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Research Article

Habitat Selection by Mountain Plovers in Shortgrass Steppe

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ABSTRACT Much of the breeding range for the mountain plover (*Charadrius montanus*) occurs in shortgrass steppe and mixed-grass prairie in the western Great Plains of North America. Studies of mountain plovers in shortgrass steppe during the 1970s and 1990s were focused in Weld County, Colorado, which was considered a key breeding area for the species. These studies, however, did not include habitats influenced by black-tailed prairie dogs (*Cynomys ludovicianus*) or prescribed fire. The role of these 2 rangeland disturbance processes has increased substantially over the past 15 years. During 2008–2009, I used radial distance point count surveys to estimate mountain plover densities early in the nesting season in 4 habitats on public lands in Weld County, Colorado. All 4 habitats were grazed by cattle during the growing season at moderate stocking rates but had different additional disturbances consisting of 1) dormant-season prescribed burns, 2) active black-tailed prairie dog colonies, 3) black-tailed prairie dog colonies affected by epizootic plague in the past 1–2 years, and 4) rangeland with no recent history of fire or prairie dogs. Mountain plover densities were similar on active black-tailed prairie dog colonies ($\bar{X} = 6.8$ birds/km², 95% CI = 4.3–10.6) and prescribed burns ($\bar{X} = 5.6$ birds/km², 95% CI = 3.5–9.1). In contrast, no plovers were detected at randomly selected rangeland sites grazed by cattle but lacking recent disturbance by prairie dogs or fire, even though survey effort was highest for this rangeland habitat. Mountain plover densities were intermediate (2.0 birds/km², 95% CI = 0.8–5.0) on sites where black-tailed prairie dogs had recently been extirpated by plague. These findings suggest that prescribed burns and active black-tailed prairie dog colonies may enhance breeding habitat for mountain plovers in shortgrass steppe and illustrate the potential for suppressed or altered disturbance processes to influence habitat availability for declining wildlife species. © 2011 The Wildlife Society[†]

KEY WORDS disturbance regimes, grassland, grazing, plague, prescribed fire, rangeland management, resource selection.

The migratory mountain plover (*Charadrius montanus*) breeds in the western Great Plains of North America and in shrub steppe and montane rangelands immediately west of the Great Plains (Knopf and Wunder 2006). Mountain plover numbers declined substantially between 1960s and 1990s (Knopf and Wunder 2006) and the bird was proposed for listing as threatened under the Endangered Species Act in 1999 due to concerns including habitat loss and the influence of domestic livestock management practices in combination with loss of native herbivores (U.S. Fish and Wildlife Service [USFWS] 1999). This proposal was withdrawn in 2003 but subsequently reinstated in 2010 (USFWS 2010), and the mountain plover is currently listed as a Species of Concern by the United States Forest Service, a Bird of Conservation Concern by the USFWS, and a species of Special Concern in Colorado (Colorado Division of Wildlife 2003).

A common feature of mountain plover nesting habitat in rangelands of the western Great Plains is high levels of bare soil (Olson and Edge 1985, Knopf and Miller 1994, Plumb

et al. 2005). In mixed-grass prairie of the northwestern Great Plains, mountain plovers breed primarily on black-tailed prairie dog (*Cynomys ludovicianus*) colonies or hardpan clay soils with sparse vegetation (Dinsmore et al. 2005, Childers and Dinsmore 2008). The more arid sagebrush steppe of Wyoming and high-elevation grasslands in Colorado can support breeding mountain plovers without prairie dog disturbance (Wunder et al. 2003, Plumb et al. 2005). In shortgrass steppe of the southwestern Great Plains, where vegetation cover is intermediate between mixed-grass prairie and sagebrush steppe, mountain plover breeding habitat includes rangeland grazed by livestock, black-tailed prairie dog colonies, and tilled cropland (Knopf and Miller 1994, Shackford et al. 1999, Tipton et al. 2008).

Inferences from studies of habitat use and selection can be constrained by the kinds of habitats available within a given study area (Johnson 1980). For species of conservation concern, it is important to consider whether key habitats may be missing from the landscape. In the case of the mountain plover, studies during 1970–1990s found that the birds commonly breed in shortgrass steppe grazed only by livestock (Graul 1975, Miller and Knopf 1993, Knopf and Miller 1994). However, those authors also noted that some mountain plover populations in shortgrass rangeland appeared to be declining and suggested that the areas with short, sparse

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vegetation may be important as breeding habitat (Graul and Webster 1976, Knopf and Miller 1994, Knopf and Wunder 2006, Knopf 2008). Although fire and grazing by black-tailed prairie dogs were widespread disturbances in shortgrass steppe prior to European settlement (Wright and Bailey 1982, Knowles et al. 2002), both have been suppressed in the southern Great Plains over the past century. Recently, a single-year survey documented higher densities of mountain plovers on black-tailed prairie dog colonies than on shortgrass steppe grazed only by livestock (Tipton et al. 2009). Less is known about the influence of fire on mountain plover habitat, but Svingen and Giesen (1999) observed higher densities of mountain plovers on prescribed burns than on unburned shortgrass steppe during migration. These studies suggest that disturbances that were rare within the landscape studied by Graul (1975) and Knopf and Miller (1994) may be important to breeding mountain plovers.

Graul and Webster (1976) identified the Pawnee National Grassland (PNG) in northeastern Colorado as a breeding stronghold for mountain plovers and noted the need for improved estimates of mountain plover densities in eastern Colorado due to concerns that populations were declining. Graul (1975) described breeding habitat as predominantly blue grama (*Bouteloua gracilis*) and buffalograss (*Buchloe dactyloides*) with scattered patches of cacti (*Opuntia polyacantha*) and western wheat grass (*Pascopyrum smithii*). Plover densities ranged from 4 birds/km² to 32 birds/km², with an average of about 20 birds/km² during 1969–1974 (Graul and Webster 1976). On the adjacent Central Plains Experimental Range (CPER), Giezentanner and Ryder (1969) reported 8.2 plovers/km² in 1969. During 1990–1994, annual surveys using distance sampling on PNG estimated annual plover densities varying from 2.0 birds/km² to 4.7 birds/km², but few plovers were observed after 1994 (Knopf 2008).

The National Grasslands, including PNG, were established in response to the extensive wind erosion and soil movement that occurred during the Dust Bowl of the 1930s, and management of these public lands during the subsequent half century focused on vegetation establishment, soil stabilization, and successional development of native shortgrass plant communities (Hart 2008, Peters et al. 2008). Since the early 1990s, however, the PNG has undergone 2 additional changes in the disturbance regime (Fig. 1). First, the area of black-tailed prairie dog colonies expanded nearly 7-fold, from an annual average of 218 ha during 1990–1994 (Stapp et al. 2004) to an annual average of 1,452 ha during 2003–2007 (Fig. 1). Although prairie dog populations have increased, plague epizootics influence prairie dog distribution and abundance on PNG, resulting in a mosaic of colonies in varying stages of expansion or contraction (Stapp et al. 2004, Savage 2007, Hartley et al. 2009).

Another change in the disturbance regime on the PNG was initiation of a prescribed burning program in 1995, in part to evaluate its role in improving habitat for mountain plovers (Fig. 1; Knopf 2008). Fire was historically a common disturbance of the shortgrass steppe, particularly following years with above-average precipitation (Wright and Bailey

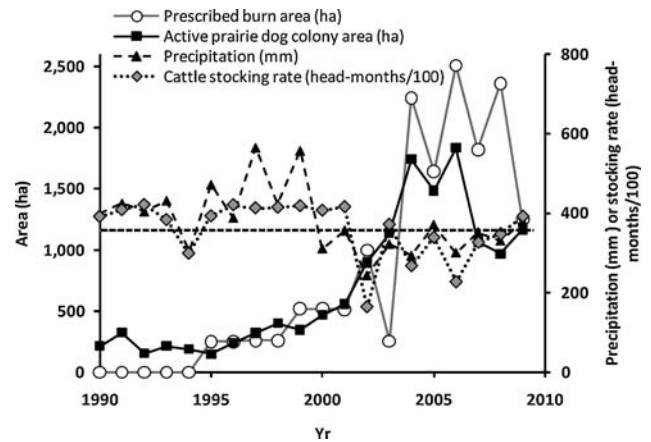


Figure 1. Annual variation in area (ha) of active black-tailed prairie dog colonies, area (ha) of prescribed burns, precipitation (mm), and cattle stocking rates (total authorized head-months divided by 100) on the Pawnee National Grassland (PNG) in northeastern Colorado during 1990–2009 (Stapp et al. 2004; C. Armstrong, N. Clark, E. Humphrey, and P. White, United States Dairy Association—Forest Service, unpublished data). Annual precipitation is the average of 3 long-term gauges maintained within the study area. Dashed line shows the long-term (1981–2008) mean annual precipitation based on the same 3 gauges.

1982), but fire return intervals have been difficult to estimate. Wildfires have largely been suppressed for the past century in the shortgrass steppe. Prescribed burns have occurred on PNG annually since 1995, varying from 252 ha in 2003 (following a drought in 2002) to 2,400 ha in 2006 (Fig. 1). During 1999–2006, plover nests were observed on most of these prescribed burns (Knopf 2008). The increased use of fire and changes in prairie dog populations reflect a growing interest in the restoration of historic disturbance regimes as one means to sustain biodiversity in Great Plains rangelands (Fuhlendorf and Engle 2001, Samson et al. 2004, Fuhlendorf et al. 2006, Dermer et al. 2009).

One factor that has remained consistent within the study area over the past 4 decades is livestock grazing management. Most of the PNG is grazed by cattle during May–October each year using moderate stocking rates that have been shown to sustain long-term plant and livestock production (Klippel and Costello 1960, Bement 1969). Over the past 20 years, notable reductions in stocking rates have only occurred in response to droughts in 1994, 2002, 2004, and 2006 (Fig. 1). Compared with other rangelands worldwide, the shortgrass steppe is highly resistant to effects of livestock grazing in terms of plant cover, density, and productivity (Milchunas et al. 2008). Increased stocking rates (to 50% above the moderate rate) can increase dominance of warm-season shortgrasses, blue grama (*Bouteloua gracilis*), and buffalograss (*Buchloe dactyloides*), and reduce dominance of cool-season midgrasses such as western wheat grass (*Pascopyrum smithii*) without significant losses of vegetation cover (Milchunas et al. 2008). An ongoing experiment at CPER is examining the potential for extremely high stocking rates (100% above moderate) to increase bare soil exposure in shortgrass steppe, but such practices are not currently used on PNG so I could not evaluate them as part of my study of mountain plover habitat selection.

Given the history of mountain plover research and shifts in disturbance regimes on PNG, my objectives were to compare densities of mountain plovers in 4 habitats: 1) shortgrass steppe grazed by cattle at moderate stocking rates, but lacking other disturbances, 2) prescribed burns conducted during the dormant season (late fall or late winter), 3) active black-tailed prairie dog colonies, and 4) black-tailed prairie dog colonies recently affected by plague.

STUDY AREA

The study area encompassed approximately 60,900 ha of public lands in Weld County, Colorado consisting of all of CPER, all allotments on the western unit of PNG, and 6 allotments on the eastern unit of PNG. The climate was semiarid, with cold, dry winters; most precipitation fell as rain between April and September (Lauenroth and Milchunas 1992). Long-term mean annual precipitation measured at 3 gauges within the study area during 1981–2009 was 355 mm. Annual precipitation was 331 mm in 2007, 317 mm in 2008, and 379 mm in 2009. Topography is flat to gently undulating, with slopes typically 0–3%. Vegetation was dominated by the perennial, grazing-tolerant shortgrasses, blue grama, and buffalograss (Milchunas et al. 1989, Lauenroth and Milchunas 1992).

METHODS

Survey Design

The field team sampled 4 habitats: 1) sites where black-tailed prairie dogs were currently present (active colonies), 2) sites where black-tailed prairie dogs had been extirpated by plague 1–2 years prior to the survey (plagued colonies), 3) sites treated with prescribed fire either in the fall (Nov) or late winter (Mar) preceding the survey, and 4) rangeland with no recent history of fire or prairie dog presence. All 4 habitats were grazed at moderate stocking rates (0.63–0.75 animal unit months/ha) during summer (approx. mid-May to mid-Oct) each year, which was the standard grazing management practice on PNG and CPER. Because the rangeland habitat (grazed by cattle, but no recent history of prairie dogs or burns) represented 95–99% of the study area annually, it was an important baseline for comparison to the prairie dog and burned habitats (see also Childers and Dinsmore 2008). To sample the rangeland habitat, I used the United States Public Land Survey System to identify all 260 ha parcels (sections) within the study area where >50% of the section consisted of PNG or CPER. I then randomly selected a subsample of 12 sections with no known history of prairie dog presence or burning within the past 3 years. We surveyed each section with a 10 point grid, with points spaced at 250-m intervals along 2 parallel transects and 500 m between the 2 transects.

Past research and surveys in the study area identified several rangeland sites with a long history of breeding mountain plover presence (Giezentanner and Ryder 1969, Sordahl 1991, Knopf 2008). The field team also conducted surveys

at 3 non-randomly selected pastures identified by these past studies. Two sites were located on PNG and harbored nesting mountain plovers in the early 2000s (Knopf 2008). The third site, located on CPER, was a consistent breeding area for mountain plovers during studies in the 1960s and 1990s (Giezentanner and Ryder 1969, Sordahl 1991). These 3 sites had no recent history of fire or prairie dogs. The CPER site was grazed by cattle during summer at a stocking rate 50% higher than most pastures in the study area since 1939 (Milchunas et al. 2008). We sampled a 10 point grid at each of these 3 non-random rangeland sites in addition to the 12 randomly selected rangeland sites.

Prescribed burns (>130 ha each) were conducted on PNG in March 2008 (7 burns) and 2009 (4 burns). Three prescribed burns (16 ha each) were conducted on CPER each November preceding the 2008 and 2009 growing seasons. We sampled all prescribed burns using 10 points per burn for large (≥ 130 ha) burns and 2 points per burn for small (16 ha) burns. The 10 point grids followed the same design as for the rangeland sites. In 2008, we surveyed 10 burns (76 points) and in 2009 we surveyed 7 burns (46 points).

For black-tailed prairie dog colonies, my sampling design addressed spatiotemporal variability in the size and location of active colonies caused by epizootic plague outbreaks (Stapp et al. 2004, Augustine et al. 2008*b*). In 2008, we sampled all known active black-tailed prairie dog colonies within the study area that were ≥ 16 ha in size based on previous surveys by PNG staff. We also surveyed 3 colonies where prairie dogs had been extirpated by plague in 2007. We overlaid both the active and the inactive colonies with a grid of survey points with 250 m spacing between points, such that the number of points varied with colony size and shape, up to a maximum of 10 points. The plague-affected colonies we sampled in 2008 contained only a few small areas with surviving prairie dogs, such that no survey point had >50% of the area within a 200 m radius of the point affected by current prairie dog activity.

In 2009, we again surveyed all colony grids surveyed in 2008. In addition, we established new survey grids on 4 previously plague-affected colonies that expanded substantially between 2008 and 2009, plus 2 active colonies that were present but not known in 2008. Prairie dogs re-colonized portions of 2 of the plague-affected colonies sampled in 2008, and plague extirpated prairie dogs from portions of 4 of the active colonies sampled in 2008. Finally, 2 small new colonies had expanded into 2 of the random rangeland sampling grids. Thus, in 2009 we sampled 19 colonies: 12 active colonies, 4 colonies partially or completely affected by plague during the previous year, and 3 colonies affected by plague 2 years ago that now contained small areas of prairie dog activity. In early April 2009, we delineated and mapped boundaries of the active portion of all colonies using a Global Positioning System (GPS). Our sampling grids overlaid the active portion of the colony plus those areas where plague had extirpated prairie dogs within the past 2 years. We defined survey points as active when >50% of the area surrounding the point was affected by current prairie dog activity.

Survey Methods

The field team used radial distance point count surveys (Buckland et al. 2001) to sample mountain plovers in each of the 4 habitats during 21 April–7 May 2008 and 2009. Most mountain plovers arrive during March or early April and establish territories during April. Nests are initiated as early as 20 April, and peak-nesting activity is in mid- to late-May (Graul 1975). At the time of our surveys, mountain plovers were active (feeding and courtship behaviors) in the morning and had not yet initiated the cryptic behaviors that accompany peak nesting season in mid- to late-May. We conducted surveys between sunrise and 10:30 hr, during standardized weather conditions (no precipitation, 0–24 km/hr wind speed), with the same 2 observers in both years. We used a GPS with preprogrammed point locations to navigate to all points. The observer drove to each point using an all-terrain or 4-wheel drive vehicle and then exited the vehicle for the count. Vehicles are recommended for use in mountain plover surveys because plover flushing distances may be within 3 m for vehicles but 50–100 m for humans approaching on foot (USFWS 2002). At each survey point, the observer searched for 5 min using 8×42 binoculars and used a laser rangefinder to measure the exact distance from the point to any plover(s) detected. We treated plovers detected together (i.e., the second plover was detected as a result of the first detection) as one observation and recorded group size, whereas we recorded plovers detected independent of one another as separate observations. If we observed plover movements suggesting that a bird could have moved between points and may have been detected twice, we noted it and excluded the second detection from the dataset.

To examine whether plovers were indeed initiating nests at sites where we detected them during point counts, during the first 2 weeks of May we returned to as many of the sites as possible where we detected plovers on ≥ 1 sampling point. On the return visit, we used an all-terrain vehicle to systematically drive across the area covered by the point-count grid, stopping approximately every 100 m, stepping away from the vehicle, and scanning with binoculars for mountain plovers. Once we found a bird, we observed the individual until we found its nest or until we observed behaviors suggesting that the bird did not have a nest (e.g., >1 hr of observation with no bobbing of the head or digging of nest bowls observed).

Statistical Analyses

I used Program DISTANCE (Thomas et al. 2006) to model the probability of detecting a plover as a function of the distance from a given survey point (i.e., the detection function). I then used DISTANCE to derive habitat-specific density estimates and associated confidence intervals based on the number of plover detections, number of survey sites, and number of survey points per site in each habitat in combination with the derived detection function. I evaluated 6 robust models for the detection function as suggested by Buckland et al. (2001): 1) uniform key with cosine expansion, 2) uniform key with simple polynomial expansion, 3) half-normal key with cosine expansion, 4) half-normal key with hermite polynomial expansion, 5) hazard-rate key with

cosine expansion, and 6) hazard-rate key with simply polynomial expansion. I also fitted models that included year, observer, and habitat as covariates for the detection function (Marques and Buckland 2004). I used Akaike's Information Criteria (AIC) to identify the best-approximating model and evaluate whether the covariates improved model fit. The test for the habitat covariate compared active prairie dog colonies, prescribed burns, and other habitats combined. Prior to fitting models, I inspected a histogram and global detection function for all detections combined, and truncated the dataset at a detection distance of 175 m, which corresponded the distance at which the detection probability was approximately 0.1, as recommended by Buckland et al. (2001). I then used a combination of the detection function selected based on AIC and the habitat-specific number of plover detections, survey sites, and survey points to calculate plover density in each habitat.

RESULTS

We detected 82 mountain plovers, of which 34 (41%) were on prescribed burns and 37 (45%) were on active prairie dog colonies. We did not detect any plovers on the randomly selected rangeland grids in either year. A detection function based on the half-normal key with cosine expansion that included observer identity as a covariate produced the best-approximating model (AIC = 177.08). The model also fit the data well based on a chi-squared goodness-of-fit test ($P > 0.15$). Including year in the model did not improve model fit (AIC = 179.35; Δ AIC = 2.27), so I pooled detection distances across years for the detection function. Inclusion of a habitat covariate consisting of 3 categories (burns, active prairie dog colonies, all other habitats) also did not improve model fit (AIC = 180.29; Δ AIC = 3.20). The selected detection function gave an effective sampling radius surrounding survey points of 110.1 m (CV = 5.5%, 95% CI = 98.6–122.9 m) and a mean overall probability of detection of 0.40 (95% CI = 0.32–0.49).

Plover densities were similar on prescribed burns and active prairie dog colonies in both 2008 and 2009 (Table 1). Across years, plover densities averaged 5.6 birds/km² (95% CI: 3.5–9.1) on burns and 6.8 birds/km² (95% CI: 4.3–10.6) on active prairie dog colonies (Table 1). Mean plover density on inactive prairie dog colonies (2.0 birds/km²) was significantly lower than the estimate for active colonies at the 90% but not at the 95% confidence level (Table 1). We did not detect plovers on the 12 randomly selected rangeland grids in either year (Table 1). However, we detected plovers at 2 of the 3 non-random rangeland sites each year, indicating that plovers do occur in rangeland habitat, albeit at densities so low that the birds could not be detected using a survey that allocated more effort to the randomly selected rangeland grids than to any other habitat. Features distinguishing the 3 non-random rangeland sites from the rest of the landscape are not known, hence it is not possible to determine what proportion of the study area these non-random rangeland sites represent.

To determine whether plover detected during point counts were in fact breeding at the sites we surveyed, we revisited 13

Table 1. Estimates of mountain plover density in different habitats on the Pawnee National Grassland and Central Plains Experimental Range in northeastern Colorado during 2008–2009.

	2008 Birds/km ²					2009 Birds/km ²					2 years mean Birds/km ²					
			Lower	Upper	Sites			Lower	Upper	Sites			Lower	Upper	Lower	Upper
	Sites	Points	Mean	95% CI		Points	Mean	95% CI	Points		Mean	95% CI	Points	Mean	95% CI	90% CI
Active prairie dog colonies	9	53	5.3	2.7	10.3	18	83	7.8	4.6	13.4	6.8	4.3	10.6	4.7	9.9	
Prescribed burns	10	76	4.5	2.3	8.9	7	46	7.4	4.1	13.5	5.6	3.5	9.1	3.7	8.4	
Inactive prairie dog colonies	3	29	1.6	0.4	5.8	7	45	2.3	0.7	7.5	2.0	0.8	5.0	0.9	4.3	
Unburned rangeland; no prairie dogs	12	120	0			12	118	0			0					

sites where we detected plovers during point counts in 2008 and 15 sites where we detected plovers during point counts in 2009. Overall, we documented evidence of breeding activity at 26 out of 28 (93%) of the sites. There was no difference among habitats in the proportion of sites where we detected evidence of breeding behavior (Table 2; Fisher’s exact test for 2 × 3 table, *P* = 0.27).

DISCUSSION

One challenge to identifying habitats important to declining wildlife species is that such habitats may already be lost or rare in areas where the species persists (McCleery et al. 2007). In the case of the mountain plover, many past studies in shortgrass steppe were conducted in moderately grazed rangeland where black-tailed prairie dogs and fire had largely been eliminated from the landscape (Graul 1975, Knopf and Miller 1994). Within my study area, research over the past 3 decades suggests that densities of breeding mountain plovers in rangeland habitat lacking prairie dogs and fire have declined substantially, from 4–32 birds/km² during 1969–1974 (Giezantanner and Ryder 1969, Graul and Webster 1976) to 2.0–4.7 birds/km² during 1990–1994 (Knopf 2008), to the current situation in 2008–2009 where random sampling at a similar intensity did not detect any plovers.

During 1995–2009, shortgrass steppe on PNG and CPER continued to undergo successional recovery from the Dust

Bowl years of the 1930s, whereby long-lived, perennial shortgrasses have slowly regained dominance (Coffin et al. 1996, Peters et al. 2008). During the same time period, fire and black-tailed prairie dogs were restored to a portion of the landscape (Fig. 1). Black-tailed prairie dogs and prescribed fires both reduce the height and cover of standing vegetation and litter (Augustine and Milchunas 2009; Hartley et al. 2009; D. Augustine, USDA—Agriculture Research Service, unpublished data). Within the current landscape, mountain plovers exhibit strong selection for prescribed burns and active black-tailed prairie dog colonies (Table 1), which together comprised 3.4% of the study area during 2008–2009. One potential explanation consistent with both the long-term decline in plover densities in rangeland habitat and current plover selection for burns and prairie dog colonies is that late-successional, moderately grazed shortgrass steppe represents suboptimal breeding habitat. When fire and prairie dogs were absent from the landscape, Knopf and Miller (1994) found that the mountain plovers still selected microsites for nesting that had more bare soil (32%) compared to non-nest sites (14%). Knopf and Miller (1994:505) noted that “additional observations suggest 30% bare ground is likely closer to a minimum habitat requirement than an optimal one in Mountain Plover ecology. First, Mountain Plovers nest in the more xeric landscapes west of the shortgrass prairie province. Second most nesting attempts by plovers on the PNG are initiated from late April to May (Graul 1975), a period when the shortgrass species remain dormant. Third, plovers often raise broods in the vicinity of excessive, local disturbance as at cattle watering or loafing areas.” Subsequent studies found that mountain plovers also breed in cultivated fields with bare ground, sparse vegetation, or stubble cover (Shackford et al. 1999, McConnell et al. 2009, Tipton et al. 2009). Non-mutually exclusive explanations for mountain plover selection of disturbed habitats for nesting include enhanced accessibility of insects, enhanced ability of plovers to detect predators, and the ability of mountain plovers to avoid detection in areas with high levels of bare soil (Knopf and Wunder 2006, Schneider et al. 2006).

Studies in mixed-grass prairie of Montana have clearly demonstrated the importance of prairie dog colonies as mountain plover habitat (Olson and Edge 1985, Dinsmore et al. 2005, Childers and Dinsmore 2008), but fewer studies have examined the role of prairie dog colonies in the southern Great Plains. A survey of primarily private lands in Colorado in 2005 found higher mountain plover

Table 2. Observations of mountain plover breeding activity during 2008–2009 on the Pawnee National Grassland and Central Plains Experimental Range in northeastern Colorado, at sites where we detected mountain plovers during point count surveys and then revisited during the first 2 weeks of May to search for nests.

Habitat	Number of sites revisited	Breeding plovers present ^a	No. breeding plovers found
2008			
Prescribed burns	6	6	0
Prairie dog colonies	5	4	1
Rangeland	2	2	0
2009			
Prescribed burns	5	5	0
Prairie dog colonies	8	8	0
Rangeland	2	1	1
Total	28	26	2

^a At one burned site in 2009, we observed plovers exhibiting courtship behaviors but did not find a nest. At the other 25 sites, we found ≥1 mountain plover nest.

densities on prairie dog colonies (2.3 plovers/km²) compared to rangeland without prairie dogs (0.23 plovers/km²; Tipton et al. 2009), providing initial evidence for the importance of prairie dog colonies for mountain plovers in shortgrass steppe. My surveys support this conclusion, documenting a mean of 6.8 plovers/km² on prairie dog colonies in shortgrass steppe in Weld County, Colorado, which is similar to densities of 7.2 plovers/km² found on prairie dog colonies in mixed grass prairie in Phillips County, Montana (Childers and Dinsmore 2008). In addition, revisits to sites where we detected plovers showed they were using these sites for nesting.

Plover nesting success on prairie dog colonies has only been reported for mixed-grass prairie in Montana (Dinsmore et al. 2002). Baker et al. (2000) reported greater predation rates on artificial nests placed on black-tailed prairie dog colonies compared with off-colony sites. However, artificial nests were not defended by adult birds, and mountain plovers are known to charge and chase ground squirrels, including black-tailed prairie dogs, which may reduce nest predation (D. Augustine, unpublished data; Knopf and Wunder 2006). Mountain plover nest survival studies in multiple habitats throughout the plover range, combined with demographic modeling, suggest that survivorship of broods and adults, rather than nest survival, has the greatest influence on plover population dynamics (Dinsmore et al. 2002, Dinsmore 2003, Dreitz and Knopf 2007). Most notably, recent telemetry studies in east-central Colorado found that mountain plover chicks hatched on prairie dog colonies had 3 times higher survival rate than chicks that hatched on rangeland sites without prairie dogs (Dreitz 2009). Collectively, these studies from multiple locations and years suggest that although mountain plovers occur in a variety of habitats in the shortgrass steppe, active black-tailed prairie dog colonies may be a particularly important breeding habitat.

A major factor influencing prairie dogs in the western Great Plains is epizootic outbreaks of plague caused by the bacterium *Yersinia pestis* (Collinge et al. 2005, Antolin et al. 2006, Cully et al. 2006, Augustine et al. 2008b). These outbreaks rapidly decimate colonies, causing >95% mortality over a short time period (Cully et al. 2006, Pauli et al. 2006). In northeastern Colorado, plague has created a shifting mosaic that includes prairie dog colonies affected by epizootic plague within the past year, colonies that are recovering as small pockets of survivors or immigrants expand and coalesce (roughly 1–5 years after plague), and larger active colonies that have not been affected by plague in the past several years (Stapp et al. 2004, Savage 2007). A previous study of mountain plover nest sites on expanding colonies (including PNG) recorded plovers nesting on both young (colonized in the past 1–2 years) and old (colonized for >6 years) portions of prairie dog colonies in proportion to availability (Augustine et al. 2008a). My study indicates that plovers also reduce use of prairie dog colonies on PNG during the first 1–2 years after epizootic plague (Table 1).

In addition to their selection for active prairie dog colonies, I found that mountain plovers occurred at similar densities on recent prescribed burns (Table 1). We documented plover

nests or courtship behaviors at all 11 of the burns that we revisited 1–2 weeks after detecting plovers during point counts. These findings provide initial support for the use of prescribed burning to enhance mountain plover breeding habitat in shortgrass steppe. However, little is known about plover breeding success in this habitat. Svingen and Giesen (1999) reported that 22 of 45 nests (49%) on a burn on the Comanche National Grassland in southeastern Colorado hatched successfully, which was similar to nest success rates reported in other sites and habitats (0.33–0.49 in MT, Dinsmore et al. 2002; 0.37 in Colorado, Dreitz and Knopf 2007). In 2008, we observed adult plovers with chicks on 4 of the burns, but I am unaware of estimates of chick survival in burned rangeland. Future efforts to quantify brood rearing success would provide insight to the degree to which prescribed burning can assist in conservation of this species.

I did not address the potential influence of livestock grazing on mountain plover habitat. The entire study area was managed for cattle grazing, and stocking rates were consistent over the past 4 decades, with reductions in the past 2 decades only occurring during droughts in 1994, 2002, 2004, and 2006 (N. Clark and R. Reichert, USDA—Forest Service, personal communication; Fig. 1). Given the potential importance of bare soil exposure for mountain plover breeding habitat, research on the role of grazing management is clearly needed. However, soil exposure may not increase substantially with increased stocking rates because the dominant shortgrasses are highly resistant to grazing (Milchunas et al. 2008). Furthermore, increased stocking rates can lead to losses of perennial cool-season midgrasses and long-term reductions in the sustainability of forage and cattle production (Klipple and Costello 1960, Bement 1969, Milchunas et al. 2008). Future studies comparing the relative influence of prescribed fire, prairie dogs, and livestock grazing on vegetation structure and cover in the shortgrass steppe may provide insight concerning advantages and disadvantages to different approaches to habitat management for mountain plovers.

Future studies may additionally consider interactions between fire and prairie dog management. Prescribed fire can be used to encourage prairie dog colony expansion where such an outcome is desirable (Augustine et al. 2007). Prairie dog colonies experience plague outbreaks more frequently during wet, El Niño years (Stapp et al. 2004), which are also associated with increased grass production and hence fuels for prescribed burns. Given the philopatry exhibited by mountain plovers in their breeding territory (Graul 1973, Dinsmore 2003, Skrade and Dinsmore 2010), conducting burns near prairie dog colonies that have recently been affected by plague may be one strategy to maintain breeding habitat in that locality until the prairie dog colony begins to recover.

One limitation of my study was the inability to provide a confidence interval for plover density in the rangeland habitat, even though we did observe mountain plovers in this habitat through directed, non-random sampling based on past observations. This raises questions concerning: 1)

detectability of plovers in the rangeland habitat compared to burns and prairie dog colonies and 2) factors that may differentiate rangeland sites that are occupied versus not occupied by plovers. Regarding detectability, mean vegetation height on moderately grazed rangeland and plague-affected prairie dog colonies (5.8–8.0 cm) is greater than mean height on active colonies (5.0–5.5 cm; Hartley et al. 2009), but mountain plovers are taller than vegetation in all of these habitats (Knopf and Wunder 2006). Also, plover detections were truncated at 175 m, which reduces the influence of vegetation obstruction at low sight angles. Finally, mean distance for the 8 plover detections under 175 m on the plagued prairie dog colonies and the non-random rangeland sites (94 m) was similar to mean detection distance for burns and active prairie dog colonies (90 m). Childers and Dinsmore (2008) used similar reasoning when applying a detection function derived from other habitat to estimate mountain plover density in rangeland of northern Montana. The non-random rangeland sites where we observed plovers did not differ visibly from random rangeland sites in terms of vegetation height or species composition, but an ongoing study is examining whether more detailed aspects of plant basal area, bare soil exposure, and soil texture may be related to plover presence.

Finally, I conducted my study in years when rainfall was near or above the long-term mean. Given that mountain plover use of sites disturbed by fire or prairie dogs may be related to increased bare soil (Knopf and Miller 1994), it is unclear how important such disturbances may be following drought years, when exposure of bare soil may be greater throughout the landscape. Longer-term monitoring of mountain plover populations is needed to address habitat selection under varying rainfall patterns.

MANAGEMENT IMPLICATIONS

My findings indicate that management for active prairie dog colonies in the shortgrass steppe can be an important means to provide breeding habitat for mountain plovers. Low densities of breeding plovers on prairie dog colonies extirpated by plague in the past 1–2 years suggest that assessments of mountain plover habitat should incorporate factors affecting annual fluctuations in the distribution and size of active prairie dog colonies. My findings also suggest that dormant-season prescribed burns may be an effective means to create breeding habitat for mountain plovers in shortgrass steppe. Measures of brood rearing success on prescribed burns are needed to further evaluate the contribution of burns to plover conservation. Management for mountain plovers may additionally consider the potential for combined management of fire and prairie dogs. Conducting burns near prairie dog colonies that have recently been affected by plague may be one strategy to maintain breeding habitat in a locality until prairie dogs begin to recover.

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