

Metrics for conservation success: Using the “Bird-Friendliness Index” to evaluate grassland and aridland bird community resilience across the Northern Great Plains ecosystem

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Abstract

Aim: Evaluating conservation effectiveness is essential to protect at-risk species and to maximize the limited resources available to land managers. Over 60% of North American grassland and aridlands have been lost since the 1800s. Birds in these habitats are among the most imperilled in North America, yet most remaining habitats are unprotected. Despite the need to measure impact, conservation efforts on private and working lands are rarely evaluated, due in part to limited availability of suitable methods.

Location: Northern Great Plains of North America.

Methods: We developed a novel metric to evaluate grassland and aridland bird community response to habitat management, the Bird-Friendliness Index (BFI), consisting of density estimates of grassland and aridland birds weighted by conservation need and a functional diversity metric to incorporate resiliency. We used the BFI to identify ecologically significant areas for grassland and aridland birds and compared them with other prioritizations. Then, we used the BFI to evaluate the effects of simulated habitat management on grassland and aridland bird communities.

Results: The most resilient bird communities were found in the Prairie Potholes region of Alberta, Saskatchewan and North Dakota and the lowest BFI values in the southern and western regions of the Northern Great Plains. BFI values were significantly greater in areas included in one or more prioritization, and increased with the number of prioritizations an area was included within. BFI values increased in response to simulated bird-friendly habitat management.

Main conclusions: Our findings suggest that practices recommended for use in bird-friendly grassland habitat management plans will increase the abundance and resilience of the grassland and aridland bird community and that bird community responses to management will be detected using the BFI. The BFI is a tool by which conservationists and managers can carry out adaptive management and accountable conservation now and into the future.

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KEYWORDS

accountable conservation, aridland birds, bird-friendly, conservation, grassland birds, grazing, indices, management, ranching, working lands

1 | INTRODUCTION

Grasslands and aridlands, and the birds that rely on them, are under threat. Over 60% of North America's grasslands (i.e., grass-dominated habitats) have been lost since the 1800s, with some of the greatest losses in the Northern Great Plains Tallgrass and Mixed-grass Prairie regions (Comer et al., 2018). Similarly, aridlands (i.e., arid shrub-dominated communities) have experienced habitat loss and degradation due to urban sprawl and energy development, drought, desertification and invasive species (North American Bird Conservation Initiative & U.S. Committee, 2014; Ryan et al., 2008). Though much of the grassland and aridland loss and degradation occurred in the nineteenth and twentieth centuries, agricultural conversion, urban sprawl and energy development continue today (Samson et al., 2004; Turnbull et al., 2014). For example, 3 million acres of cropland—a less-suitable habitat for grassland birds—were created during 2008–2012, 77% of which were converted from grassland and 8% from shrubland (Lark et al., 2015). Additionally, grassland and aridland birds face other ongoing threats including agricultural intensification and pesticides (Stanton et al., 2018). Consequently, populations of grassland-dependent birds have declined by an estimated 53.3% during 1970–2017, while aridland birds declined 17%–46% during 1968–2013, (North American Bird Conservation Initiative, 2014; Rosenberg et al., 2019). These extensive declines highlight the need for effective grassland and aridland bird conservation efforts. Conservation metrics often rely on evaluations of habitat quality for single species that are treated as indicators (Simberloff, 1998). Yet single species rarely serve as effective proxies for larger communities, due to limited overlap in habitat requirements, functional roles and responses to management actions among species (Cushman et al., 2010; Johnson et al., 2019; Larsen et al., 2009; Lindenmayer et al., 2015; Winter et al., 2005). Single-species indicator approaches are particularly ill-suited to communities with threatened species, as more common species often perform poorly as surrogates for species of conservation concern (Stephens et al., 2019). Additionally, species-specific abundance alone is rarely a reliable predictor of habitat quality (Johnson, 2008). Instead, metrics incorporating distributions, habitat relationships and diversity of multiple species better quantify the response of ecological communities to conservation actions (Kapos et al., 2008, 2009; Nuttle et al., 2003).

By using multispecies metrics, we can incorporate the variability among species in their habitat use, foraging preferences and functional contributions to ecosystems (e.g., pest control, seed dispersal; Şekercioğlu et al., 2016). Together, these suites of species and their associated functional traits and services influence the integrity (i.e., ecosystem structure, composition and function; Karr, 1981) and resilience (i.e., ability to resist or recover from disturbances; Scheffer et al., 2015) of the larger ecological community (Fischer et al., 2007).

Functional traits such as diet, body size and habitat use influence species' ecological impacts, and communities composed of species with complementary suites of traits will have greater ecological integrity and resilience than communities with greater redundancy (Cardinale et al., 2012; McGill et al., 2006). Consequently, functional diversity metrics incorporating functional traits and species abundance provide an indirect way to measure resilience and integrity (Standish et al., 2014), and are increasingly used in large-scale assessments of North American bird and ecological communities (Schipper et al., 2016; Schleuter et al., 2010).

Private lands offer a potential source of hope for grassland and aridland birds, as 84% of grasslands and 44% of aridlands are held in private ownership; therefore, private land conservation efforts could have substantial population-level impacts (Askins et al., 2007; North American Bird Conservation Initiative & U.S. Committee, 2011). Over 90% of the breeding distribution of seven obligate grassland-breeding birds is on private lands, including species such as the Eastern Meadowlark (*Sturnella magna*) whose populations are in steep decline, making conservation efforts a top priority (North American Bird Conservation Initiative & U.S. Committee, 2013). As a result, private lands are essential to ensure conservation of imperilled grassland and aridland birds in the future (North American Bird Conservation Initiative, 2016). Recognizing the importance of private land efforts to bird conservation, the National Audubon Society developed the Conservation Ranching programme (<https://www.audubon.org/conservation/ranching>), a market-based conservation solution that aims to conserve imperilled grassland and aridland bird species and habitats through partnership with private landowners. Through this programme, Audubon rangeland biologists partner with ranchers to co-develop bird-friendly Habitat Management Plans.

In order to evaluate the success of habitat enhancement initiatives such as the Conservation Ranching programme in conserving birds and their habitats on private lands in North American grasslands and aridlands, we developed a metric evaluating bird community response to habitat management practices. This metric, the Bird-Friendliness Index (BFI), uses avian count, functional trait and conservation status data together with remotely sensed environmental data to evaluate the capacity of a landscape to support an abundant, diverse and resilient bird community. The BFI was designed to enable inference at multiple spatial and temporal scales and incorporates standardization methods that facilitate spatial comparisons among North American grassland and aridland bird communities spanning broad climatic and community gradients. We use the Northern Great Plains of North America as a case study and highlight the insights the BFI provides into the plight of grassland and aridland birds and the integrity and resilience of the larger ecosystems in which they reside. Here, we address two objectives 1) to demonstrate the use of the BFI to map grassland bird resilience

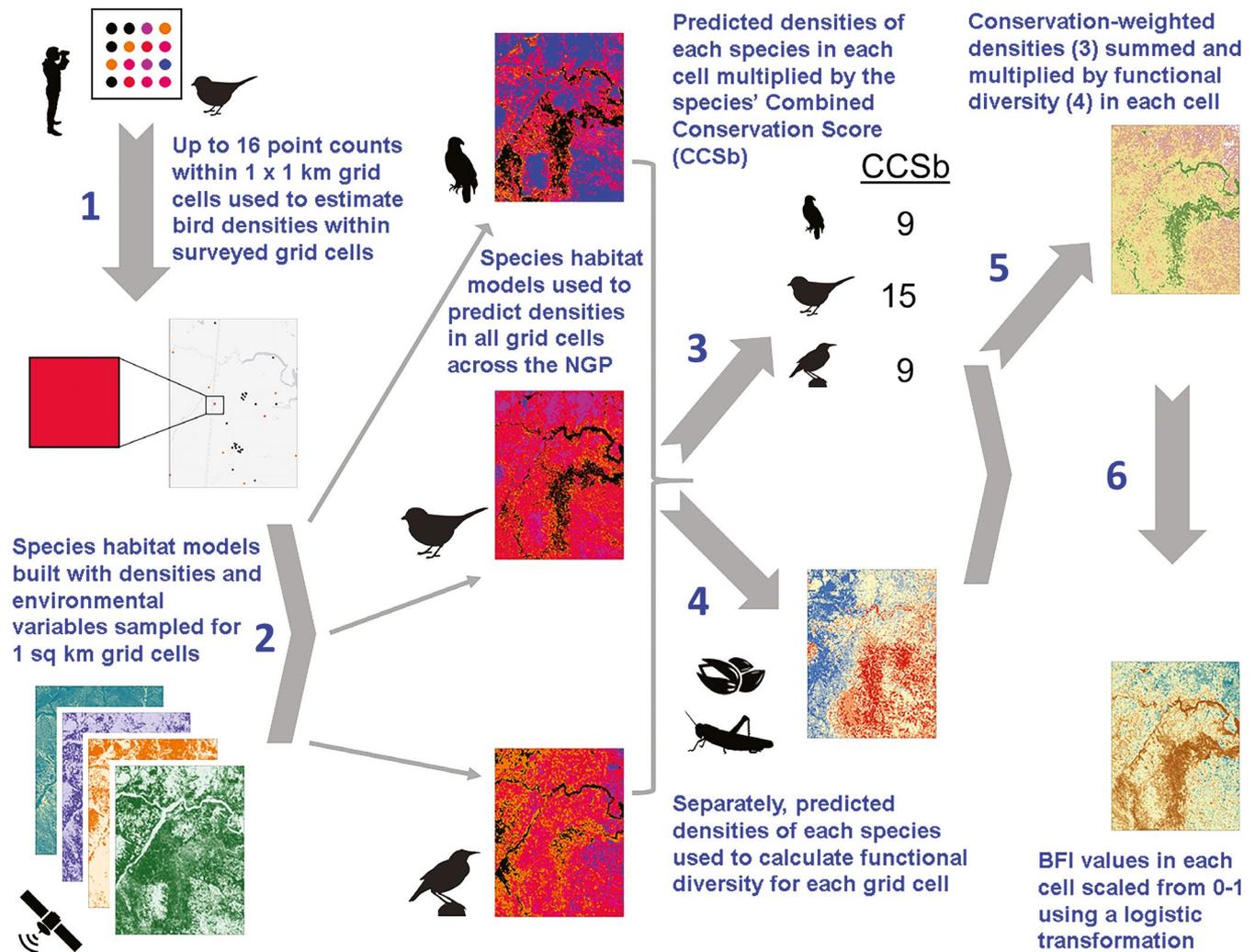


FIGURE 1 Flowchart visualizing the methods for calculating the Bird-Friendliness Index. The six numbered steps are shown in blue. Under the Integrated Monitoring in Bird Conservation Regions protocol, birds were surveyed at up to 16 points within a 1 km² grid cell. First, densities were estimated on a species-specific basis (here: Baird's Sparrow) for surveyed grid cells using distance and time removal information to correct for imperfect detection (1). Estimated densities were combined with remotely sensed environmental covariates in bird-habitat models to predict densities across the Northern Great Plains (NGP) study area (2; here Swainson's Hawk, Baird's Sparrow and Western Meadowlark). Predicted densities across the NGP were multiplied by species-specific breeding season Combined Conservation Scores (CCSb; 3). Predicted densities and species' functional traits (e.g., diet composition) were used to calculate functional diversity for each grid cell (4). Conservation-weighted densities were summed for each grid cell and multiplied that cell's functional diversity (5). Finally, values were scaled from zero to one using a logistic transformation (6)

across the Northern Great Plains and identify ecologically significant areas for grassland and aridland birds and 2) to evaluate the effects of bird-friendly habitat management practices on grassland and aridland bird communities.

2 | METHODS

2.1 | Bird-Friendliness Index overview

We calculated the BFI using species density, conservation status and diversity of the entire community of grassland birds at a site. Bird abundance and changes in relative abundance over time are presently the most common tool for monitoring and evaluating landbird

populations (e.g., Rosenberg et al., 2016; Sauer et al., 2017) and form the foundation of the BFI. We also include conservation status to ensure that the BFI does not underestimate conservation need by diluting the contribution of vulnerable species (Beissinger, 2000). Lastly, the BFI also includes a measure of functional diversity as a measure of the intactness and resilience of both the grassland and aridland bird community and their larger ecological community (Fischer et al., 2007).

The BFI is the sum product of estimated avian density and conservation status times a measure of functional diversity based on species traits (i.e., diet, foraging strata and body mass), as follows:

$$\text{BFI} = \sum_{\text{species}} (\text{Density} \times \text{ConservationStatus}) \times \text{FunctionalDiversity}$$

We calculated the index in six steps, all of which are calculated at the resolution of 1 km² grid cells (Figure 1). We chose this resolution because it represents the smallest scale at which the avian survey data used are summarized (see Section 2.4), and is large enough to contain entire home ranges of most study species. Calculation of BFI scores at this resolution also enables BFI values to be easily aggregated to larger scales (e.g., ranch, state) by calculating the mean or median of grid cells included within the area of interest.

2.2 | Study area

Our study area encompasses the Northern Great Plains ecosystem, as defined by the West-Central Semi-arid Prairies level II ecoregion (Commission for Environmental Cooperation (CEC), 2009). This region stretches from the Prairie Pothole region of south-central Canada to Nebraska and from eastern North Dakota to western Montana (Figure S1). It consists primarily of Northwestern Great Plains Mixed-grass Prairie, with some Northern Great Plains Fescue Mixed-grass Prairie along the northern edge of the study area, and Western Great Plains Shortgrass Prairie and Western Great Plains Sand Prairie to the south (Comer et al., 2018). The study area also includes extensive arid sagebrush steppe habitat (i.e., aridlands) intermixed with grasslands (Connelly et al., 2004).

2.3 | Study species

We used the 2016 State of North America's Birds species assessment to identify 107 bird species that use grasslands or aridlands as their primary or secondary breeding habitat (North American Bird Conservation Initiative, 2016). We selected grassland and aridland birds because the two habitats are intermixed at relatively fine spatial scales in this region, and private landholdings on which management practices are implemented often include both habitat types and, consequently, communities. These species were evaluated for inclusion in the BFI (see Section 2.4).

2.4 | Avian data

We used avian point count data collected under Bird Conservancy of the Rockies's Integrated Bird Monitoring in Bird Conservation Regions (IMBCR) programme (Pavlacky et al., 2017). The IMBCR uses a hierarchically nested sampling design in which spatially balanced random stratification is used to select 1 km² grid cells within BCRs. Each grid cell includes up to 16-point count locations evenly spaced at distances of 250 m. At each point, five- (2009) or six-minute (2010–2014) unlimited-distance point count surveys were conducted between one-half hour before and five hours after sunrise during the late spring and early summer. Observers recorded species, age (where possible), first minute of observation, type of

detection (call, song, visual), day, time of day (2010–2014 only) and primary habitat type at the point and measured distance using a laser rangefinder. We excluded flyover observations, juvenile birds and observations with missing minutes or distances.

We assigned IMBCR survey points to spatial strata defined as unique combinations of BCRs and states or provinces (Figure S1a). We defined strata this way because BCRs delineate ecoregions with similar bird communities and habitats; combining with states/provinces ensures that each stratum faces similar policy and management regimes. These strata are also used in hierarchical modeling of Audubon Christmas Bird Count data (Soykan et al., 2016) and North American Breeding Bird Survey data (Sauer et al., 2017), so represent a comparable scale of summarization. We used a subset of the IMBCR data for strata sampled annually from 2009 to 2014. This limited us to surveys conducted within Montana, North Dakota, South Dakota, Wyoming and Nebraska. Our study area encompassed five BCRs: Northern Rockies (BCR 10), Prairie Potholes (BCR 11), Badlands and Prairies (BCR 17), Shortgrass Prairie (BCR 18) and Central Mixed-grass Prairie (BCR 19; Figure S1a).

Our dataset spanned 2009–2014 and included a total of 1102 unique 1-km² grid cells, with 242–594 grid cells surveyed annually and an average of 11 points sampled per grid cell. Of these, we included 22 of our initial list of 107 species with sufficient data from which to estimate density (defined as a minimum of 60 detections; see Section 2.6; Table 1), including 20 species that use grasslands and 5 species that use aridlands as a major breeding habitat. Three species (Lark Sparrow, Loggerhead Shrike and Western Kingbird) use both grasslands and aridlands as major breeding habitats (Table 1), and many species use both habitats facultatively. The final dataset for these 22 species included 43,285 observations of 97,067 birds.

2.5 | Environmental data

We used 12 environmental variables in individual species models to project avian densities across the study area (see Section 2.7). We sampled all environmental variables for 799,015 1 km² grid cells, covering the study area and aligned with the avian density grid cells. The 12 variables comprised five classes of environmental predictors (climate, ecosystem, land cover, temporal and topographical) and were collected or calculated either once (e.g., land cover from 2010) or annually (e.g., ecosystem variables; Table 2). We included three land cover types—grassland, shrubland and cropland—as they represent habitat frequently used by grassland and aridland birds; other potential land cover types (e.g., developed) were too sparse to explain much variation in bird occurrence or abundance. We calculated proportion cover for all land cover types, and mean patch size and patch cohesion index (a measure of physical connectedness; McGarigal et al., 2002) for grassland only using package *SDMTools* (VanDerWal et al., 2014) in R version 3.5.2 (R Core Team, 2018). We selected environmental variables based on known relationships with distribution and abundance of grassland and aridland birds (Table S1). Correlations among predictors were minimal (all $r \leq |.70|$).

TABLE 1 Grassland and aridland bird species included in Bird-Friendliness Index estimation for the NGP, their major breeding habitats, functional species grouping, breeding season Combined Conservation Score (CCSb) and mean density (birds/km²) at surveyed grid cells

Common name	Scientific Name	Major breeding habitats	Functional species	CCSb	Density (mean ± SE)
Baird's Sparrow	<i>Cetronyx bairdii</i>	Grassland	Ground-foraging invert- & granivores	15	0.38 ± 0.04
Bobolink	<i>Dolichonyx oryzivorus</i>	Grassland	Ground-foraging invert- and frugivores	12	2.07 ± 0.16
Brewer's Sparrow	<i>Spizella breweri</i>	Aridland	Ground-foraging invert- and granivores	11	9.25 ± 0.63
Chestnut-collared Longspur	<i>Calcarius ornatus</i>	Grassland	Ground-foraging invert- and granivores	15	3.72 ± 0.27
Clay-coloured Sparrow	<i>Spizella pallida</i>	Grassland	Ground-foraging invert- and granivores	9	4.77 ± 0.47
Eastern Kingbird	<i>Tyrannus tyrannus</i>	Grassland, forest	Understorey-foraging invert-, frug- and granivores	10	1.90 ± 0.08
Grasshopper Sparrow	<i>Ammodramus savannarum</i>	Grassland	Ground- or aerial-foraging invertivores	12	28.01 ± 0.84
Horned Lark	<i>Eremophila alpestris</i>	Grassland, tundra	Ground-foraging invert- and granivores	9	21.58 ± 0.95
Lark Bunting	<i>Calamospiza melanocorys</i>	Grassland	Ground-foraging invert- and granivores	12	17.10 ± 0.68
Lark Sparrow	<i>Chondestes grammacus</i>	Grassland, aridland	Ground-foraging invert- and granivores	9	10.44 ± 0.49
Loggerhead Shrike	<i>Lanius ludovicianus</i>	Grassland, aridland	Small ground-foraging invert-, vertivores and scavengers	11	0.04 ± 0.005
Long-billed Curlew	<i>Numenius americanus</i>	Grassland	Ground- or aerial-foraging invertivores	14	0.14 ± 0.01
Northern Harrier	<i>Circus hudsonius</i>	Grassland, wetland	Large ground-foraging invert-, vertivores and scavengers	11	0.10 ± 0.01
Rock Wren	<i>Salpinctes obsoletus</i>	Aridland	Small ground-foraging invert- and vertivores	10	1.44 ± 0.07
Savannah Sparrow	<i>Passerculus sandwichensis</i>	Grassland, tundra	Ground-foraging invert- and frugivores	8	1.71 ± 0.18
Sharp-tailed Grouse	<i>Tympanuchus phasianellus</i>	Grassland, forest	Midsized ground-foraging herb- and granivores	10	0.26 ± 0.02
Sprague's Pipit	<i>Anthus spragueii</i>	Grassland	Ground- or aerial-foraging invertivores	14	0.17 ± 0.02
Swainson's Hawk	<i>Buteo swainsoni</i>	Grassland	Large ground-foraging invert- and vertivores	9	0.02 ± 0.003
Upland Sandpiper	<i>Bartramia longicauda</i>	Grassland	Ground- or aerial-foraging invertivores	10	1.27 ± 0.06
Vesper Sparrow	<i>Pooecetes gramineus</i>	Grassland	Ground-foraging invert- and granivores	10	14.70 ± 0.42
Western Kingbird	<i>Tyrannus verticalis</i>	Grassland, aridland	Mid- to upperstorey foraging invertivores	7	0.92 ± 0.05
Western Meadowlark	<i>Sturnella neglecta</i>	Grassland	Ground-foraging invert- and granivores	9	28.24 ± 0.60

Note: Conservation scores and habitat associations come from the 2016 State of the Birds (North American Bird Conservation Initiative, 2016).

2.6 | Density estimation

As the avian dataset included both time of first detection and distance of observations, we used a conditional multinomial maximum-likelihood formulation of time removal and distance

sampling models developed by Sólymos et al. (2013) and implemented using command *cmulti* in R package *detect* (Solymos et al., 2016). This method improves upon distance estimation alone, which allows estimation of just one component of detection probability (perceptibility), by utilizing minute of first observation to

TABLE 2 Environmental variables used as predictors in the species distribution models, with their sources, frequency of collection or calculation and citations

Type	Variable	Frequency	Source
Climate	Climatic moisture deficit	Single	Climatic Research Unit Time series 3.22 dataset, 1981–2010 statistically downscaled climate normal (Wang et al., 2016)
Climate	Spring degree-days below 0°C	Single	Climatic Research Unit Time series 3.22 dataset, 1981–2010 statistically downscaled climate normal (Wang et al., 2016)
Ecosystem	Litter biomass (g C/m ²)	Annual	CASA ecosystem model (Potter et al., 1993, 2007)
Ecosystem	Net primary productivity (g C/m ²)	Annual	CASA ecosystem model (Potter et al., 1993, 2007)
Ecosystem	Nitrous oxide flux (g N ₂ O/m ² -day)	Annual	CASA ecosystem model (Potter et al., 1993, 2007)
Ecosystem	Soil moisture (cm; 0–10 cm depth)	Annual	CASA ecosystem model (Potter et al., 1993, 2007)
Land cover	Proportion cropland	Single	Derived from Commission for Environmental Cooperation's North American Environmental Atlas 2010 (Canada Centre for Remote Sensing (CCRS) et al. 2017)
Land cover	Proportion grassland	Single	Derived from Commission for Environmental Cooperation's North American Environmental Atlas 2010 (Canada Centre for Remote Sensing (CCRS) et al. 2017)
Land cover	Proportion shrubland	Single	Derived from Commission for Environmental Cooperation's North American Environmental Atlas 2010 (Canada Centre for Remote Sensing (CCRS) et al. 2017)
Land cover	Grassland mean patch area	Single	Derived from Commission for Environmental Cooperation's North American Environmental Atlas 2010 (Canada Centre for Remote Sensing (CCRS) et al. 2017)
Land cover	Grassland patch cohesion index	Single	Derived from Commission for Environmental Cooperation's North American Environmental Atlas 2010 (Canada Centre for Remote Sensing (CCRS) et al. 2017)
Temporal	Year	Annual	Year
Topographical	Terrain ruggedness index	Single	Derived from digital elevation model (Riley et al., 1999)

estimate availability, defined here as the singing rate (Matsuoka et al., 2014). Availability and perceptibility are further defined in the Supporting Information.

We combined data from 2009 to 2014 in a single model per species to maximize the number of species with sufficient data for modelling, but allowed availability and perceptibility to vary among years. We modelled species with a minimum of 60 detections (Buckland et al., 1993). The number of grid surveys per species ranged from 79 (Swainson's Hawk) to 1391 (Western Meadowlark), with a median of 306 ± 333 SD surveys. To account for species absence or non-detection, we created absence records for all surveys at which each species was not observed.

While numerous methods for analysing point count data while correcting for imperfect detection exist, we chose to follow the “common standards” of Ralph et al. (1993) and Matsuoka et al. (2014), which partitions both minute of first observation (for availability) and distance (for perceptibility) into multiple bins that are applied to all species for consistency. Specifics of how we modelled availability and perceptibility, and their response to temporal and environmental variables, are provided in the Supporting Information.

We used the best-fitting model of each type to estimate the singing rate and effective area sampled (in ha), defined as the area within the effective detection radius, for each species at each survey

point. We multiplied the estimated singing rate and effective area sampled for each species and point and took the logarithm of the product to estimate a correction formula, per Sólymos et al. (2013). To estimate point-level density, we entered the correction factors as offsets in generalized linear mixed models calculated using R package *lme4* (Bates et al., 2015). We used a Poisson distribution and included primary habitat as a fixed effect and the unique combination of grid cell and year as a random effect. We left-truncated density estimates at 0.08 birds/km², equivalent to the density estimate for a point with a single bird observed at the maximum observed distance in the dataset (2,000 m). We extrapolated annual densities across the entire 1 km² area of all surveyed grid cells by taking the median of annual point-level densities (birds/ha) within each surveyed grid cell multiplied by 100.

2.7 | Species habitat modelling

We built species habitat models, with density as the response variable, for each species using boosted regression trees (BRTs), and used these to estimate densities of our 22 focal species in all 799,015 grid cells across the study region. BRTs are a machine-learning approach ideal for modelling complex species–environment relationships with

multiple predictors and are robust to correlations among predictors (Elith et al., 2008). Many species had skewed density distributions with absences in many grid cells (i.e., zero inflation), which violates the Poisson model assumption that the mean equals the variance. Therefore, we implemented a hurdle model approach in which we separately modelled occurrence and density, then combined the models to estimate density only at grid cells that met a threshold occurrence level. For the occurrence model, we used presence/absence as the response variable and for the density model detection-corrected abundance scaled to 1 km². For the density model, we first rounded densities and used a Poisson distribution. If these models failed to converge, we log-transformed estimated densities to improve fit and used a Gaussian distribution.

We used the 12 environmental variables described above plus year as predictor variables (Table 2). We iteratively tuned the regularization parameters to optimize model fit and used geographic filtering and spatially stratified cross-validation to reduce effects of spatial autocorrelation. Finally, we combined the occurrence and density models to project density across the study area. We calculated a minimum probability of occurrence threshold using maximum sensitivity + specificity in R package *SDMTools* (VanDerWal et al., 2014). We used this to mask median predicted density such that densities were retained only at grid cells that surpassed the threshold hurdle. Details of BRT model fitting are provided in Supporting Information.

2.8 | Conservation scores

We incorporated the breeding season Combined Conservation Score (CCS_b) from the State of North America's Birds (North American Bird Conservation Initiative, 2016) into the BFI as a weighting measure to account for each species' conservation status. We chose these scores as they were calculated consistently and rigorously for all native bird species across North America. CCS_b values ranged from 7 to 20 based on population size, distribution, threats and trends (Panjabi et al., 2012). We multiplied each species' CCS_b by predicted densities for each 1 km² grid cell to produce conservation-corrected densities, which we then summed across all 22 species. We explored scaling CCS_b values by dividing each score by the root mean square (range: 0.60–1.37), but BFI values calculated with scaled CCS_b were directly correlated with BFI values calculated with raw CCS_b values in each year ($\rho = 0.98$ – 0.99). Therefore, we used the raw scores in BFI estimation.

2.9 | Functional diversity

Functional diversity indices take a variety of forms, differing in the resolution of the trait information they include and the use of presence versus abundance data (Schipper et al., 2016; Schleuter et al., 2010). We implemented a two-step approach that used environmental filtering (Varela et al., 2014) to group species into

functional groups (Table 1) based on dietary, foraging and size-based traits (Table 3). We then calculated Shannon's diversity using functional group identities in place of species identity. The methods for calculating functional diversity are further detailed in the Supporting Information.

2.10 | BFI calculation and evaluation

We calculated BFI values by multiplying the summed conservation-weighted densities by functional diversity for each 1-km² grid cell. We chose to multiply rather than add densities and functional diversity, as the summed conservation-weighted densities are orders of magnitude larger than functional diversity indices and vary widely among grid cells. As a result, functional diversity information would be swamped if simply added to densities. Alternatively, raising densities to the power of functional diversity would greatly increase the range and skewness of BFI scores.

Finally, we standardized raw BFI values by annually scaling from zero to one using a logistic distribution, to accommodate the lower limit of zero and the occasional, exceptionally high BFI value. Standardization expresses each cell's bird-friendliness relative to the study area, producing a ranked index scaled from zero to one that is easily interpreted and compared across an extensive, climatically and topographically variable range. It also controls for large-scale factors such as climatic fluctuations or population-level density-dependent processes that influence population trends, thus isolating the effects of local management actions. For study area mapping, we standardized using all grid cells to enable comparison across the NGP as a whole. However, for the assessment of bird community response to simulated habitat management, we standardized using only grid cell scores from the surrounding spatial strata (e.g., South Dakota–Badlands and Prairies; Figure S1a) to highlight effects of management relative to nearby areas with similar climate and land cover.

We quantitatively evaluated the BFI in two ways. First, we quantified spatial variation by analysing differences in mean BFI values among strata across the study period using a generalized linear mixed model with strata as a fixed effect and year as a random effect. We explored the fit of four distributions (gamma, beta, lognormal and Gaussian) using R package *fitdistrplus* (Delignette-Muller & Dutang, 2015). As the beta distribution best approximated the distribution of BFI values, we fit the model in R package *glmTMB* (Brooks et al., 2017) and conducted Tukey's post hoc evaluations of differences among strata using *emmeans* (Lenth, 2020). Second, we evaluated the BFI's ability to identify ecologically significant areas for grassland and aridland birds by comparing BFI values among areas previously identified as priorities using other criteria. We used the Consensus Priorities identified by Grand et al. (2019), which identified overlaps between five prioritization scenarios: Grassland Climate Strongholds, Grassland Climate and Land use Strongholds, Vulnerable Grassland Climate Strongholds (all from Grand et al., 2019), Grassland Priority Conservation Areas (Pool &

TABLE 3 Functional traits used in environmental filtering to identify functional groups, including how traits were measured

Trait	Description	How measured
Body mass	Mean body mass	Grams
Diet: fruit	Per cent of the diet comprised of fruits	Estimated % use (0%–100%, 10% bins)
Diet: invertebrate	Per cent of the diet comprised of invertebrates	Estimated % use (0%–100%, 10% bins)
Diet: nectar	Per cent of the diet comprised of nectar, pollen, or other plant exudates	Estimated % use (0%–100%, 10% bins)
Diet: other plant material	Per cent of the diet comprised of other plant material, including grass, forbs, lichen, moss, cultivated crops and twigs	Estimated % use (0%–100%, 10% bins)
Diet: scavenged	Per cent of the diet comprised of scavenged material, garbage, offal, or carrion	Estimated % use (0%–100%, 10% bins)
Diet: seed	Per cent of the diet comprised of seeds, maize, nuts, spores, or grains	Estimated % use (0%–100%, 10% bins)
Diet: vertebrates, endotherms	Per cent of the diet comprised of birds or mammals	Estimated % use (0%–100%, 10% bins)
Diet: vertebrates, ectotherms	Per cent of the diet comprised of reptiles or amphibians	Estimated % use (0%–100%, 10% bins)
Foraging strata	Dominant foraging strata: ground, understorey, mid-storey, upperstorey, canopy or aerial	Categorical

Note: Trait data from Elton Traits 1.0 (Wilman et al., 2014).

Panjabi, 2011) and Grassland Potential Conservation Areas (Comer et al., 2018; Figure S1c). There were no overlaps between all five prioritizations; therefore, we compared mean BFI values among areas included in 0–4 prioritizations using a beta regression in *betareg* (Cribari-Neto & Zeileis, 2010) with the number of prioritizations used as a categorical fixed effect, and conducted Tukey's post hoc evaluations using *emmeans* (Lenth, 2020).

2.11 | Sensitivity analyses

We conducted a sensitivity analysis to assess the relative influence of species density, conservation score and functional diversity on BFI values. We used bootstrapping (1000 iterations) to evaluate the influence of increasing the variability (here, standard deviation) of each component of the BFI on the index value. Details of the sensitivity analysis methods are provided in the Supporting Information.

2.12 | Habitat management case study

We simulated an evaluation of the effects of management actions on a theoretical private property, for example an Audubon Conservation Ranch. In these simulations, we modelled the effects of common bird-friendly habitat management actions including reducing cropland cover and N₂O and increasing litter biomass, net primary productivity, soil moisture, proportion grassland, grassland cohesion and grassland patch area. Our selection criteria for using these covariates are explained in the Supporting Information.

We used observed covariate values (Table 2) to produce simulated temporal change that accumulated incrementally over time at a

rate of 10% per year, plus random variation. We then used these simulated environmental conditions to re-calculate bird densities and functional diversity and re-estimate the BFI each year within that property. We simulated habitat improvement beginning in 2011 to simulate a before–after control–impact design with monitoring for two years prior to management. We then compared changes in simulated BFI values post-management to actual, estimated BFI from 2009 to 2014.

3 | RESULTS

3.1 | Bird-Friendliness Index estimation

3.1.1 | Estimated density

Densities estimated at surveyed grid cells for the 22 species during 2009–2014 ranged from 0.02 ± 0.00 (Swainson's Hawk) to 28.24 ± 0.60 (Western Meadowlark; Table 1). The five most abundant species observed across the study area were Western Meadowlark, Grasshopper Sparrow, Horned Lark, Lark Bunting and Vesper Sparrow.

3.1.2 | Species habitat modelling

Fit of species habitat models varied among species and data type (presence/absence versus density), but the predictors generally explained 30%–40% of the variation in the data after cross-validation using geographic block partitioning. For presence/absence models, cross-validated AUC averaged 0.71 ± 0.03 SE (range: 0.62–0.99),

TSS averaged 0.25 ± 0.04 SE (range: 0.05–0.72), correlation averaged 0.28 ± 0.03 (range: 0.09–0.61) and deviance explained averaged 0.32 ± 0.03 (range: 0.12–0.77; Table S2). For density models, cross-validated correlation averaged 0.27 ± 0.03 (range: 0.06–0.61) and deviance explained averaged 0.37 ± 0.04 (range: 0.01–0.72; Table S2). All Moran's I values were ≤ 0.11 , indicating that the predictors fully accounted for spatial autocorrelation in the data (Table S2).

Climate and land cover variables explained the most variation in occurrence and density of the grassland and aridland birds studied here. Climatic moisture deficit explained the most variation in occurrence (mean \pm SE: $17.10 \pm 3.54\%$), followed by proportion cropland (mean \pm SE: $13.60 \pm 3.28\%$) and proportion grassland (mean \pm SE: $12.51 \pm 2.47\%$; Figure 2a). Proportion cropland cover explained the most variation in density (mean \pm SE: $17.58 \pm 4.92\%$), followed by climatic moisture deficit (mean \pm SE: $13.42 \pm 3.24\%$) and proportion grassland (mean \pm SE: $9.18 \pm 1.94\%$; Figure 2b). Year explained the least variation in both occurrence and density. Though densities and occurrence frequencies varied among years, environmental predictors that varied across space and, in some cases, time explained more variation than year alone. Relative variable importance varied among species consistent with species-specific habitat preferences; for example, Rock Wren density was explained most by terrain ruggedness, while occurrence and density of grassland birds like Sprague's Pipit and Western Meadowlark were explained most by grassland proportion cover and patch area (Figures S2–S23). Drivers of grassland and aridland bird occurrence and abundance are further discussed in the Supporting Information.

3.1.3 | Functional diversity

Functional diversity averaged 0.72 ± 0.00 SE (range: 0.00–1.79) across the Northern Great Plains during 2009–2014

(Table S3). Functional diversity showed greater variation among strata (range: 0.37–1.02) than among years (range: 0.68–0.79). Functional diversity was greatest in the Dakotas and western Nebraska and lowest in Wyoming and southern Montana.

3.1.4 | Bird-Friendliness Index

BFI values showed large- and fine-scale regional variation in patterns that were generally consistent over time, though BFI values were higher in the Dakotas and lower in the Prairie Potholes in 2014 than earlier years (Figure 3). The most resilient grassland and aridland bird communities were found in the Prairie Potholes region of North Dakota (ND-11, mean \pm SE: 0.72 ± 0.00 ; ND-17: 0.62 ± 0.00), Alberta (AB-11: 0.63 ± 0.00), Saskatchewan (SK-11: 0.59 ± 0.00) and South Dakota (SD-11: 0.54 ± 0.00 ; SD-17: 0.52 ± 0.00 ; Figure 4a). Conversely, the lowest BFI values were found in the south-western regions of the study area, notably Wyoming (WY-10: 0.23 ± 0.00 ; WY-17: 0.30 ± 0.00) and southern Montana (MT-10: 0.26 ± 0.00)

Mean BFI values were significantly different between all pairs of strata ($p < .001$) except for the two Nebraska strata (NE-18 and NE-19, $p = .98$; Figure 4a).

BFI values were higher in areas identified by one or more previous grassland conservation prioritizations and increased with the number of prioritizations the area was included within (Figure 4b). Mean BFI values were highest in places where at least four grassland prioritizations concurred (mean \pm SE: 0.68 ± 0.00), followed by areas where at least three prioritizations concurred (0.57 ± 0.00), two prioritizations (0.54 ± 0.00), one prioritization (0.48 ± 0.00) and no prioritizations (0.42 ± 0.00). All pairwise comparisons were significant after adjustment for multiple comparisons ($p < .0001$; Figure 4b).

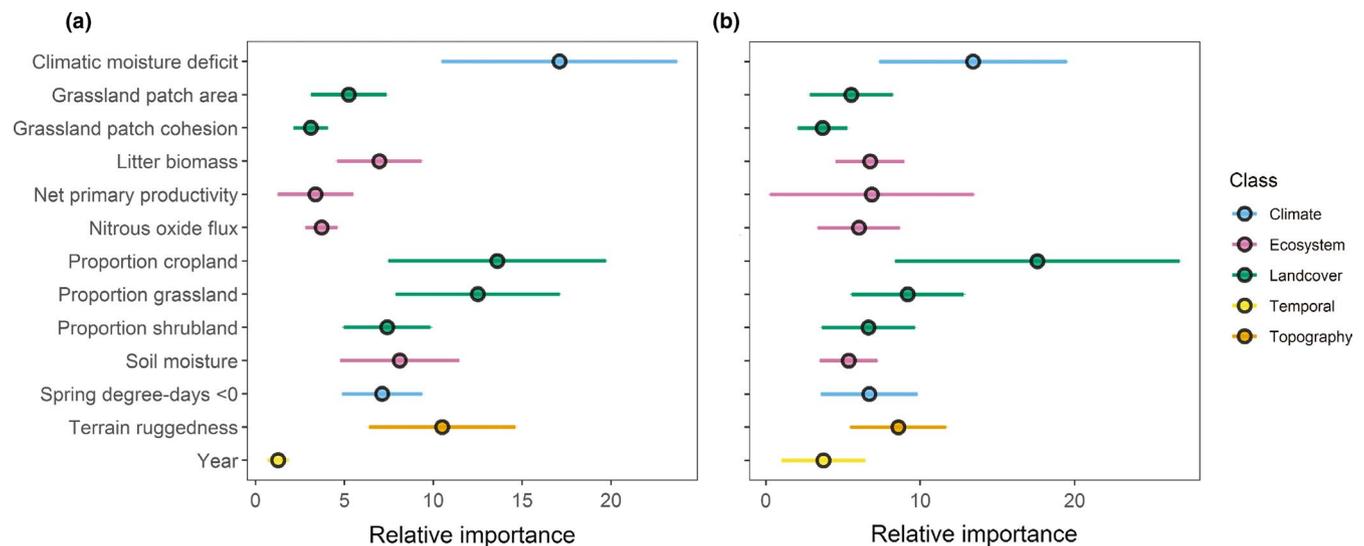


FIGURE 2 Mean variable importance and 95% confidence intervals for 12 variables used as predictors in presence/absence (a) and density (b) species habitat models for 22 grassland and aridland bird species across the Northern Great Plains during 2009–2014. Climate and land cover variables explained the most variation in occurrence and density

3.1.5 | Sensitivity analyses

Sensitivity analyses revealed that BFI values were most sensitive to variation in functional diversity (Figure 5). BFI calculated with resampled functional diversity overlapped with original BFI values by $72.69 \pm 0.03\%$, while resampling bird densities produced $83.88 \pm 0.02\%$ overlap, and resampling conservation scores produced $96.43 \pm 0.02\%$ overlap (Figure 5).

3.2 | Case study: effects of land management practices on grassland and aridland bird communities

BFI values representing grassland and aridland bird community response to simulated management were 47% higher in 2014 than BFIs estimated from observed data (Figure 6). Additionally, BFI values with simulated bird-friendly habitat management significantly increased over the six-year period (slope = 0.06 ± 0.03 SE, $p = .04$), while estimated BFI values (without bird-friendly habitat management) did

not change during the same time period (slope = -0.02 ± 0.01 SE, $p = .19$). This suggests that practices recommended for use in bird-friendly grassland and aridland habitat management plans will increase the abundance and resilience of the bird community and will be detected using the BFI.

4 | DISCUSSION

The ability to quantify the impacts and success of conservation and management actions is crucial to the adaptive management cycle (Nichols & Williams, 2006). Here, we show that the Bird-Friendliness Index (BFI) served as a proxy for identifying ecologically significant areas for grassland and aridland birds, with mean BFI values increasing with the degree of consensus across published grassland conservation prioritizations. Moreover, the BFI detected regional variation in community richness and functional diversity, further supporting its utility as a metric for identifying ecologically significant areas for grassland and aridland birds. The BFI also effectively detected

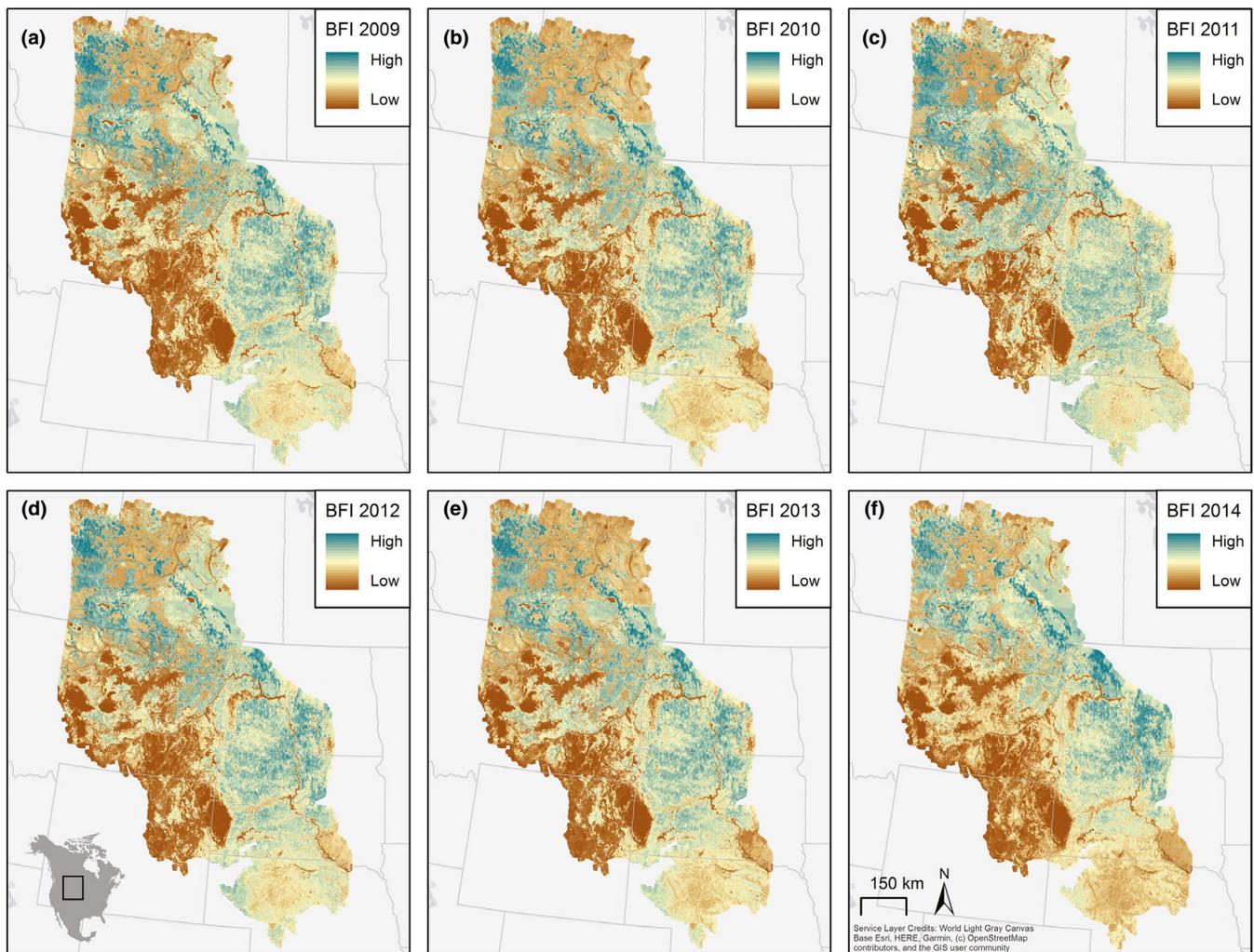


FIGURE 3 Bird-Friendliness Index across the Northern Great Plains study area for 2009–2014 (a–f). BFI values were highest in Canada, northern Montana and the Dakotas and lowest in southern Montana and Wyoming. BFI values showed little variation over time

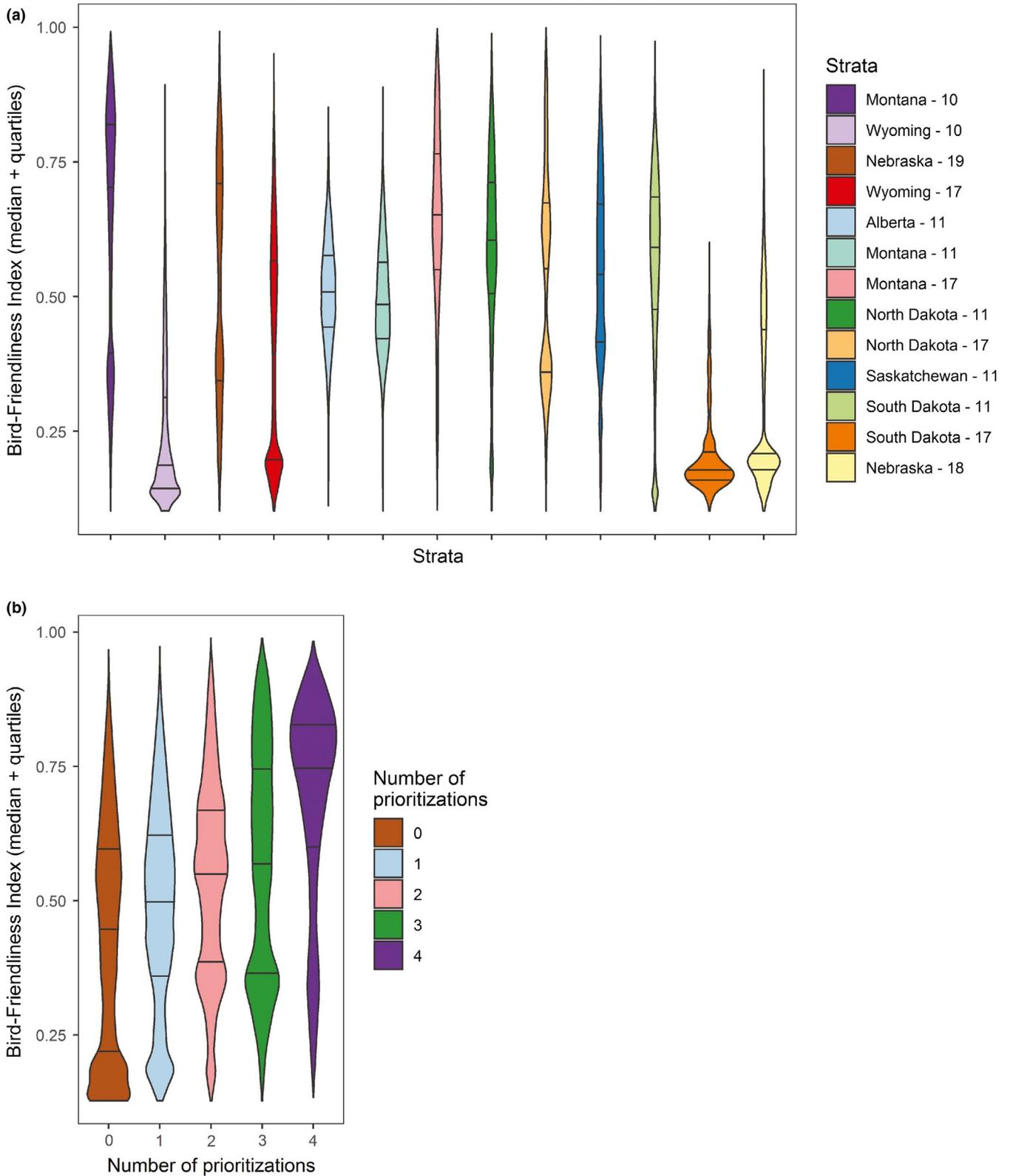


FIGURE 4 Probability densities of Bird-Friendliness Index values by strata (a; unique combinations of states or provinces and Bird Conservation Regions) and the number of grassland conservation prioritizations the area was included within (b). Lines represent median and quartiles of BFI distributions (25%, 50%, 75%). Letters represent statistical significance of comparisons among mean BFI values by strata (a), and number of prioritizations (b). Strata (a) are arranged left to right in the same order as in the legend

the response of grassland and aridland birds to simulated habitat management, ensuring its ability to provide accountability and transparency for implementation of grassland and aridland habitat enhancement. This, in turn, can inform selection or adaptation of habitat management practices for the subsequent year, thus informing the adaptive management process.

The most resilient (as revealed by high BFI scores) grassland and aridland bird communities were found in the Prairie Potholes region of the Dakotas, Alberta and Saskatchewan. Functional diversity (FD) values – to which BFI values were most sensitive – were similarly highest in the Dakotas, but the highest functional diversity was observed in South Dakota (SD-11, mean BFI = 0.54, mean FD = 1.02), while Alberta had the second-highest BFI value but the seventh-highest functional diversity (AB-11, mean BFI = 0.63, mean FD = 0.76). This difference highlights the interplay between bird densities, conservation weights and FD in BFI estimation; South Dakota had a more balanced and diverse representation of functional traits but lower densities in its grassland and aridland bird community, whereas Alberta had higher densities overall—or higher densities of species of conservation concern—but lower FD. Conversely, the lowest BFI values were found in the south-western regions of the study area, notably southern Montana, Wyoming and southern South Dakota. These regions tend to be drier with more sparse vegetation, as well as lower soil moisture and productivity. While these regions provide critical habitat for aridland bird species such as Brewer's Sparrow, they are less suitable for grassland birds that prefer lush and more productive habitats (Fedy et al., 2018; Fisher & Davis, 2010; Harrower et al., 2017; Renfrew et al., 2013). As a result, they have fewer bird species, many of which share

similar functional traits (e.g., ground-dwelling insect- and granivores; Table 1). Consequently, aridlands have lower functional diversity and lower resilience (Figure 5).

The BFI was consistent with recent prioritizations using different methods, datasets and assumptions. Prairie Pothole regions with concentrations of high BFI values coincide with Grassland Potential Conservation Areas identified in the N Great Plains Fescue Mixed-grass and NW Great Plains Mixed-grass Prairie grasslands (Comer et al., 2018). Similarly, many Grassland Priority Conservation Areas (Pool & Panjabi, 2011) and Climate and Land Use Strongholds (Grand et al., 2019) were identified across the US and Canadian Prairie Potholes, coinciding with areas of high BFI values. BFI values were significantly higher where there was consensus across one or more of these prioritizations and increased further with growing consensus.

In addition to identifying ecologically significant areas consistent with other, independent prioritizations, the BFI was also able to detect the response of grassland and aridland birds to simulated habitat management. In our case study, we simulated the effects of bird-friendly habitat management practices based on published relationships between environmental predictors used in the species habitat models and management practices such as reducing cropland cover, restoring grassland and reducing grazing intensity. These practices directly impact land cover metrics, while grazing intensity increases nitrous oxide (N₂O) emissions (Allard et al., 2007). By simulating increases (e.g., grassland cover and patch area) and decreases (e.g., cropland cover and N₂O) in these predictors, we were able to generate predicted density estimates under bird-friendly management, and use these to derive functional diversity and, ultimately, the BFI. This exercise shows great promise for habitat management

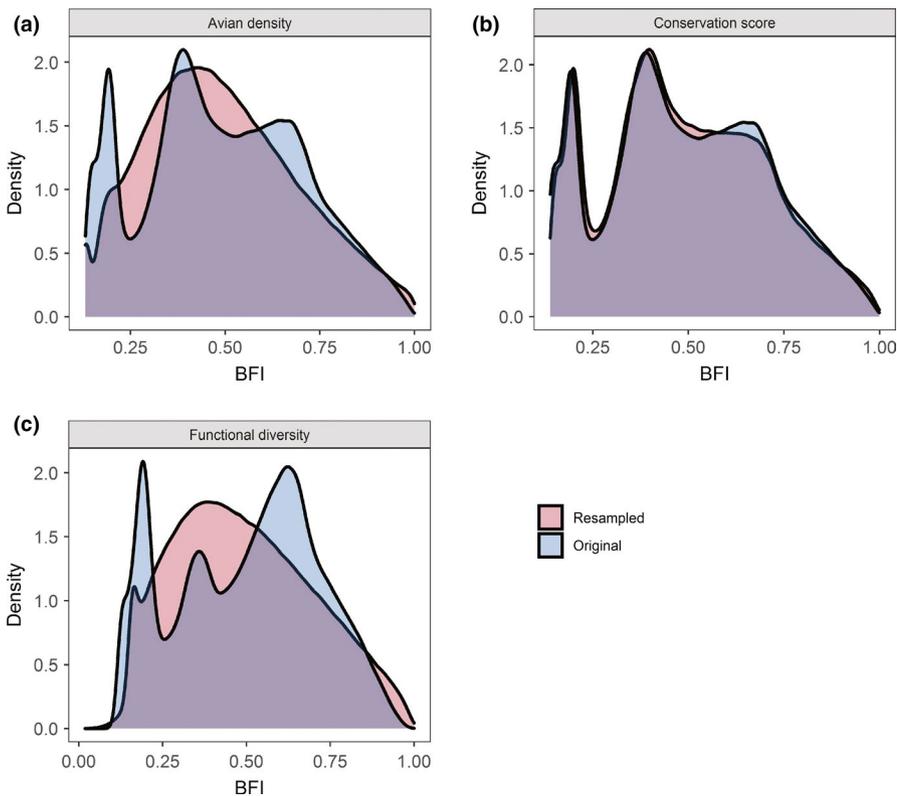


FIGURE 5 Results of sensitivity analyses evaluating the relative contribution of densities (a), conservation scores (b) and functional diversity (c) to the BFI. The distribution represented by the resampled data is depicted in pink and the distribution represented by the observed data is depicted in blue, with the overlap shown in purple. Variation in functional diversity had the greatest influence on the BFI

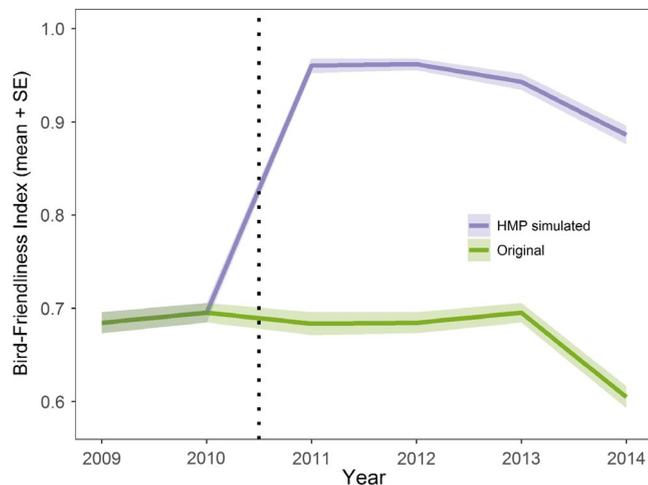


FIGURE 6 Case study showing estimated and simulated BFIs for a hypothetical management area within the northern Great Plains. Simulated BFIs representing the response to hypothetical bird-friendly habitat management implemented during 2011–2014 were higher overall than estimated BFIs from observed data. Mean (lines) and SE (ribbons) are shown

programmes that implement similar practices to protect grassland and aridland birds.

Implementation of the BFI to evaluate actual bird-friendly management practices will enable us to further refine our understanding of grassland and aridland bird response to bird-friendly management. The National Audubon Society (Audubon) has deployed the BFI as an accountability metric for its Conservation Ranching programme (<https://www.audubon.org/conservation/ranching>). This programme is a market-based conservation solution that aims to conserve imperilled grassland and aridland bird species and habitats through partnership with private landowners. Other public/private grassland and aridland conservation efforts, for example, World Wildlife Fund's Grasslands programme (<https://www.worldwildlife.org/habitats/grasslands>), Sodsaver (enacted as part of the Agricultural Act of 2014) and Natural Resources Conservation Service's EQIP (<https://www.nrcs.usda.gov/wps/portal/nrcs/main/national/programs/financial/eqip/>), could benefit from the accountability provided by this metric. Similarly, other bird-friendly products—even those from different habitats, for example shade-grown coffee—could benefit from incorporating an accountability metric that assures consumers that their decisions are demonstrably benefiting birds. The BFI could also be expanded to apply to other habitats and guilds that can be surveyed using methods that produce density, or even relative abundance, estimates. Audubon has already used a similarly designed “ecological integrity index” to evaluate communities of wetland and forest birds in Colombian National Parks (Wilsey, Michel, et al., 2019; Wilsey, Taylor, et al., 2019).

There are many opportunities for further refinement and application of the BFI. Additional bird survey data, through inclusion of additional years and/or an expanded survey area, would enable estimation of robust density estimates for additional grassland and

aridland bird species, enabling the BFI to represent a broader slice of the community. Further, we were unable to incorporate annual land cover in this case study due to the lack of robust annual land cover classifications that span the US/Canada border. Future applications of the BFI should take advantage of annual land cover data, where available, because of these inherent sensitivities. We designed the BFI for the purpose of conducting spatial comparisons within a single time period, or evaluating site-level trends in resiliency relative to other sites or the region as a whole, but not for evaluating population trends at the species or region level. Our standardization approach enabled us to isolate local-scale effects, such as bird response to habitat management or regional differences in land use, from large-scale processes such as long-term population-level declines due to shared drivers such as climate oscillations or wintering ground effects (Gorzo et al., 2016; Macías-Duarte et al., 2017). However, the standardization step could be removed for studies with an objective of tracking long-term trends in ecological resiliency across large spatial scales, rather than evaluating local response to conservation actions.

The BFI serves as a tool for quantifying grassland and aridland bird community response to management, which enables its use to inform a robust adaptive management process (Lancia et al., 1996). Conservation efforts should aim to do more than prevent the extinction of species, but rather should be aimed at preventing species from becoming threatened in the first place, as well as providing conditions that enhance ecosystem processes (Rodrigues, 2006). Under this framework, scientists and land managers would work together to refine habitat management protocols to increase abundance and functional diversity—and consequently resiliency—of grassland and aridland bird communities on privately managed lands. To be able to do this effectively requires indicators that are rigorous, repeatable and easily understood (Balmford et al., 2005).

Recent grassland and aridland bird population declines call for implementation of new conservation and restoration efforts that demonstrably improve habitat conditions and slow or reverse declines. By identifying early on which species and communities are doing well or poorly, and where, the BFI can pinpoint priority strongholds for conservation, or opportunities for restoration, both of which may contribute to population stabilization or even growth. In short, the BFI is a tool that can be used by conservationists and managers to develop and measure accountable conservation now and into the future.

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CONFLICT OF INTEREST

None.

PEER REVIEW

The peer review history for this article is available at <https://publons.com/publon/10.1111/ddi.13163>.

DATA AVAILABILITY STATEMENT

The R scripts used to conduct the analyses and simulations are available through Data Dryad at <https://doi.org/10.5061/dryad.gxd2547jj>. Integrated Monitoring in Bird Conservation Regions data are publicly available through the Rocky Mountain Avian Data Center: <http://rmbo.org/v3/avian/Home.aspx>.

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BIOSKETCH

The authors of this research consist of avian conservation biologists at National Audubon Society and a partner consulting firm. Our research priorities include delineating trends and spatial patterns in bird abundance, occupancy and occurrence; identifying climate and habitat relationships; providing the science needed to inform conservation actions; and evaluating population- and community-level responses to conservation and management actions.

Author contributions: M.H. and A.H. developed the initial concept for the Bird-Friendliness Index (BFI). N.M., C.B., C.W. and G.L. developed the initial structure of the BFI index. C.B. produced preliminary analyses and report. N.M. conducted all analyses, prepared tables and figures and wrote the manuscript.

SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.

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