comata*	0.3	0
Needlegrass spp.	*Achnatherum spp.*	0	0.8
Orchardgrass	*Dactylis glomerata*	0.1	0
Prairie Junegrass	*Koeleria macrantha*	0.5	0
Purple Oniongrass	*Melica spectabilis*	0	5
Purple Three-Awn	*Aristida purpurea*	1.3	0
Sandberg Bluegrass	*Poa secunda*	27.5	5.8
Sedge spp.	*Carex spp.*	0.1	4.5
Slender Wheatgrass	*Elymus trachycaulus*	0.6	0.9
Smooth Brome	*Bromus inermis*	2.2	0
Thickspike Wheatgrass	*Elymus lanceolatus*	0.3	0
Threadleaf Sedge	*Carex filifolia*	1.5	3.9
Timber Oatgrass	*Danthonia intermedia*	0.1	0
Timothy	*Phleum spp.*	0.5	0
Tufted Hairgrass	*Deschampsia cespitosa*	4.3	4.4
Water Sedge	*Carex aquatilis*	1.8	5.5
Western Wheatgrass	*Pascopyrum smithii*	4.9	0
Wheeler's Bluegrass	*Poa wheeleri*	0	1.4
Unknown Grass	--	11.9	2.4
No Grass	--	9.7	0

Appendix B

We analyzed bird occupancy using a multispecies occupancy model (Dorazio et al. 2010), with a hierarchical multiscale parameterization to match our sampling design (Mordecai et al. 2011, Pavlacky et al. 2012). We treated detection data, **y**, as representing 4 dimensions; y_{ijkt} = 1 indicates species *i* (*i* = 1, …, *M*; *M* = 111) was detected at point *j* (*j* = 1, …, *J*; *J* = 16) within grid cell *k* (*k* = 1, …, *K*; *K* = 17 grid cells) in year *t* (*t* = 1, …, *T*; *T* = 4). We used removal sampling to estimate detectability (Rota et al. 2009). We compiled an array, **R**, that paralleled **y** but whose elements indicated time to detection ($r_{ijkt} \in \{1,2, \ldots, 6\}$ when y_{ijkt} = 1, or r_{ijkt} = 6 when y_{ijkt} = 0). We modeled data generation as

$$y_{ijkt}|u_{ijkt} \sim Binomial(r_{ijkt}, p_{ijkt} \times u_{ijkt}),$$

where p_{ijkt} is the probability of detecting species *i* during a one-minute interval given occupancy of point *j* in grid cell *k* in year *t*. We modeled point occupancy as

$$u_{ijkt}|z_{ikt} \sim Bernoulli(\theta_{ijkt} \times z_{ikt}),$$

where θ_{ijkt} is the point occupancy probability for species *i* given grid cell *k* was occupied in year *t*. We modeled grid cell occupancy as

$$z_{ikt}|w_i \sim Bernoulli(\psi_{ikt} \times w_i),$$

where ψ_{ikt} is the grid cell occupancy probability in year *t* for species *i* given that species *i* belonged to the super community for the study area. Finally, we modeled whether species *i* belonged to the super community as $w_i \sim Bernoulli(\Omega)$.

We modeled point occupancy probability as a logit-linear function of vegetation covariates (vegetation model) or cattle days (grazing model; see Table 2),

$$logit(\theta_{ijkt}) = \alpha_{0,i} + \alpha_{dev,it} + \boldsymbol{\alpha_i} \times \boldsymbol{X_{jkt}},$$

where $\alpha_{0,i}$ is the logit-linear species-specific mean occupancy, $\alpha_{dev,it}$ is the deviation from the mean in year *t*, and $\boldsymbol{\alpha_i}$ is a vector of logit-linear relationships with covariates $\boldsymbol{X_{jkt}}$. We modeled variation in grid cell occupancy probability solely as a function of species,

$$logit(\psi_{ikt}) = \beta_{0,i},$$

where $\beta_{0,i}$ is the logit species-specific grid cell occupancy probability. We estimated intercept terms ($\alpha_{0,i}$ and $\beta_{0,i}$) and covariate relationships ($\boldsymbol{\alpha_i}$) as a species-specific normal random effects, with yearly deviations ($\alpha_{dev,it}$) governed by an additional normal random effect with mean = 0.

We modeled detectability as

$$logit(p_{ijkt}) = \zeta_{0,i} + \boldsymbol{\zeta_i} \times \boldsymbol{X_{jkt}},$$

where $\zeta_{0,i}$ is the logit-linear species-specific intercept for detectability and $\boldsymbol{\zeta_i}$ is a vector of covariate relationships. We modeled detectability with quadratic effects of day of year (DOY + DOY^2) and time since sunrise (Time + $Time^2$), and either two linear effects of shrub cover and height (vegetation model) or one linear effect of cattle days (grazing model). We modeled the detectability intercept parameter and all detectability covariate relationships as species-specific normal random effects.

We inferred species richness relationships by plotting predicted richness (posterior median and 95% credible intervals) in relation to covariates. We predicted species richness for hypothetical (un-surveyed) points representing the sampled population by summing unconditional point occupancy probabilities:

$$\hat{N}_{\text{pred,j}kt} = \sum_{i=1}^{M} \Omega \times \psi_{ikt} \times \theta_{ijkt}.$$

Additionally, we estimated richness for surveyed points using a finite-sample estimator:

$$\hat{N}_{\text{FS,j}kt} = \sum_{i=1}^{M} u_{ijkt}.$$

We implemented data augmentation to fully correct for imperfect detection when predicting species richness. Thus, we set M = 111 to represent the entire list of species comprising the potential super-community for the study area. This species list includes the 73 species detected during this study and an additional 38 species detected at any time during background IMBCR monitoring within USFS Region 4. We excluded from this list species not readily detected with passive point count surveys (i.e., raptors, owls, grouse, cranes, and water birds) and species only detected as migrants that do not breed in the study area.

We sampled posterior parameter distributions for this model using JAGS v.4 (Plummer 2003) programmed from R (Meredith 2020). We used independent noninformative priors for all parameters (for model code and data, see Appendix C). We ran three parallel MCMC chains of length 511,000, burn in = 31,100, and thinning = 100 to sample posterior distributions, after which we verified $\hat{R} \leq 1.1$ for all parameters (Gelman and Hill 2007).

References

Dorazio, R. M., M. Kéry, J. A. Royle, and M. Plattner. 2010. Models for inference in dynamic metacommunity systems. Ecology 91:2466-2475.

Gelman, A., and J. Hill 2007. Data analysis using regression and multilevel/ hierarchical models. Cambridge University Press, New York, New York, USA.

Meredith, M. 2020. saveJAGS: Run JAGS and Regularly Save Output to Files. R package version 0.0.4.

Mordecai, R. S., B. J. Mattsson, C. J. Tzilkowski, and R. J. Cooper. 2011. Addressing challenges when studying mobile or episodic species: hierarchical Bayes estimation of occupancy and use. Journal of Applied Ecology 48:56-66.

Pavlacky, D. C., Jr., J. A. Blakesley, G. C. White, D. J. Hanni, and P. M. Lukacs. 2012. Hierarchical multi-scale occupancy estimation for monitoring wildlife populations. Journal of Wildlife Management 76:154-162.

Plummer, M. 2003. JAGS: A program for analysis of Bayesian graphical models using Gibbs sampling. In Proceedings of the 3rd International Workshop on Distributed Statistical Computing (DSC 2003), March 20-22, Vienna, Austria.

Rota, C. T., R. J. Fletcher Jr, R. M. Dorazio, and M. G. Betts. 2009. Occupancy estimation and the closure assumption. Journal of Applied Ecology **46**:1173-1181.

Appendix C.

Bird species detected during surveys of montane shrublands in Upper Green River and Upper Gros Ventre management areas within the Bridger-Teton National Forest. Max detections = number of point surveys during which the species was recorded. The vegetation model was fitted to data from both management units, whereas the grazing model only included data from Upper Green where grazing occurred.

Species (*taxonomic name*)	**BirdCode**	**Vegetation model detections (max = 834)**	**Grazing model detections (max = 448)**
Common Nighthawk (*Chordeiles minor*)	CONI	2	1
Broad-tailed Hummingbird (*Selasphorus platycercus*)	BTHU	4	2
Calliope Hummingbird (*Selasphorus calliope*)	CAHU	1	1
Killdeer (*Charadrius vociferus*)	KILL	18	16
Wilson's Snipe (*Gallinago delicata*)	WISN	19	8
Spotted Sandpiper (*Actitis macularius*)	SPSA	23	14
Williamson's Sapsucker (*Sphyrapicus thyroideus*)	WISA	1	1
Red-naped Sapsucker (*Sphyrapicus nuchalis*)	RNSA	19	9
American Three-toed Woodpecker (*Picoides dorsalis*)	ATTW	10	2
Hairy Woodpecker (*Dryobates villosus*)	HAWO	18	3
Northern Flicker (*Colaptes auratus*)	NOFL	50	23
Olive-sided Flycatcher (*Contopus cooperi*)	OSFL	28	6
Western Wood-Pewee (*Contopus sordidulus*)	WEWP	33	28
Willow Flycatcher (*Empidonax traillii*)	WIFL	9	3
Least Flycatcher (*Empidonax minimus*)	LEFL	1	1
Hammond's Flycatcher (*Empidonax hammondii*)	HAFL	24	6

Dusky Flycatcher (*Empidonax oberholseri*)	DUFL	116	51
Cordilleran Flycatcher (*Empidonax occidentalis*)	COFL	3	0
Warbling Vireo (*Vireo gilvus*)	WAVI	103	47
Canada Jay (*Perisoreus canadensis*)	CAJA	33	15
Steller's Jay (*Cyanocitta stelleri*)	STJA	7	1
Clark's Nutcracker (*Nucifraga columbiana*)	CLNU	26	5
Black-billed Magpie (*Pica hudsonia*)	BBMA	2	0
Common Raven (*Corvus corax*)	CORA	6	5
Horned Lark (*Eremophila alpestris*)	HOLA	23	23
Bank Swallow (*Riparia riparia*)	BANS	1	0
Tree Swallow (*Tachycineta bicolor*)	TRES	109	54
Violet-green Swallow (*Tachycineta thalassina*)	VGSW	15	3
Northern Rough-winged Swallow (*Stelgidopteryx serripennis*)	NRWS	3	2
Cliff Swallow (*Petrochelidon pyrrhonota*)	CLSW	16	11
Black-capped Chickadee (*Poecile atricapillus*)	BCCH	6	1
Mountain Chickadee (*Poecile gambeli*)	MOCH	188	62
Red-breasted Nuthatch (*Sitta canadensis*)	RBNU	60	15
Brown Creeper (*Certhia americana*)	BRCR	14	4
House Wren (*Troglodytes aedon*)	HOWR	73	26
Blue-gray Gnatcatcher (*Polioptila caerulea*)	BGGN	3	1
Golden-crowned Kinglet (*Regulus satrapa*)	GCKI	8	1
Ruby-crowned Kinglet (*Regulus calendula*)	RCKI	153	58
Mountain Bluebird (*Sialia currucoides*)	MOBL	34	19

Hermit Thrush (*Catharus guttatus*)	HETH	56	17
American Robin (*Turdus migratorius*)	AMRO	284	121
Cedar Waxwing (*Bombycilla cedrorum*)	CEDW	3	3
Pine Grosbeak (*Pinicola enucleator*)	PIGR	16	6
Cassin's Finch (*Haemorhous cassinii*)	CAFI	25	7
Red Crossbill (*Loxia curvirostra*)	RECR	19	4
White-winged Crossbill (*Loxia leucoptera*)	WWCR	2	0
Pine Siskin (*Spinus pinus*)	PISI	167	60
Lark Sparrow (*Chondestes grammacus*)	LASP	4	1
Chipping Sparrow (*Spizella passerina*)	CHSP	59	32
Brewer's Sparrow (*Spizella breweri*)	BRSP	359	227
Fox Sparrow (*Passerella iliaca*)	FOSP	85	46
Dark-eyed Junco (*Junco hyemalis*)	DEJU	264	97
White-crowned Sparrow (*Zonotrichia leucophrys*)	WCSP	444	233
Vesper Sparrow (*Pooecetes gramineus*)	VESP	103	70
Savannah Sparrow (*Passerculus sandwichensis*)	SAVS	131	124
Song Sparrow (*Melospiza melodia*)	SOSP	44	25
Lincoln's Sparrow (*Melospiza lincolnii*)	LISP	390	222
Green-tailed Towhee (*Pipilo chlorurus*)	GTTO	123	65
Spotted Towhee (*Pipilo maculatus*)	SPTO	1	1
Western Meadowlark (*Sturnella neglecta*)	WEME	1	1
Red-winged Blackbird (*Agelaius phoeniceus*)	RWBL	25	25

Brown-headed Cowbird (*Molothrus ater*)	BHCO	12	12
Brewer's Blackbird (*Euphagus cyanocephalus*)	BRBL	41	36
Common Grackle (*Quiscalus quiscula*)	COGR	2	2
Orange-crowned Warbler (*Leiothlypis celata*)	OCWA	4	1
MacGillivray's Warbler (*Geothlypis tolmiei*)	MGWA	78	15
Common Yellowthroat (*Geothlypis trichas*)	COYE	22	19
Yellow Warbler (*Setophaga petechia*)	YEWA	104	63
Yellow-rumped Warbler (*Setophaga coronata*)	YRWA	183	71
Wilson's Warbler (*Cardellina pusilla*)	WIWA	129	63
Western Tanager (*Piranga ludoviciana*)	WETA	44	15
Black-headed Grosbeak (*Pheucticus melanocephalus*)	BHGR	3	2
Lazuli Bunting (*Passerina amoena*)	LAZB	34	8

Appendix D.

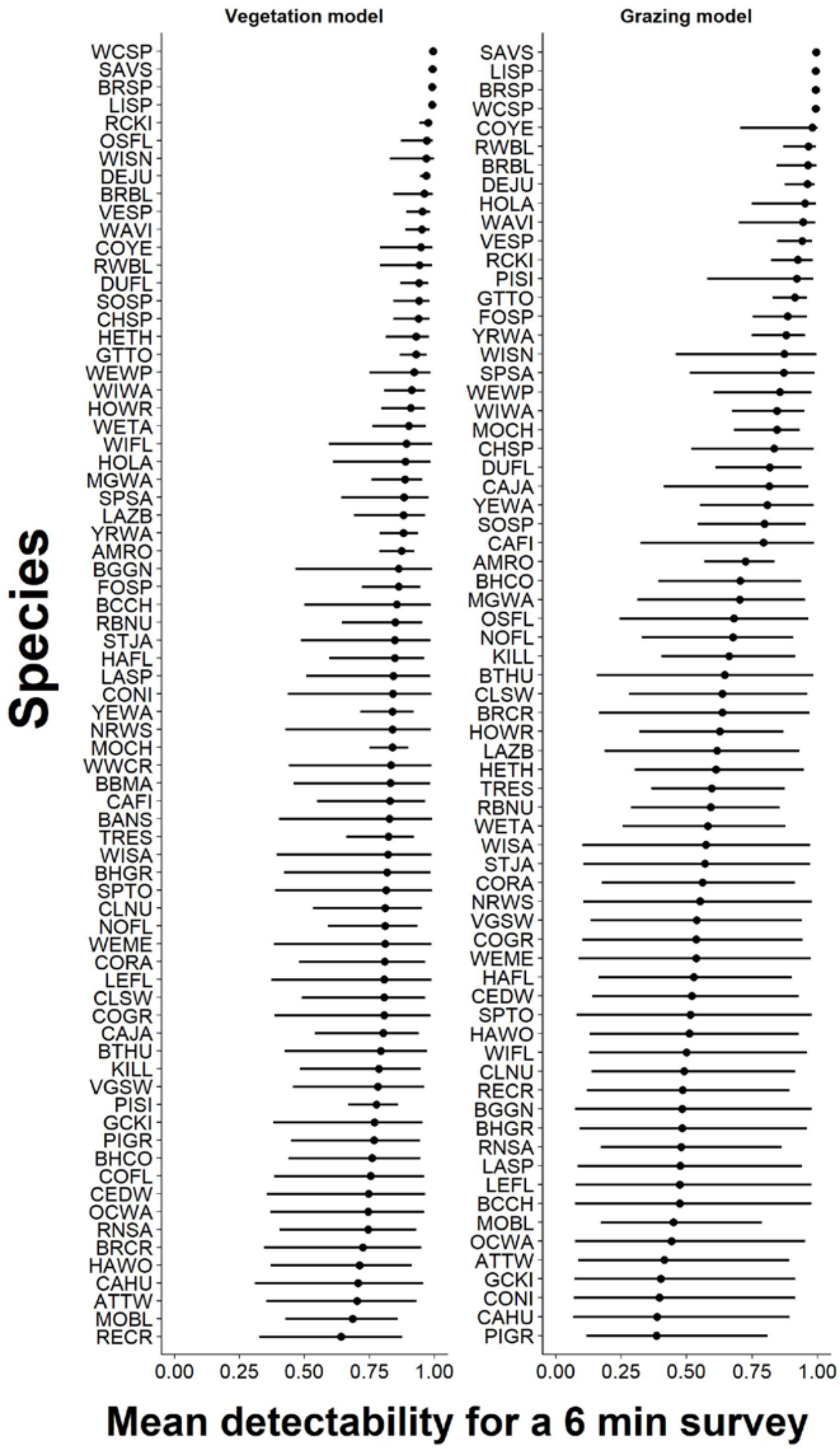

Figure D1. Species-specific mean detection probabilities for a 6-min survey averaged across all surveys informing the vegetation model (left panel) and the grazing model (right panel). Only species detected at least once are shown (data augmented species are not shown). Estimates represent mean values for $p^* = 1 - (1 - p)^6$ and are arranged from lowest to highest detectability. For full species names, see Appendix C.

Appendix E.

Plots in this appendix show species occupancy probabilities for survey points in relation to covariates. Each plot shows all statistically supported relationships with a different covariate. Occupancy relationships are depicted for each site (Gros Ventre [black] and Upper Green [green]). For species with a statistically supported difference in occupancy between sites (represented here as the difference between the two lines), species names are followed by “S*” in parentheses.

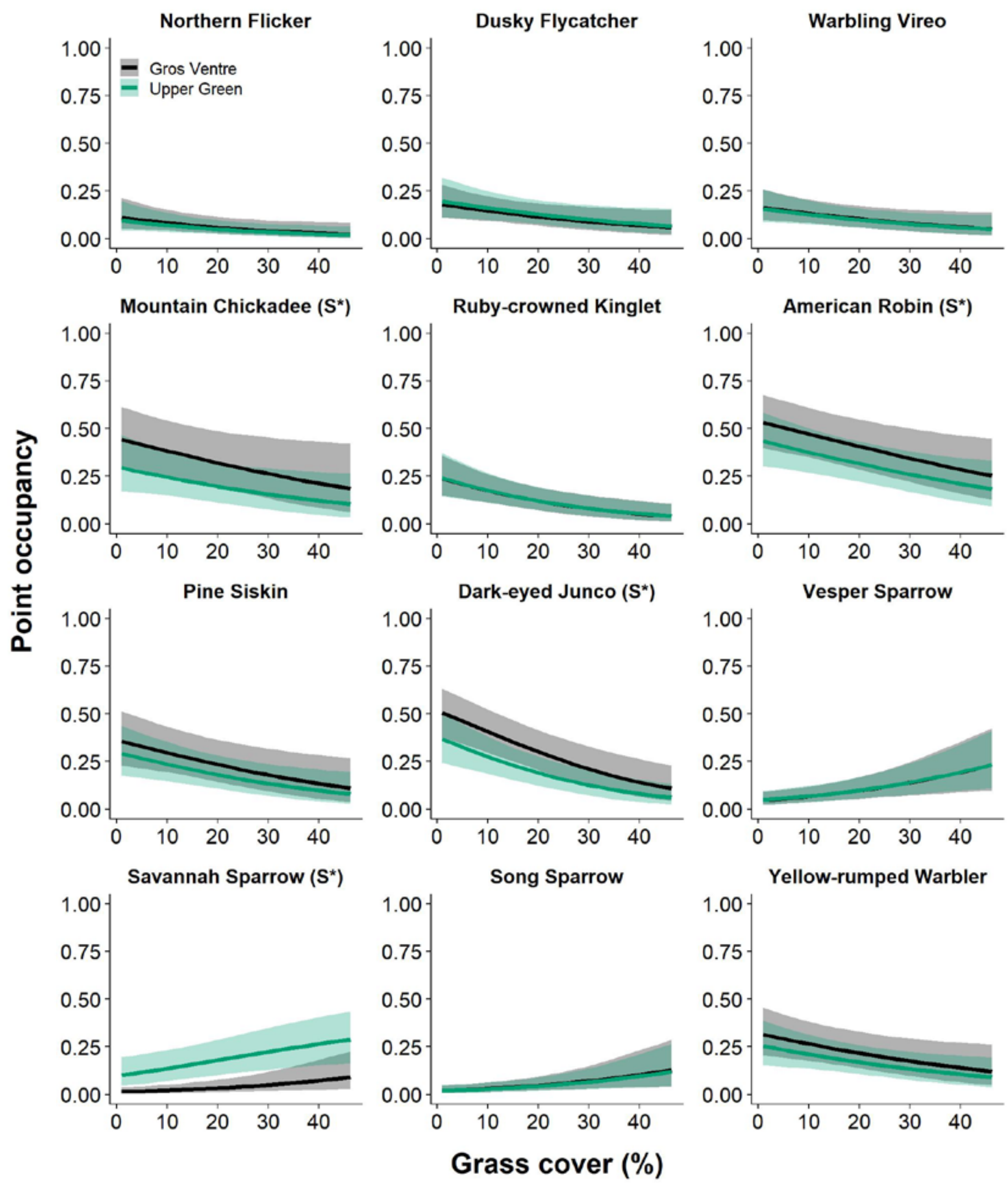

Figure E1. Statistically supported relationships with grass cover.

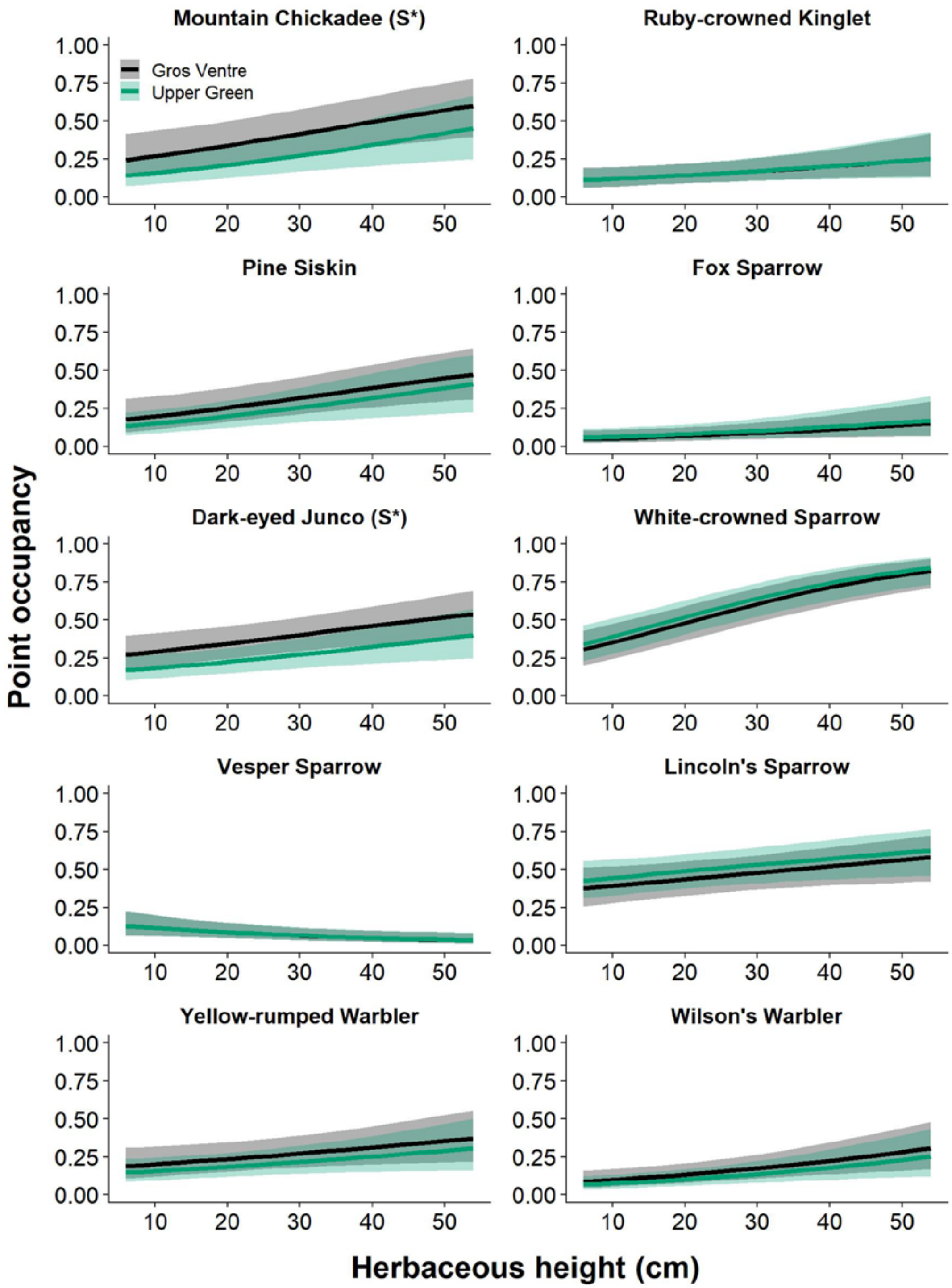

Figure E2. Statistically supported relationships with herbaceous height.

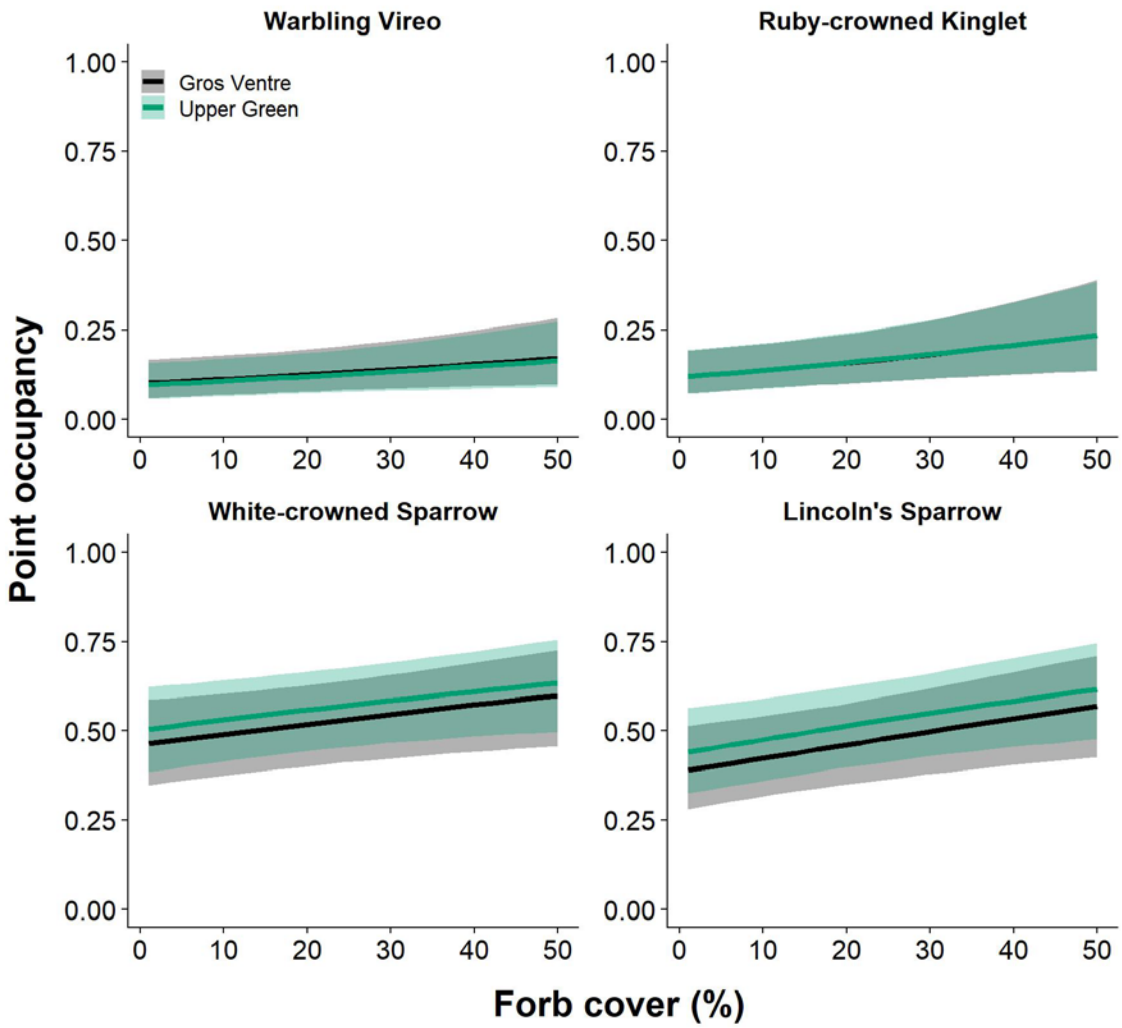

Figure E3. Statistically supported relationships with forb cover.

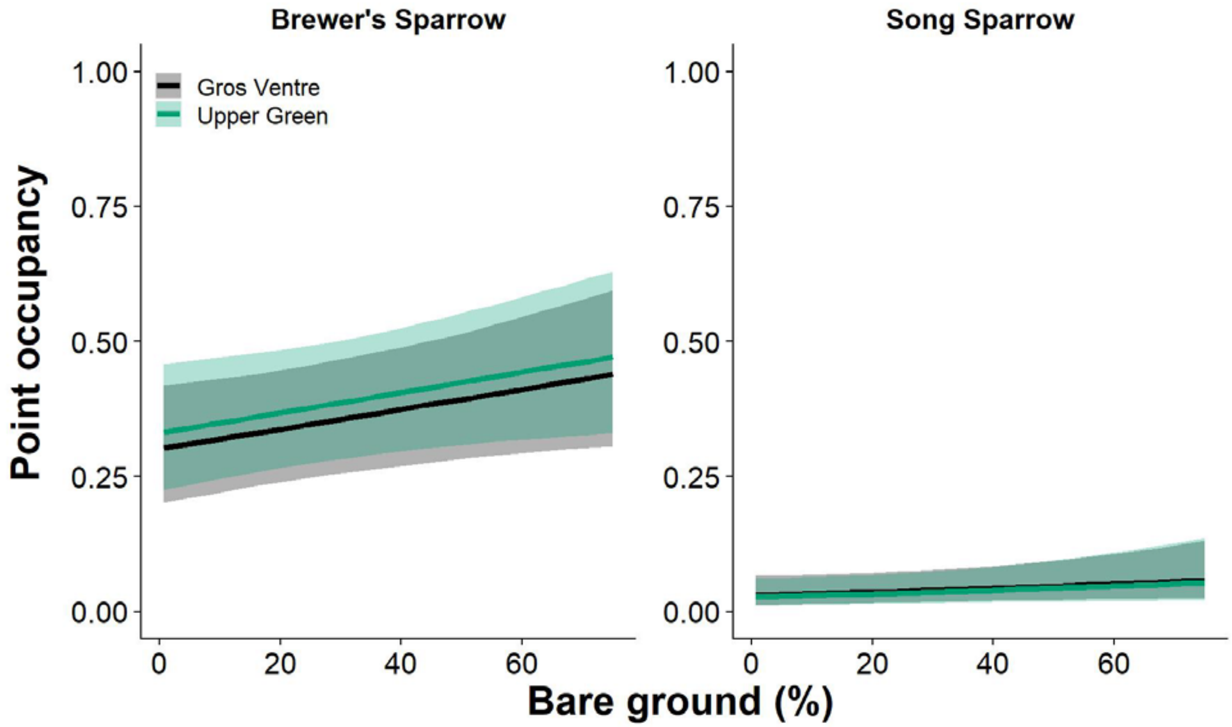

Figure E4. Statistically supported relationships with bare ground.

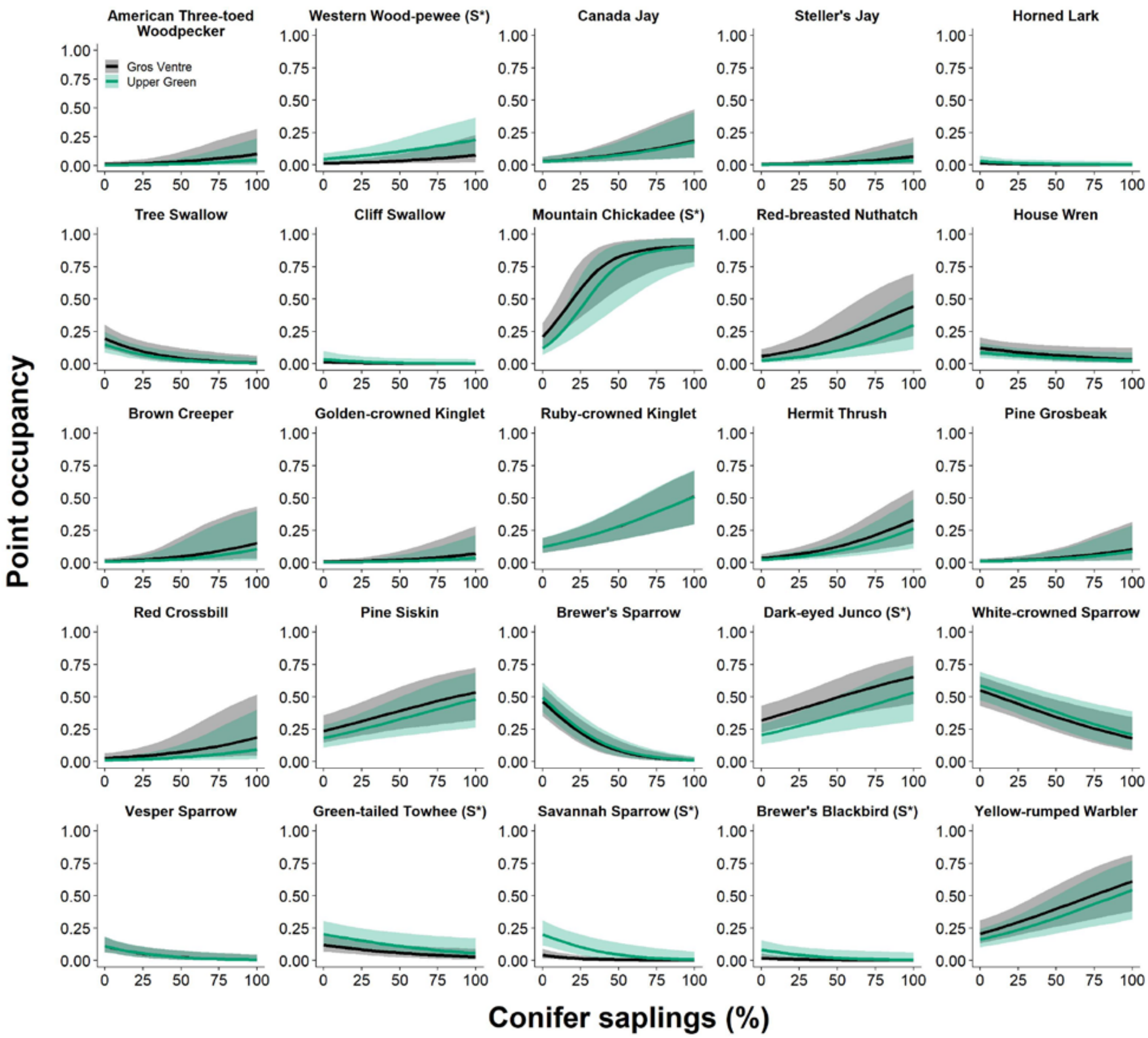

Figure E5. Statistically supported relationships with conifer shrubs (<3 m in height).

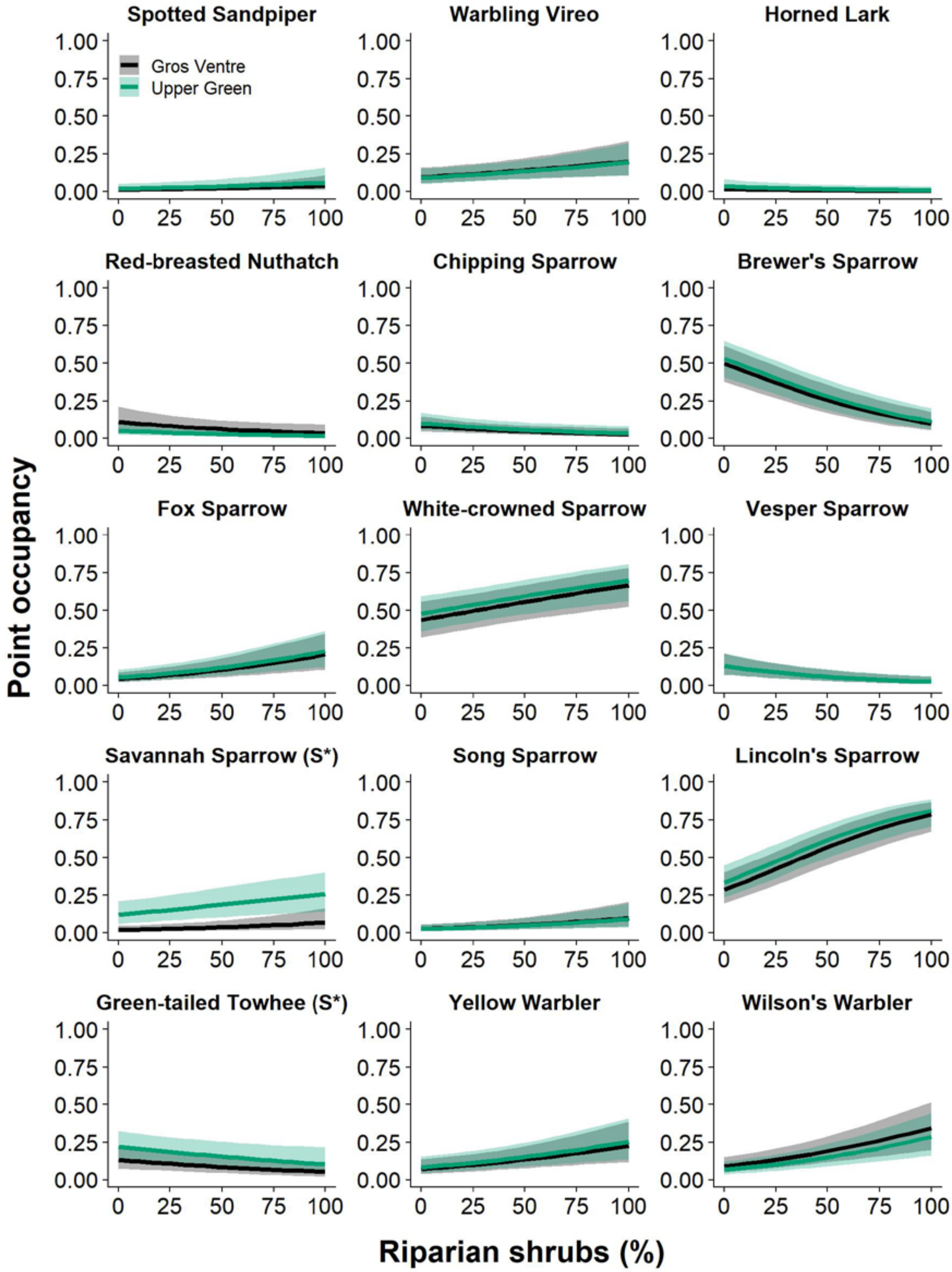

Figure E6. Statistically supported relationships with riparian shrubs (<3 m in height).

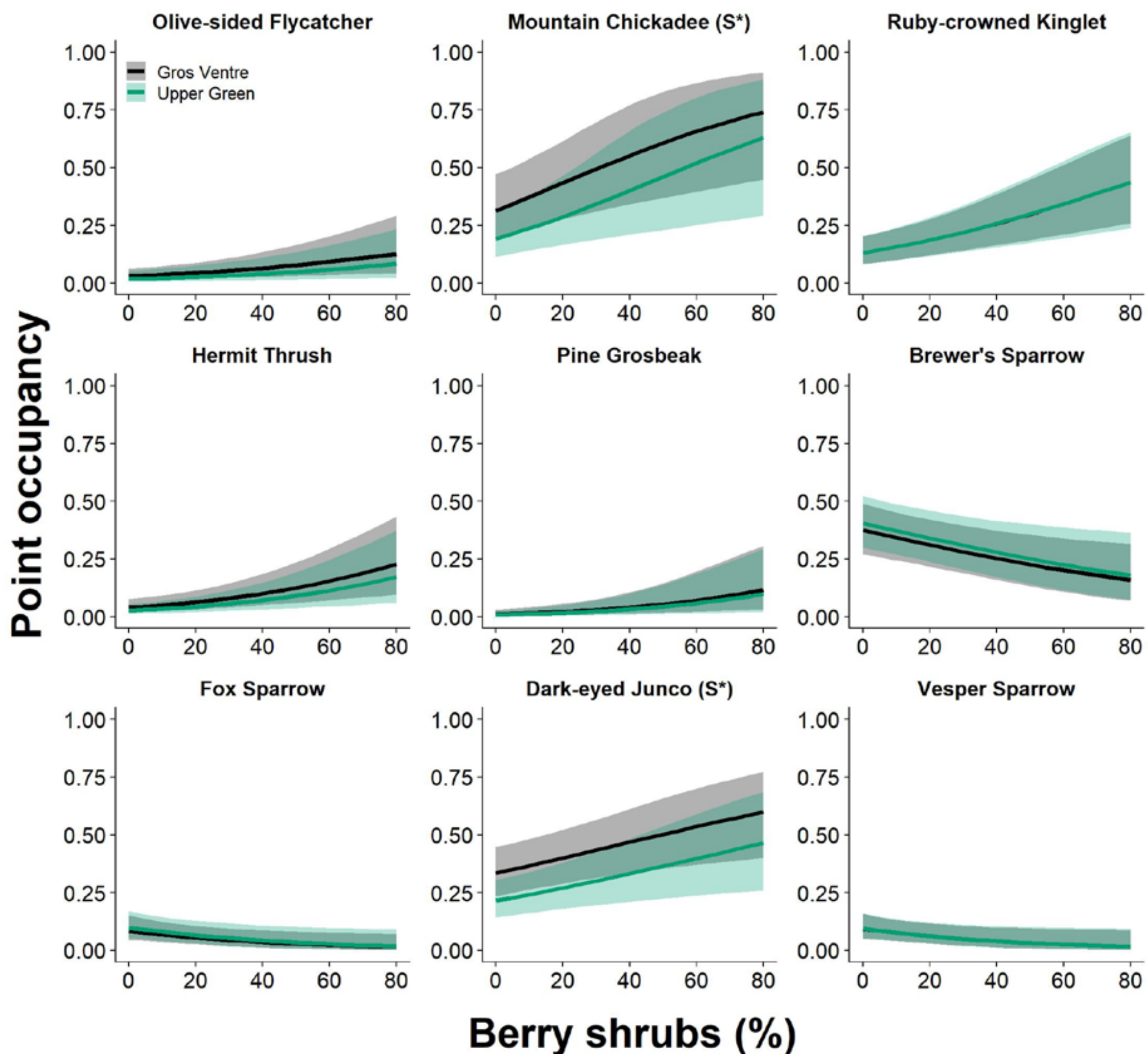

Figure E7. Statistically supported relationships with berry-producing shrub species (<3 m in height).

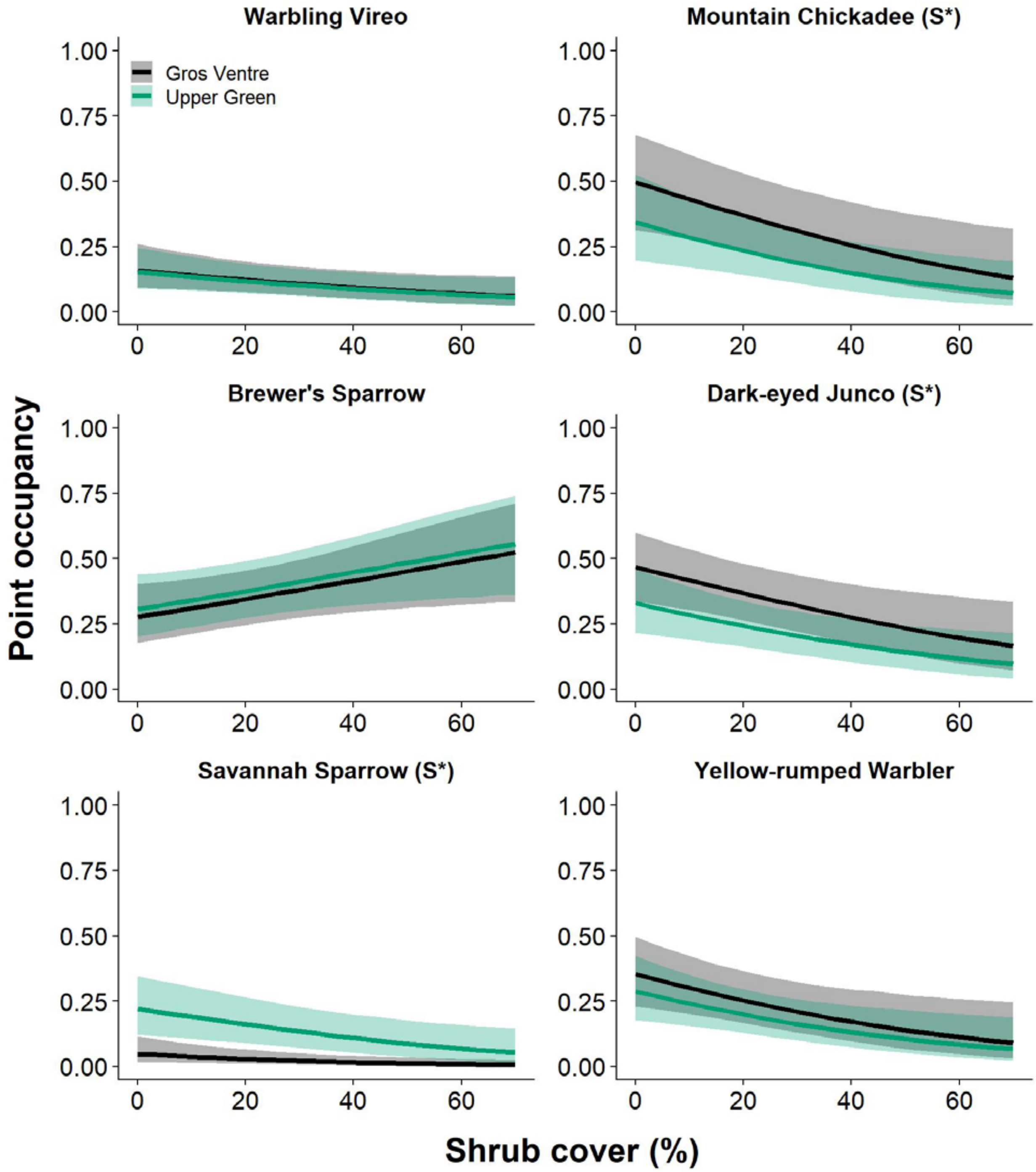

Figure E8. Statistically supported relationships with shrub cover (<3 m in height).

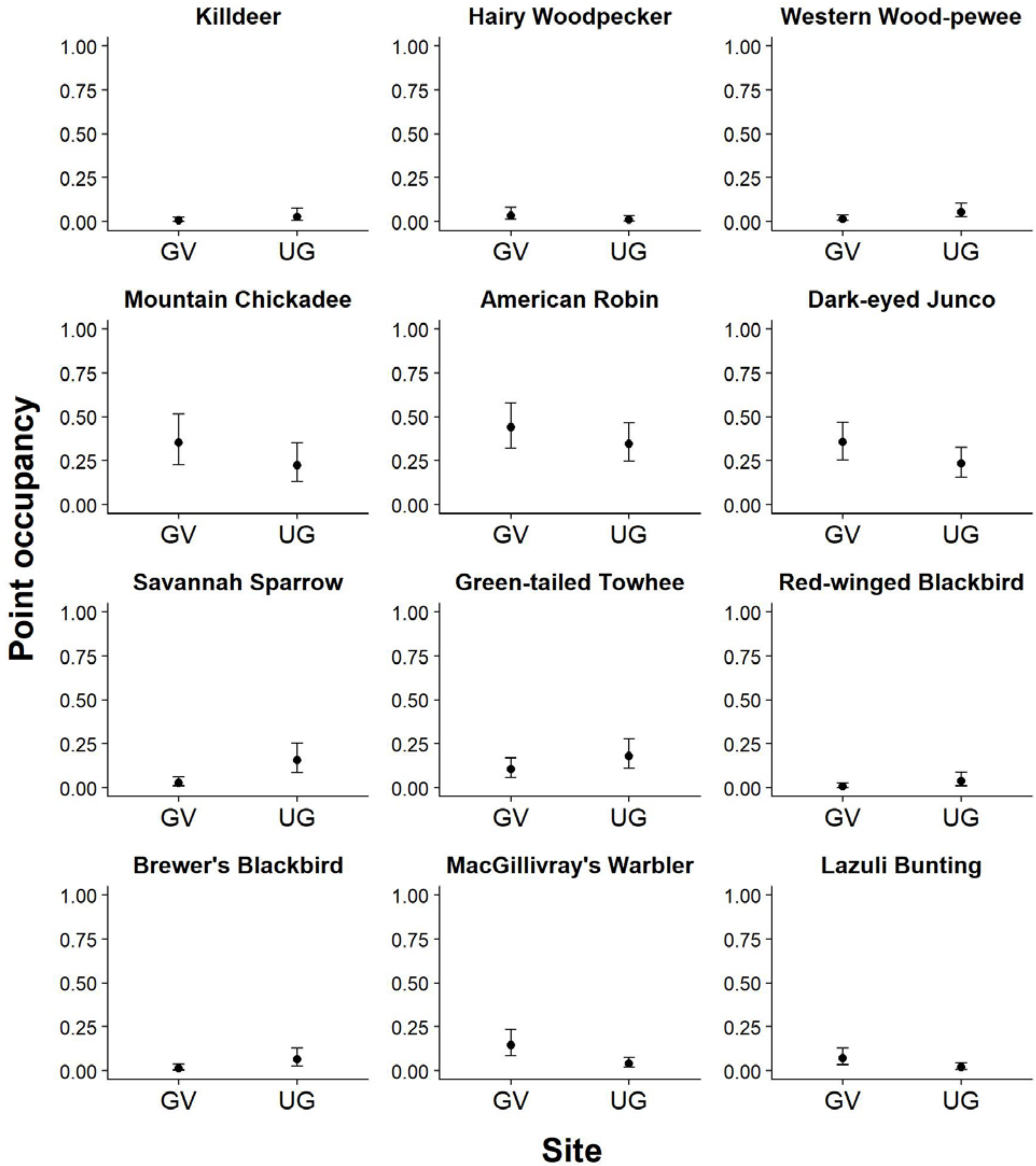

Figure E9. Statistically supported differences in species occupancy between Gros Ventre (GV) and Upper Green (UG) sites.

Appendix F.

Table F1. Prior distributions and estimates for hyper-parameters governing community occupancy models. For full parameter descriptions and model structures, see Appendix B.

Hyper-parameter	Prior	Estimate (95% BCI)	
		Vegetation model	Grazing model
Ω	Uniform(0,1)	0.79 (0.65,0.94)	0.83 (0.68,1)
mean(β_0)	*t*(0,1.6,7.8)	-0.78 (-1.49,-0.18)	-1.06 (-1.9,-0.36)
SD(β_0)	half-Cauchy(0,1)	1.89 (1.43,2.47)	1.78 (1.28,2.39)
mean(α_0)	*t*(0,1.6,7.8)	-2.94 (-3.53,-2.44)	-2.2 (-2.77,-1.67)
SD(α_0)	half-Cauchy(0,1)	1.43 (1.06,1.84)	1.17 (0.79,1.6)
mean(α_{Dev})	fixed at 0	--	--
SD(α_{Dev})	half-Cauchy(0,1)	0.41 (0.32,0.5)	0.51 (0.39,0.64)
mean($\alpha_{ForbCov}$)	*t*(0,1.6,7.8)	0.09 (0.03,0.15)*	--
SD($\alpha_{ForbCov}$)	half-Cauchy(0,1)	0.09 (0,0.16)	--
mean($\alpha_{GrassCov}$)	*t*(0,1.6,7.8)	-0.05 (-0.14,0.06)	--
SD($\alpha_{GrassCov}$)	half-Cauchy(0,1)	0.3 (0.21,0.4)	--
mean(α_{HerbHt})	*t*(0,1.6,7.8)	0.09 (0,0.17)	--
SD(α_{HerbHt})	half-Cauchy(0,1)	0.23 (0.15,0.31)	--
mean(α_{Bare})	*t*(0,1.6,7.8)	0.07 (0.01,0.13)*	--
SD(α_{Bare})	half-Cauchy(0,1)	0.09 (0.01,0.17)	--
mean($\alpha_{ShrubCov}$)	*t*(0,1.6,7.8)	-0.08 (-0.18,0.02)	--
SD($\alpha_{ShrubCov}$)	half-Cauchy(0,1)	0.24 (0.16,0.33)	--
mean($\alpha_{ShrubHt}$)	*t*(0,1.6,7.8)	0.07 (-0.01,0.15)	--
SD($\alpha_{ShrubHt}$)	half-Cauchy(0,1)	0.15 (0.05,0.24)	--
mean($\alpha_{Riparian}$)	*t*(0,1.6,7.8)	0.04 (-0.08,0.17)	--
SD($\alpha_{Riparian}$)	half-Cauchy(0,1)	0.39 (0.3,0.5)	--
mean(α_{Berry})	*t*(0,1.6,7.8)	0.05 (-0.06,0.15)	--
SD(α_{Berry})	half-Cauchy(0,1)	0.31 (0.2,0.42)	--
mean($\alpha_{Conifer}$)	*t*(0,1.6,7.8)	-0.06 (-0.27,0.12)	--
SD($\alpha_{Conifer}$)	half-Cauchy(0,1)	0.66 (0.49,0.89)	--
mean($\alpha_{UpperGreen}$)	*t*(0,1.6,7.8)	-0.13 (-0.41,0.17)	--
SD($\alpha_{UpperGreen}$)	half-Cauchy(0,1)	0.95 (0.67,1.25)	--
mean(α_{Cattle})	*t*(0,1.6,7.8)	--	0.02 (-0.06,0.09)
SD(α_{Cattle})	half-Cauchy(0,1)	--	0.07 (0,0.15)
mean(ζ_0)	*t*(0,1.6,7.8)	-0.91 (-1.25,-0.62)	-1.8 (-2.53,-1.17)
SD(ζ_0)	half-Cauchy(0,1)	0.63 (0.42,0.87)	1.18 (0.77,1.72)

mean($\zeta_{ShrubCov}$)	*t*(0,1.6,7.8)	-0.12 (-0.27,0.03)	--
SD($\zeta_{ShrubCov}$)	half-Cauchy(0,1)	0.34 (0.21,0.48)	--
mean($\zeta_{ShrubHt}$)	*t*(0,1.6,7.8)	0.04 (-0.09,0.16)	--
SD($\zeta_{ShrubHt}$)	half-Cauchy(0,1)	0.27 (0.16,0.38)	--
mean(ζ_{Cattle})	*t*(0,1.6,7.8)	--	0.08 (-0.04,0.18)
SD(ζ_{Cattle})	half-Cauchy(0,1)	--	0.11 (0,0.25)
mean(ζ_{DOY})	*t*(0,1.6,7.8)	-0.04 (-0.16,0.08)	-0.21 (-0.51,0.11)
SD(ζ_{DOY})	half-Cauchy(0,1)	0.21 (0.08,0.35)	0.78 (0.5,1.09)
mean(ζ_{DOY^2})	*t*(0,1.6,7.8)	-0.1 (-0.21,0)	-0.18 (-0.38,0)
SD(ζ_{DOY^2})	half-Cauchy(0,1)	0.21 (0.13,0.32)	0.32 (0.16,0.51)
mean(ζ_{Time})	*t*(0,1.6,7.8)	-0.03 (-0.14,0.09)	-0.01 (-0.18,0.15)
SD(ζ_{Time})	half-Cauchy(0,1)	0.27 (0.14,0.44)	0.39 (0.21,0.58)
mean(ζ_{Time^2})	*t*(0,1.6,7.8)	-0.06 (-0.14,0.03)	-0.02 (-0.15,0.11)
SD(ζ_{Time^2})	half-Cauchy(0,1)	0.17 (0.08,0.27)	0.26 (0.12,0.42)
$\rho_{\psi\theta}$	Uniform(-1,1)	0.87 (0.72,0.97)	0.67 (0.4,0.87)
$\rho_{\theta p}$	Uniform(-1,1)	0.44 (0.07,0.77)	0.72 (0.33,0.97)

*Community-level means for covariate relationships whose credible intervals excluded zero, suggesting statistical support for covariate relationships with species richness.

Appendix G.

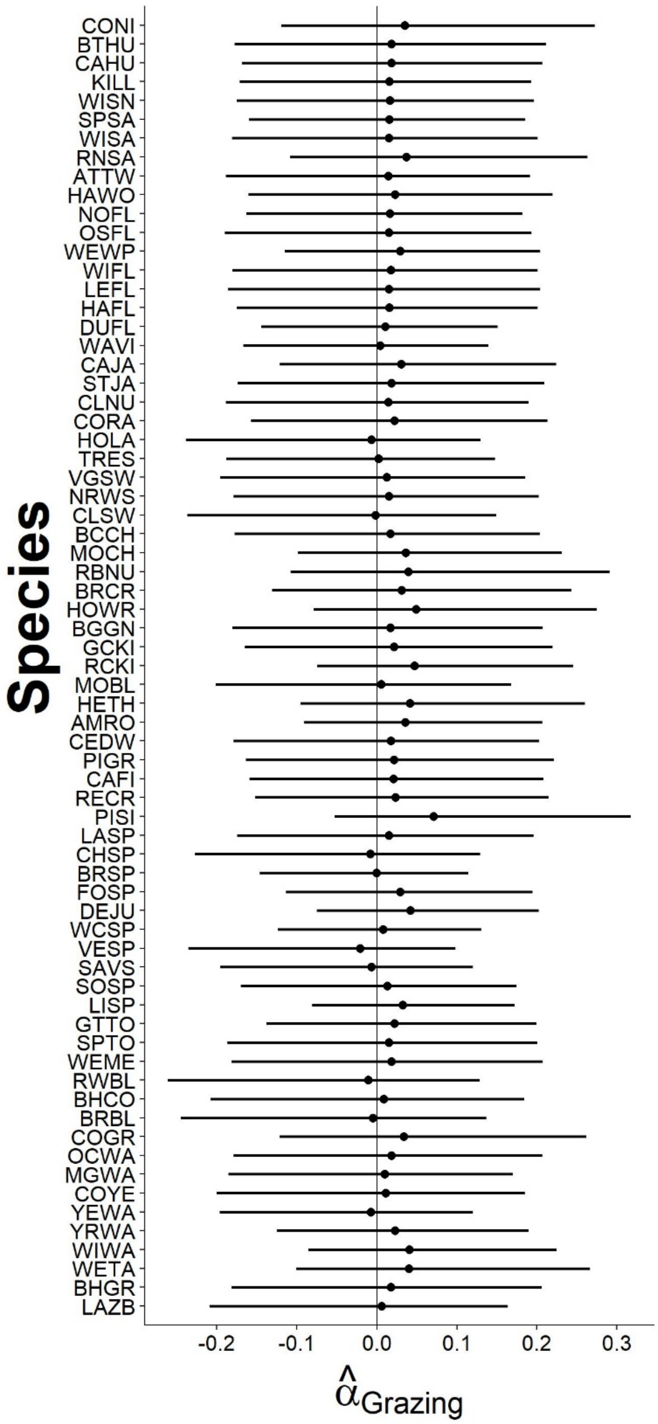

Figure G1. Estimated logit-linear relationships of species occupancy with cattle days (Grazing) estimated with the grazing model fitted to data from the Upper Green site. Only species detected at least once at Upper Green are shown (data augmented species are not shown).

Appendix H.

Data supplement – we provide a zip file ("Bridger-Teton Grazing Data Supplement.zip") containing 1) R scripts for data compilation, analysis, results summaries, and plots, 2) data contained in an R workspace needed to run scripts, and 3) an Excel file ("Occupancy_model_estimates.xlsx") containing all model parameter estimates.