Bison were instrumental in shaping North America’s Great Plains. Interest in restoring this iconic species and their ecological role in grassland ecosystems is rapidly gaining momentum. To evaluate the potential for bison to enhance habitat quality for wildlife and catalyze public engagement in grassland conservation, we assessed both the ecological and social effects of a recent bison reintroduction (2015) to northern Colorado. Specifically, we explored the effect of bison reintroduction on: (a) bird density and habitat use, (b) mammal habitat use, (c) vegetation composition and structure, and (d) visitor connectedness, known as place attachment, to a shortgrass prairie. We predicted that bison reintroduction would reduce cover and height of some grasses and shrubs, which would increase density and habitat use for obligate shortgrass prairie birds, and increase habitat use for coyote and lagomorphs. In addition, we predicted that visitors would express stronger place attachment to this grassland once bison were reintroduced. To measure ecological and social responses, we surveyed birds, mammals, and plants; and conducted structured visitor surveys before and after bison reintroduction. We found few short-term effects of bison on grassland bird density and habitat use, mammal habitat use, and vegetation composition and structure. However, we measured a significant increase in visitor place attachment to the grassland site 1 year after bison reintroduction. Our results suggest that a new bison reintroduction may have immediate positive benefits for connecting people to conservation, and that the ecological and social effects may unfold over different time scales. We recommend that future bison reintroduction efforts monitor ecological and social outcomes to advance reintroduction biology.

**KEYWORDS**

bird communities, ecological restoration, grassland conservation, place attachment, reintroduction, shortgrass prairie

---

**INTRODUCTION**

The decline or extinction of animals, known as defaunation, has important consequences for natural communities and human well-being (Dirzo et al., 2014). The loss of “mega-herbivores” has threatened ecological and evolutionary interactions across the globe (Donlan et al., 2005), and disrupted sociocultural systems that co-evolved with those species (Isenberg, 2000). Today, large herbivores that remain functionally extinct once served as ecological engineers by shaping trophic guilds (Fritz, Duncan, Gordon, & Illius, 2002) and contributing to species diversity and abundance (Ogada, Gadd, Ostfeld, Young, & Keesing, 2008). Reintroduction has the potential to restore ecological communities and reestablish important sociocultural connections between people, wildlife, and landscapes.
Plains bison (*Bison bison*) were once the dominant large herbivore in North American prairies, but currently occupy less than 1% of their historic range (Hedrick, 2009). Along with natural fire regimes, bison-shaped biodiversity, and ecosystem services in prairie ecosystems (Samson, Knopf, & Ostlie, 2004). Bison hunting on the Great Plains sustained one of North America's longest running socioecological systems—the Plains Indians’ bison economy (Flores, 1999). These habitats and ecosystem services have been lost over time due to industrial agriculture, resource exploitation, and the large-scale loss of native grazing animals (DeLuca & Zabinski, 2011). As a keystone species, bison could help restore these service by altering plant community composition (Knapp et al., 1999; Towne, Hartnett, & Cochran, 2005), changing soil nutrient cycling (Frank & Evans, 1997), and causing shifts in bird species richness (Griebel, Winter, & Steuter, 1998), bird abundance (Powell, 2006), and small mammal abundance (Matlack, Kaufman, & Kaufman, 2001). Despite their critical contributions to land and wildlife health, bison have been nearly extirpated from North America. Some argue that this range contraction has rendered bison ecologically extinct (Freese et al., 2007)—meaning they no longer serve the same foundational role in grassland ecosystems. Popular and political interest in restoring this iconic species for cultural and ecological benefit is rapidly gaining momentum (Steenweg, Hebblewhite, Gummer, Low, & Hunt, 2016).

As charismatic, native mega-herbivores, bison are an ideal species for advancing a cross-disciplinary understanding of reintroduction. These grazers are popular with the public, and could serve as a flagship species or focal species for grasslands conservation (Walpole & Leader-Williams, 2002). While flagships can serve as an important conservation tool, these species may also be associated with greater human-wildlife conflict (Woodroffe, Thirgood, & Rabino-witz, 2005). Yet, few studies examine the realities of coexistence with reintroduced species for local communities and other stakeholders (Seddon, Armstrong, & Maloney, 2007). A review of the recent species reintroduction literature found that social factors, such as human attitudes and perceptions of reintroductions, account for only 4% of 454 papers in this body of work (Seddon et al., 2007). Thus, increasing knowledge about the ecological and human dimensions of reintroduction is critical to achieving successful species reintroductions.

We investigated whether ecological and social effects were evident in the 2 years following a bison reintroduction in northern Colorado. Previous studies assessing the effects of bison reintroduction on plants and animals have generally occurred over longer time scales (4–10 years or more) at sites with higher densities of bison (1.2–1.7 animal units/ha; Griebel et al., 1998; Matlack et al., 2001; Towne et al., 2005). At our study site, bison densities averaged 0.05 animal units/ha following bison reintroduction in 2015. Our specific research questions evaluated the short-term effects of bison reintroduction on: (a) bird habitat use and density, (b) mammal habitat use, and (c) visitor place attachment to a shortgrass prairie.

## 2 METHODS

### 2.1 Study area

We studied the ecological effects of reintroducing a single bison herd to Soapstone Prairie Natural Area and Red Mountain Open Space (hereafter, Soapstone and Red Mountain) located approximately 48 km north of Fort Collins, Colorado, United States (Figure 1). Elevation in the study area ranges from 1,219 to 2,200 m and 70% of the area is classified as shortgrass prairie.

The study area was grazed by large herds of bison up until approximately 100 years ago (Isenberg, 2000), when homesteaders arrived and began to graze sheep and cattle (Martin, Harness, & Harness, 2009). In November 2015, 11 bison were reintroduced to a fenced 393-ha pasture (hereafter “bison site”; Figure 1). The bison site was not grazed by cattle for 5 years prior to the reintroduction (J. Frederickson, personal communication). The herd has since tripled, with 54 bison grazing the site as of July 2018. Bison were not confined to a central feeding area during the winter and were free-ranging through large portions of the property year-round. Several areas (308 ha) near the bison pasture have only infrequently been grazed by cattle for approximately 10 years (hereafter “reference sites”).

To assess potential ecological changes, we surveyed birds, mammals, and plants at bison and reference sites from May through November 2015 (pre-bison reintroduction) and May through November 2016 and 2017 (post-bison reintroduction). To understand how bison reintroduction shaped visitor place attachment to this shortgrass prairie, we conducted structured surveys with visitors to Soapstone. Visitors were intercepted from June through October before (2015) and after (2016) the bison reintroduction. Our assessment of bison reintroduction’s effects on natural area visitors was restricted to Soapstone because the bison are not visible from Red Mountain’s roads or trails.

### 2.2 Bird surveys

To estimate bird habitat use and density, we randomly selected point count locations within the bison (*n* = 20) and reference (*n* = 14) sites. The point count locations (Figure 1) were buffered 200 m from fences and stands of trees (Fuhlendorf et al., 2006), and spaced 200–250 m to minimize the likelihood of double-counting individuals.
We excluded stands of trees to restrict our data collection to facultative or obligate grassland bird species. This resulted in 14 points split between two reference sites. Birds were surveyed between 5:30 and 10 a.m. from May–June, 2015–2017. We identified all bird species in 5-min intervals visually or aurally (Hanni et al., 2014). Using a rangefinder, we measured the distance (m) between the observer and each bird. Each bird point count location was surveyed five times per field season at the bison site (n = 100 surveys) and the reference sites (n = 70 surveys) to account for imperfect detection in occupancy (habitat use) models (Bailey, Hines, Nichols, & MacKenzie, 2007). We estimated wind speed, rainfall, and cloud cover during each survey using standard bird monitoring protocols (Hanni et al., 2014).

2.3 Mammal surveys
We used remotely triggered wildlife cameras (Cuddeback Long Range IR Trail Camera, Cuddeback Capture, Bushnell Primos Truth Cam 35, and Cuddeback Attack IR 1156), to estimate habitat use by mid- to large-sized mammals before and after the bison reintroduction. We evenly distributed different camera models among the sites (Figure 1). To select locations for wildlife cameras, we used ArcGIS software to divide bison and reference sites into 200 × 200 m grids. We then randomly selected 20 grids at each site, and identified areas within each grid that had signs of wildlife (e.g., trails and scat). We placed cameras at least 200 m apart to maximize detection (O’Connell & Bailey, 2011). We did not buffer cameras from fences, but if we placed a camera near a fence dividing different sites, we oriented the camera toward the site of interest. We placed cameras 60–80 cm above the ground on posts hammered into the ground. Wildlife cameras at bison (n = 19) and reference sites (n = 20) operated from May–November, 2015–2017. We replaced batteries and SD cards every 2–4 weeks based on the camera type and weather. We downloaded photographs from each camera monthly and uploaded photos to the CPW Photo Warehouse program (Newkirk, 2016). At least two observers viewed each photo and identified all mammals to species. Discrepancies in species identification were resolved by the lead author (K.W.).

2.4 Vegetation surveys
To measure plant composition and structure (June–July, 2015–2017), we established one 50 m transect at each wildlife camera and point count location in bison (n = 40) and reference sites (n = 38). We used a Daubenmire frame (Daubenmire, 1959) and modified Robel pole (Vinton, Hartnett, Finck, & Briggs, 1993) to estimate percent canopy cover and height, respectively, every 10 m along each transect. We placed the Daubenmire frame to the right of the transect tape and alternated sides of the tape every 10 m. Within each frame, we recorded the percent canopy cover of bare ground, litter, rock, grasses, forbs, and shrubs with non-overlapping percentages (Fletcher & Koford, 2002).
identified all grasses, forbs, and shrubs to species. For this study, we were unable to discern whether wallowing activity from the bison created the patches of bare ground in the bison site after reintroduction. To measure vegetation structure, we placed the modified Robel pole (3.4 cm PVC pipe, 1 m tall, 1 cm increments marked by alternating black and white bands) in the center of each Daubenmire frame. To estimate vegetation height, we observed the pole from each cardinal direction (N, S, E, W) at a distance of 4 m and a height of 1 m (Robel, Briggs, Dayton, & Hulbert, 1970). We also counted and identified to species all shrubs and sub-shrubs that occurred within 1 m of each transect.

2.5 | Visitor surveys

Place attachment refers to the emotions, meanings, and values people tie to specific places, including protected areas (Folmer, Haartsen, & Huigen, 2013). Higher levels of emotional connection to place have been associated with pro-environmental behavior (Halpenny, 2010). In addition, recent scholarship has identified connections between viewing wildlife and positive emotional attachment to protected areas (Kil, Holland, & Stein, 2010). To better understand how bison reintroduction affected visitor place attachment to Soapstone, we conducted structured visitor surveys to compare visitor demographics, place attachment, and motivations for visiting the study area before and after bison reintroduction. Weekday visitation to Soapstone is very low. Thus, to ensure a sufficiently large sample size, we intercepted every vehicle at the public entrance gate to Soapstone on weekends during peak visitation (June through October) in 2015 and 2016. We surveyed one person per vehicle that pulled over and agreed to take the survey.

The survey instrument consisted of a structured 44-item questionnaire containing open- and close-ended questions. Researchers conducted 5- to 10-minute visitor surveys using the data collection software, iSurvey (Contact Software Limited, York Bay, Lower Hutt, New Zealand). The survey instrument asked visitors about their demographics, place attachment to Soapstone, and whether Soapstone was important to them. The place attachment scale included four 5-point Likert items based on a rating scale developed and field tested by Folmer et al. (2013). Visitors were asked to indicate how much they agreed or disagreed with this series of four statements (see Supporting Information). Each Likert item was ranked and the respondent's average score provided an overall measure of their visitor place attachment, or the level of connection visitors felt, to Soapstone. Visitors were also asked to respond to two survey questions: “Is Soapstone important to you?” and, “If yes, why?” These follow-up questions were designed to elucidate reasons for visitors’ place attachment to Soapstone, including how charismatic flagship species like bison may shape visitor affinities for grassland landscapes (Walpole & Leader-Williams, 2002). Our survey instrument, and social science research design, were approved by Colorado State University’s Institutional Review Board (Protocol: 15-5816H).

2.6 | Data analysis

2.6.1 | Bird density

To determine whether bird species densities differed before and after bison reintroduction, we employed a two-stage approach (see Supporting Information; Buckland, Rexstad, Marques, & Oedekoven, 2015). For the first stage, we used program distance (Thomas et al., 2010) to get estimates of detection probability and the effective detection radius. In the second stage (see Supporting Information), the effective detection radius served as an offset in generalized linear mixed effects models built with the Rdistance package (McDonald, Nielson, & Carlisle, 2015) to estimate bird densities. For species with detection probability models that did not converge or meet our criteria for the goodness-of-fit test (see Supporting Information) in program distance, we estimated occupancy (habitat use). Three obligate grassland bird species met our criteria for estimating density: Horned Lark (Eremophila alpestris), Vesper Sparrow (Pooecetes gramineus), and Western Meadowlark (Sturnella neglecta). We used Akaike’s information criterion (AIC) to rank models of bird detection probability and density produced in program distance and R-Studio, respectively. We report information on competing models with a $\Delta$AIC <2.0, and AIC weights (Burnham & Anderson, 2002).

2.6.2 | Bird habitat use

To determine whether habitat use by birds differed before and after bison reintroduction, we used a dynamic occupancy modeling framework (MacKenzie, Nichols, Hines, Knutson, & Franklin, 2003; see Supporting Information) using the colext function in R’s unmarked package (Fiske & Chandler, 2011). We compared models using the AIC model selection process described above. Our results reflect the probability of “use” (Farr, Pejchar, & Reed, 2017) as opposed to true occupancy because the data collected did not meet all assumptions required to estimate true occupancy (O’Connell & Bailey, 2011). Thus, we refer to results from these analyses as “habitat use.” Bird species that met our criteria included Brewer’s Blackbirds (Euphagus cyanocephalus), Grasshopper Sparrows (Ammodramus savannarum), and Lark Sparrows (Chondestes grammacus).

2.6.3 | Mammal habitat use

To determine the effect of bison reintroduction on habitat use by mammals (see Supporting Information), we selected photos collected in the summer (June–September 2015–2017) to ensure that all mammals were resident in the system during the survey period (O’Connell & Bailey, 2011). We defined a sampling occasion in our analysis as 7 days, with each set of 7 days separated by a 24-hr rest period to maintain independence between occasions.
(Shannon, Lewis, & Gerber, 2014). We compared models using the AIC model selection process described for birds above.

### 2.6.4 Plant community characteristics

To assess the effect of bison reintroduction on plants, we averaged vegetation cover and height data for cool and warm season grasses, forbs, shrubs, and bare ground from point count stations and wildlife cameras at the bison and reference sites. We then built linear mixed effects models using the individual location (camera or point count station) as a random effect in the model. Response variables included vegetation type, and covariates included site (bison or reference) and year. We ranked models using the same AIC process described previously.

### 2.6.5 Visitor place attachment surveys

To assess the effects of bison reintroduction on visitor place attachment, we first tested the internal reliability of the scale items (see Supporting Information). We then averaged the scores and performed a Welch's two-sample, unpaired, one-sided t-test to test the hypothesis that visitors’ overall mean place attachment score would be higher after bison reintroduction (Bernard, 2011; see Supporting Information). For the open-ended question, “Why is Soapstone important to you?”, we inductively coded responses into themes using NVivo Pro 11. To quantify themes, we calculated the percent of visitors that mentioned each theme (Bernard, 2011; see Supporting Information).

### 3 RESULTS

#### 3.1 Bird density and habitat use

Across all sites and years, we observed 50 species of birds (Supporting Information, Table S7). For bird species with sufficient detections for analysis, we report densities and habitat use (Figure 2), estimates of detection probability, and the variables that appeared in the top model for all modeled species (see Appendix S1). We found no strong or consistent effect of bison reintroduction on birds (Figure 2). Top models for bird density or habitat use did not include site by year interactions (Tables S8-S14). Horned Lark and Vesper Sparrow densities remained constant over time at bison and reference sites (Figure 2). Western Meadowlark densities and Lark Sparrow and Brewer’s Blackbird habitat use also did not change at the bison site relative to the reference site (Figure 2). The proportion of sites occupied by Grasshopper Sparrows in the bison pasture increased after bison reintroduction (Figure 2).

#### 3.2 Mammal habitat use

Across all sites and years, we observed 14 species of mammals (Table S15). The species or taxa with a sufficient number of detections for occupancy analyses included coyote (Canis latrans), mule deer, and lagomorphs: black-tailed jackrabbits (Lepus californicus), white-tailed jackrabbits (Lepus townsendii), and desert cottontails (Sylvilagus auduboni). Bison reintroduction did not affect habitat use of coyote, and models with site by year interactions for colonization and extinction probabilities did not converge for any species. We observed a decreasing trend in habitat use for lagomorphs and mule deer at the bison site compared to the reference site (Figure 2), and mule deer extinction probabilities were higher at the reference site compared to the bison site (Tables S16 and S17).

#### 3.3 Plant community characteristics

We documented 19 grass species, 40 forb species, and 14 shrub species at bison and reference sites (Table S18). Both warm and cool season appeared along the 50 m transects at each point count location or wildlife camera. Top models for height and cover of forbs and cover for bare ground included an interaction between site and year (Table 1). Cover for bare ground was significantly higher in the bison site compared to the reference site and declined over time in both sites. The cover and height of forbs, warm and cool season grasses, and shrubs did not differ as a function of bison reintroduction (Table S19; Figure S20).

#### 3.4 Visitor place attachment

We intercepted 243 people before, and 525 people after bison reintroduction. Our response rate was 75% (n = 184) in 2015 and a 56% (n = 302) in 2016. We surveyed approximately the same ratio of women to men in each year (74%; 76%), and most visitors were in the age range of 36–55 (49%; 45%), and were local to the Fort Collins, Colorado area (81%; 85%). The majority of visitors identified as Caucasian (94%) and had either bachelors or graduate degrees (40%; Table S20). We did not detect a nonresponse bias based on the survey question that all visitors answered (2015: $\chi^2 = 1.52, df = 1, p = 0.21$ and 2016: $\chi^2 = 3.10, df = 1, p = 0.08$), nor did we find a difference in group size between participants and nonrespondents (2015: $\chi^2 = 4.02, df = 7, p = 0.78$ and 2016: $\chi^2 = 9.75, df = 5, p = 0.08$).

Of the people who agreed to take the survey, all completed the place attachment survey. Visitors had significantly higher place attachment scores after bison reintroduction ($t = 3.19, p = 7.7e−04$; Figure 3; Table S22). In addition, a higher percent of people agreed that they felt at home in Soapstone and wanted to spend more time in grasslands like Soapstone after the bison reintroduction (Figure S23). In response to the open-ended follow-up question, 95% of visitors said Soapstone was important to them and explained...
why. The top 10 themes were similar between years, but “Historical Significance” and the “Importance of Protecting Open Space” were more important after bison reintroduction (Figure 4; Table S24).

4 | DISCUSSION

Considering both the ecological and social dimensions of reintroduction efforts is critical to long-term species recovery (Seddon et al., 2007). Most previous research on the ecological effects of bison has focused on their keystone role in ecosystem processes (Frank & Evans, 1997; Knapp et al., 1999; Towne et al., 2005). In contrast, no previous study has quantified the ecological effects of bison reintroduction in tandem with social outcomes such as visitor place attachment. While we found few changes to bird, mammal, and plant...
communities within 2 years of bison reintroduction, we anticipate that this research could serve as baseline data that could help document longitudinal changes at this site and others. Furthermore, we found immediate and significant differences in visitors' place attachment to the site following bison reintroduction.

We found no effect of bison reintroduction on songbirds with sufficient detections for analysis. Other studies have demonstrated increases in abundance of the same or similar species in tall and mixed grass prairie, but occurred at sites that were grazed by bison over longer time periods (4–10+ years), at higher densities (1.2 animal units/ha), and were burned as well as grazed (Griebel et al., 1998; Powell, 2006). We did observe decreases in plant cover and height following bison reintroduction, which could indicate that bison have begun to create conditions that could provide high quality habitat for obligate and facultative grassland birds in the future (Towne et al., 2005).

It is possible that we detected little to no bird response to bison reintroduction because climate can be more important than grazing as a driver of bird abundance in arid grasslands, particularly where grazing intensity is low-moderate (Niemuth, Solberg, & Shaffer, 2008). For example, the decrease in density by Western Meadowlark across both bison and reference sites can probably be attributed to fluctuations in precipitation during our study period (Niemuth et al., 2008). Rainfall in 2015, the year prior to bison reintroduction, was 196 mm during May–June, substantially higher than in subsequent years (2015: 43 mm; 2016: 95 mm; “Historical Weather” 2018). Other studies have demonstrated decreases in nest survival for shortgrass prairie birds during periods of drought or increased temperatures (Conrey, Skagen, Yackel Adams, & Panjabi, 2016). For example, Lark Bunting, which are more productive in areas with higher precipitation (Skagen & Yackel Adams, 2012), were only abundant at our site in 2015, the high rainfall year.

We did not observe significant changes in mammal habitat use in response to bison reintroduction, but we did observe several trends (Figure 2). Coyote habitat use increased in both the bison and reference sites after the reintroduction, while lagomorph and mule deer habitat use decreased only in the bison site. Since coyote habitat use increased at both bison and reference sites following bison reintroduction, we cannot attribute this increase to bison. We observed decreases in lagomorph habitat use after bison reintroduction, which contrasts with previous small mammal studies that have found increased abundance following bison introduction (1.2 animal units/ha), particularly when pairing grazing with burning (Matlack et al., 2001).

We observed no change in grass cover, but forb cover, bare ground, and average plant height decreased following bison reintroduction. In contrast to our findings, Towne et al. (2005) examined plant communities on bison, cattle, and ungrazed sites over 10 years in tallgrass prairie with stocking densities of 1.7 animal units/ha, and found that warm season grass cover decreased and cool season grass cover increased on bison-grazed pastures. Others have documented higher forb biomass (Fahnstock & Knapp, 1994) and cover (Towne et al., 2005) in annually or seasonally burned bison-grazed pastures. We documented a decline in grass, forb, and shrub height after the bison reintroduction. This is consistent with past studies showing that bison in other shortgrass prairie systems graze on grasses, forbs, and subshrubs, such as Artemisia frigida and Gutierrezia sarothrae (Peden, Van Dyne, Rice, & Hansen, 1974), which are the two most common subshrubs in our study site. Yet, while our models provided some indication that plant cover and height were altered by bison grazing, we urge caution in interpreting these results as the confidence intervals for cover and height estimates before and after bison reintroduction were often overlapping (Figure S20).

Although we detected few ecological effects in the 2 years following bison reintroduction, we documented a significant increase in visitor place attachment to our study area. This result supports the idea that people connect with landscapes in which they can view charismatic wildlife (Tremblay, 2008). We found that visitors cited the importance of protecting open spaces more frequently after the bison reintroduction, suggesting that the presence of bison, as a highly recognizable flagship species, forged a stronger sense of connection between visitors and landscape conservation. Historical significance was also a more important theme following bison reintroduction. This theme potentially emerged due to increased public knowledge about and attention to the fact that bison once roamed the Great Plains as a result of news coverage about the bison reintroduction. Understanding people's place attachment to open space, and how it may change over time, can aid protected area managers in developing programs or making management decisions informed by stakeholder perceptions and values. In doing so, managers can potentially reduce future conflicts, while enhancing the visitor experience and achieving conservation objectives (Williams & Vaske, 2003).

While we documented a significant increase in visitor place attachment and some ecological responses to bison reintroduction, our study was limited spatially and temporally. We may observe stronger ecological responses to the bison as the herd grows, grazing intensity increases, and bison spend more time on the landscape. Conversely, place attachment scores in the coming years following the bison reintroduction may decrease or stabilize as the initial public excitement surrounding the reintroduction ebbs.

To fully understand how bison influence natural and human communities, future research should employ a before-after-control-impact (BACI) design, with consistent long-term monitoring of social and ecological variables, across a network of reintroduction sites with diverse stocking densities (Griebel et al., 1998; Towne et al., 2005). In
addition to grazing, fire is an important tool for creating heterogeneous habitat that is critical for sustaining prairie biota (Fuhlendorf, Engle, Kerby, & Hamilton, 2009). Most studies in tall and mixed grass prairie have observed changes to flora and fauna in response to bison grazing coupled with annual burns (Fahnestock & Knapp, 1994; Griebel et al., 1998; Matlack et al., 2001; Powell, 2006; Towne et al., 2005). Yet, the potential interactive effects of bison and fire in shortgrass prairie are virtually unknown. In addition, while studies have assessed the effects of bison grazing (Jonas & Joern, 2007) and wallowing (Nickell, Varriano, Plemmons, & Moran, 2018) on grassland arthropod communities, future research should consider the trophic interactions among bison, birds, arthropods, and other grassland taxa.

Finally, bison reintroduction efforts should also consider the social effects of bringing bison back to grasslands (Seddon et al., 2007). Our study was limited to evaluating visitor response, as measured by place attachment. Future studies should also consider a wider array of stakeholder perspectives including those of adjacent landowners, cattle and bison ranchers, tribal groups, and others (Reading, Clark, & Kellert, 2002). Although we found few short-term ecological effects of bison reintroduction, our findings provide new insight into the largely untapped social potential for bison reintroduction to catalyze public interest in grasslands, one of the world’s most threatened biomes.

ACKNOWLEDGMENTS
The authors are grateful to the City of Fort Collins Natural Areas Department and the Larimer County Department of Natural Resources for providing help and permission to access their lands for this study. We especially thank D. Figgs, M. Flenniken, C. Gindler, and J. Frederickson. This research was supported by One Health Research and Development Funds (CSU), AAUW, Audubon Society of Greater Denver, Larimer County, Prairie Biotic Research, Inc., Denver Zoo, the Colorado Chapter of the Wildlife Society, Sigma Xi, and Colorado Field Ornithologists. We thank the excellent field technicians and volunteers who spent long days helping this research succeed (M. Wing, M. Warner, A. Thedodorakos, B. Nooner, M. Spencer, T. Funabashi, M. Kurtz, A. Quintana, A. Interpreter, M. Van Eden, K. Rayens, T. Franks, B. Reidinger, C. Marshall, C. Kusaka, C. Castagnet, M. Crump, K. Barnes, T. Greene, M. Jimenez, and T. Gallo). Data is available on GitHub. For the social science data collection, visitors could decline or end the study, and we stored data in a password protected iSurvey account, which was accessible only to the research team.

CONFLICTS OF INTEREST
The authors have no conflicts of interest to report.

Author contributions
K.W. and L.P. conceived the ideas and designed the ecological methodology; R.G. designed the social science methodology; K.W. collected and analyzed the ecological data; K.W. and R.G. collected and analyzed the social science data. K.W. and L.P. led the writing of the manuscript. All authors contributed to the drafts and approved publication.

ORCID
Kate Wilkins https://orcid.org/0000-0002-9875-1149

REFERENCES
Bailey, L. L., Hines, J. E., Nichols, J. D., & MacKenzie, D. I. (2007). Sampling design trade-offs in occupancy studies with imperfect detection: Examples and software. Ecological Applications, 17, 281–290.

Bernard HR. 2011. Research methods in anthropology: Qualitative and quantitative approaches. AltaMira Press, Lanham, MD.

Buckland, S. T., Rexstad, E. A., Marques, T. A., & Oedekoven, C. S. (2015). Distance sampling: Methods and applications. New York, NY: Springer.

Bumham, K. P., & Anderson, D. R. (2002). Model selection and multimodel inference: a practical information-theoretic approach (2nd ed.). New York, NY: Springer.

Conrey, R. Y., Skagen, S. K., Yackel-Adams, A. A., & Panjabi, A. O. (2016). Extremes of heat, drought and precipitation depress reproductive performance in shortgrass prairie passerines. Ibis, 158, 614–629.

Daubenmire, R. (1959). A canopy-coverage method of vegetational analysis. Northwest Science, 33, 43–64.

DeLuca, T. H., & Zabinski, C. A. (2011). Prairie ecosystems and the carbon problem. Frontiers in Ecology and the Environment, 9, 407–413.

Dirzo, R., Young, H. S., Galetti, M., Ceballos, G., Isaac, N. J., & Collen, B. (2014). Defaunation in the Anthropocene. Science, 345, 401–406.

Donlan, J., Greene, H. W., Berger, J., Bock, C. E., Bock, J. H., Burney, D. A., … Soule, M. E. (2005). Re-wilding North America. Nature, 436, 913–914.

Fahnestock, J. T., & Knapp, A. K. (1994). Plant responses to selective grazing by bison: Interactions between light, herbivory and water stress. Vegetatio, 115, 123–131.

Farr, C. M., Pejchar, L., & Reed, S. E. (2017). Subdivision design and stewardship affect bird and mammal use of conservation developments. Ecological Applications, 27, 1236–1252.

Fiske, I., & Chandler, R. (2011). Unmarked: An R package for fitting hierarchical models of wildlife occurrence and abundance. Journal of Statistical Software, 43, 1–23.

Fletcher, R. J., & Koford, R. R. (2002). Habitat and landscape associations of breeding birds in native and restored grasslands. The Journal of Wildlife Management, 66, 1011.

Flores, D. L. (1999). Horizontal yellow: Nature and history in the near Southwest (1st ed.). Albuquerque, NM: University of New Mexico Press.

Folmer, A., Haartsen, T., & Huigen, P. P. (2013). The role of wildlife in emotional attachment to a nature-based tourism destination. Journal of Ecotourism, 12, 131–145.

Frank, D. A., & Evans, R. D. (1997). Effects of native grazers on soil N cycling in Yellowstone National Park. Ecology, 78, 2238–2248.

Freece, C. H., Aune, K. E., Boyd, D. P., Derr, J. N., Forrest, S. C., Cormack Gates, C., … Redford, K. H. (2007). Second chance for the plains bison. Biological Conservation, 136, 175–184.

Fritz, H., Duncan, P., Gordon, I. J., & Blus, A. W. (2002). Megaherbivores influence trophic guilds structure in African ungulate communities. Oecologia, 131, 620–625.

Fuhlendorf, S. D., Engle, D. M., Kerby, J., & Hamilton, R. (2009). Pyric herbivory: Rewilding landscapes through the recoupling of fire and grazing. Conservation Biology, 23, 588–598.

Fuhlendorf, S. D., Harrell, W. C., Engle, D. M., Hamilton, R. G., Davis, C. A., & Leslie, D. M., Jr. (2006). Should heterogeneity be the basis for conservation? Grassland bird response to fire and grazing. Ecological Applications, 16, 1706–1716.
