CHAPTER

6

An Assessment of Rangeland Activities on Wildlife Populations and Habitats

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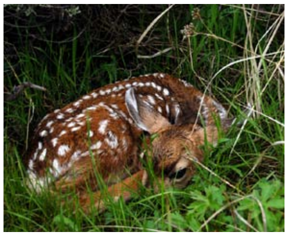
INTRODUCTION

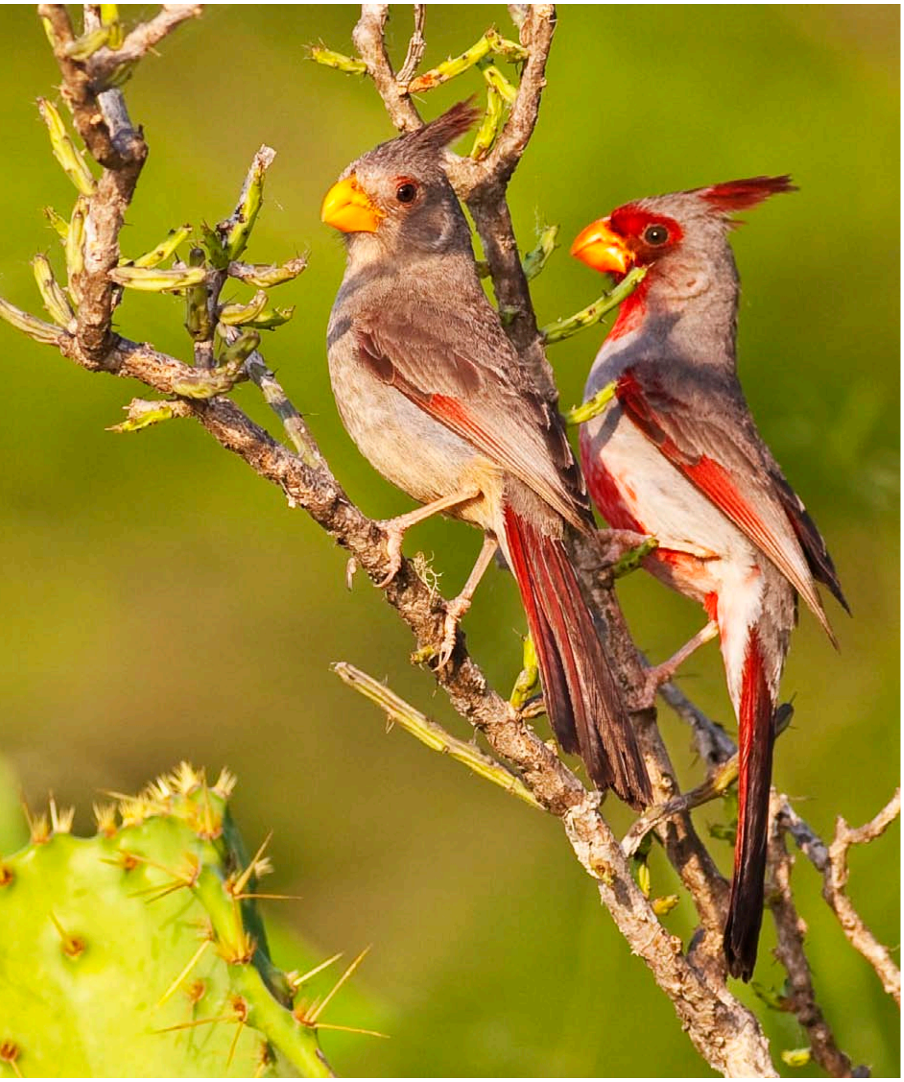
Numerous management practices are applied to rangelands in the western United States to enhance wildlife, including prescribed grazing, burning, brush management, mowing, fencing, land clearing, planting, and restoration to benefit soil and water. Indeed, the Natural Resources Conservation Service (NRCS) lists 167 conservation practices (http://www.NRCS.USDA.gov/technical/standards/nhcp.html). However, wildlife responses to conservation practices are usually species and even species-habitat specific, meaning not only that each species may respond differently to any specific practice but also that a single species may respond differently to the same practice in different vegetation associations or conditions. When managers apply conservation practices to the landscape, habitat is often altered, and managers should understand that the management will benefit some of the wildlife present but may be detrimental to others. Conservation practices were designed to help ecosystem managers think about the variables that accompany any action on the landscape. Each conservation practice has specific purposes that may influence related resource issues. For example, prescribed grazing by large herbivores can alter the structure and function of ecosystems that have direct and indirect effects on wildlife. Primary effects are often described in the literature (Mackie 2000), but there has not been an evaluation of how conservation practices affect wildlife on rangelands. However, practices like prescribed grazing are not a simple treatment but have widely divergent effects, depending on locale, timing, intensity, and species or combination of grazing animals (Briske et al. 2008). Similarly, small mammals, reptiles, amphibians, and bats represent very broad wildlife categories that may have diverse responses to various conservation practices. For example, focusing on Rodentia includes species with such widely different habitat and life history strategies that responses within the group may differ diametrically when exposed to the same management practice. Furthermore, most of the studies that have examined how anthropogenic activities on rangelands influence wildlife have not classified the management activities involved according to the NRCS conservation practices. Thus, we refer to related conservation practices on rangelands that influence wildlife as rangeland activities.

Wildlife in America has been strongly influenced by agriculture; livestock grazing is the most widespread land management practice in the world (Holechek et al. 2003) and affects 70% of the land surface in the western United States (Fleischner 1994). Traditional practices in rangeland management often homogenize grazing lands to increase forage production and maximize sustainable yield for domestic livestock. New management approaches that promote the spatial and temporal scale of heterogeneity in vegetation structure, composition, and biomass so that sufficient tracks of particular vegetation associations can accommodate desired wildlife populations are needed to improve habitat for wildlife (Fuhlendorf and Engle 2001; Bruno and Cardinale 2008).

The dynamics of native and domestic ungulates, combined with various management practices, create a complex interaction that influences plant and animal communities by altering ecosystem structure, nutrient cycling, productivity, recruitment, predator–



Deer fawn on the Theodore Roosevelt Memorial Ranch, Dupuyer, Montana along the Rocky Mountain Front. (Photo: Sonja Smith)



Pyrruhuloxias (*Cardinalis sinuatus*) occupy desert scrub and mesquite-dominated rangelands in southwestern United States. (Photo: Tim Fulbright)

prey relationships, urination and defecation, trampling, and competition. Additional modifications to landscapes, including roads, fences, anthropogenic water sources, agricultural structures, and other developments related to livestock production on western rangelands, modify wildlife behavior and complicate wildlife management. This is especially important for wildlife, as domestic stock and the related anthropogenic developments alter forage availability and cover and contribute to habitat alteration and fragmentation. Large herbivores may potentially modify landscapes in numerous ways (Senft et al. 1987; Ohmart 1996; Fuhlendorf and Engle 2001), but describing them is beyond the scope of this chapter. However, it is not surprising that the effects of prescribed grazing on wildlife have received more attention in the literature than other conservation practices. Many of the early studies of wildlife parallel livestock husbandry and range management theory in that grazing and browsing are the primary factors affecting the kinds, amounts, and quality of forage available (Mackie 2000).

Our objective was to review peer-reviewed literature to examine how conservation practices influence wildlife and wildlife habitats on rangelands in the United States, with specific reference to the NRCS Conservation Practice Standard for Upland Wildlife Habitat Management. The main purpose of this conservation standard is to treat upland wildlife habitat concerns identified during the conservation planning process that enable movement or provide shelter, cover, and food in proper amounts, locations, and times to sustain wild animals that inhabit uplands during a portion of their life cycle. We emphasized the literature compiled in the bibliography by Maderik et al. (2006) but also considered other articles to provide a more complete review.

We documented rangeland activities that influenced (i.e., positive and negative) game birds, nongame birds, carnivores, ungulates, small mammals, reptiles, and amphibians on western rangelands. Carnivores are rarely considered by NRCS, but we include them in our review because of their importance to functioning ecosystems. We also identified gaps in scientific knowledge and recommended future research to enhance management of wildlife on western rangelands in the United States. We supplemented the synthesis with literature outside the United States when similar knowledge within the United States was not available.

RESULTS OF THE LITERATURE ASSESSMENT

Very few of the NRCS conservation practices that directly affect upland wildlife habitat are addressed or evaluated in the peer-reviewed literature. We identified specific activities when appropriate; however, this review is dominated by grazing because of the high profile that grazing has received by the scientific community. Prescribed grazing, when carefully controlled, can be useful in improving habitat for specific species, but the frequency, timing, and intensity of livestock grazing for maximum wildlife benefits are different than those used for maximum livestock benefits (Holechek et al. 1982). For wildlife, the amount of critical residues left after prescribed grazing is more important than the amount removed; the condition of most ranges will deteriorate when greater than 50% of grazable vegetation is used annually (Hyder 1953; Holechek et al. 1982).

More than 25 yr ago, Holechek et al. (1982) reviewed how prescribed grazing could improve wildlife habitat and concluded that the database was limited. They argued that research into how grazing strategies influence wildlife should receive high priority. Unfortunately, peer-reviewed literature evaluating conservation practices for upland wildlife habitat management, including prescribed grazing, has not received high priority, and the complex influences on wildlife and their habitat remain largely unknown.

Rangeland Activities and Habitat for Game Birds

Conservation practices that improve habitat, if identified and implemented, may halt the decline or, in many cases, enhance the viability of game bird populations. Distribution and abundance of native grouse (subfamily Tetraoninae) that symbolize the biological diversity of western grazing lands are in decline (Knick et al. 2003; Hagen et al. 2004) or are already threatened or endangered (Storch 2007). Exceptions include spruce grouse (*Canachites canadensis* L. 1758) and blue grouse (*Dendragapus obscurus* Say 1823) populations and most white-tailed ptarmigan (*Lagopus leucurus* Richardson 1831) populations. Indigenous quail (subfamily Phasianinae) populations, though stable locally, are largely in decline in the desert Southwest (Saiwana et al. 1998; Western Quail Management Plan 2008) and in the southern Great Plains (Veech 2006). Species considered here that are native to western grazing lands include Gunnison (*Centrocercus minimus* Young et al. 2000) and greater sage-grouse (*C. urophasianus* Bonapart 1827); lesser (*Tympanuchus pallidicinctus* Ridgeway 1873), greater (*T. cupido* L. 1758), and Attwater's prairie-chicken (*T. cupido attwateri*); plains (*T. phasianellus jamesi* L. 1758) and Columbian sharp-tailed grouse (*T. p. Columbians* L. 1758); wild turkey (*Meleagris gallopavo* L. 1758); bobwhite (*Colinus virginianus* L. 1758); and scaled quail (*Callipepla squamata* Vigors 1830).

Our synthesis includes U.S. Department of Agriculture (USDA) Conservation Practices Standards that benefit native grouse and quail and is supplemented with information on exotic species (e.g., ring-necked pheasant [*Phasianus colchicus* L. 1758] and grey or "Hungarian" partridge [*Perdix perdix* L. 1758]) that are abundant regionally and provide recreational and economic benefits (Bangsund et al. 2004). We do not synthesize the rich literature for ring-necked pheasant because in-depth reviews for this species response to Farm Bill conservation practices (Haufler 2007) and other management are readily available (Trautman 1982; Berner 1988; Kimmel and Berner 1998).

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We present findings regionally because variation in climatic gradients (Fulbright and Ortega-Santos 2006), disturbance regimes (Coppedge et al. 2008), and contemporary land use change (Foley et al. 2005) influence vegetation response to management. We critically reviewed strength of evidence because variation in study design (Guthery 2007) and ecological scale of investigation (Manzer and Hannon 2005) further influence applicability of research outcomes to management. We placed recommendations within the context of landscape conservation, a well-known ecological principle (Lindenmayer et al. 2008) that is being used in management of game birds at large scales (Hagen et al. 2004; Manzer and Hannon 2005).

Landscape Conservation. Public land managers use holistic strategies that conserve entire landscapes because to be effective the scale at which conservation practices are implemented must match the scale of anthropogenic change that threatens populations. Tillage agriculture (Walker et al. 2007), urban sprawl (Knick et al. 2003; Krausman et al. 2008), tree and shrub invasion (Fuhlendorf et al. 2002), and energy development (Naugle et al. 2011) result in broad-scale loss and degradation of habitat that overwhelms management of remaining fragments (Fuhlendorf et al. 2002; Veech 2006). Wholesale fragmentation increases predation rates (Manzer and Hannon 2005), alters historic disturbance regimes (Baker

2006), promotes the spread of invasive plants (Bergquist et al. 2007), and facilitates disease (Walker et al. 2007). The concept of conserving the remaining "usable space" is a primary underpinning for quail habitat management in the south-central region of the Great Plains (Guthery 1997). Under this paradigm, managers should strive to increase the quantity of quail habitat. Reversing declines in game bird populations will require regional management of remaining usable space (Williams et al. 2004).

Rangeland Activities. Most literature documents the decline or extirpation of wildlife populations that result from chronic overgrazing. Overgrazing is defined here as the combination of stocking rates and timing of grazing that reduces wildlife reproduction and survival by altering the short- and long-term structure and composition of grassland and shrubland vegetation. This chapter may be frustrating for some readers looking for precise guidance because little experimental research has been conducted to know which conservation practices benefit game birds. Most contemporary studies lack experimental controls, are too short in duration, and fail to collect pretreatment data. Moreover, findings cannot be readily translated in conservation practices (e.g., prescribed grazing) because existing studies typically compare wildlife response to grazed and ungrazed pastures without reference to grazing strategy, regime, or system. Implications should not be extrapolated too broadly because they are most often derived from studies of specific species and local-scale management actions.

Grazing. Livestock grazing is a controversial practice because indirect evidence overwhelmingly suggests that overgrazing reduces nest success (e.g., scaled quail [Pleasant et al. 2006], ring-necked pheasant [Clark and Bogenschutz 1999] and greater sage-grouse [Beck and Mitchell 2000]) and brood survival (lesser prairie-chicken [Hagen et al. 2005] and wild turkey [Spears et al. 2007]) by decreasing height and density of herbaceous cover. Livestock grazing can have negative or positive impacts on game bird habitat, depending on timing and intensity of grazing and which habitat component is being influenced (Beck and Mitchell 2000). Light to moderate grazing can promote forb abundance (e.g., food), but heavy grazing reduces herbaceous cover and promotes invasive species (Crawford et al. 2004). Guidelines describing height and density of herbaceous cover necessary to maintain productive habitats are available for many game bird species (Connelly et al. 2000; Hagen et al. 2004). These guidelines provide the "biological sideboards" necessary to guide grazing strategies for maintaining and enhancing populations; unfortunately, the grazing strategies necessary to achieve the necessary cover requirements for game birds are poorly understood.

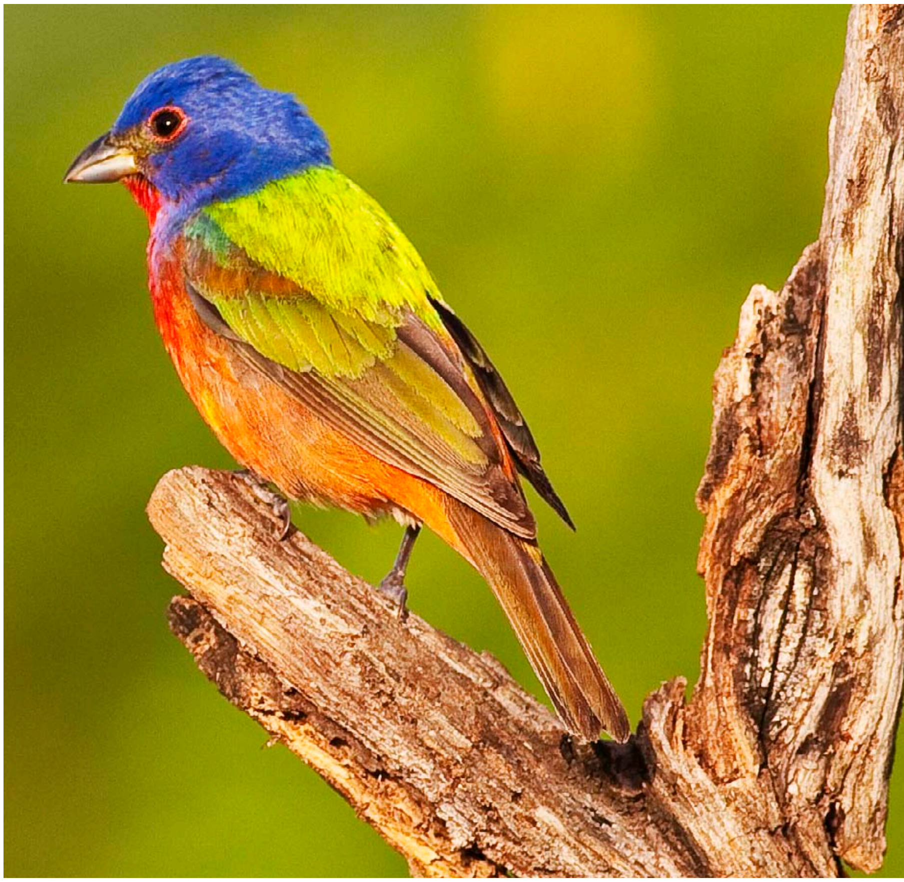
The only empirical evidence of the influence of prescribed grazing on game birds we found in the literature was an unpublished report (Rice and Carter 1982) from a 5-yr study of game birds at Fort Pierre National Grassland in central South Dakota. Authors compared deferred rotation, rest–rotation, and winter-only grazing. Pastures (404 ha) that were deferred from grazing until winter provided the highest number of plains sharp-tailed grouse and greater prairie chicken nests and broods. Rest–rotation grazing accommodated the second-highest density of nests and broods for both species. Deferred rotation did not provide blocks of undisturbed cover available in the spring for nesting, which was reflected in the lowest density of nests and broods. Pastures managed under rest–rotation grazing, which had the highest cattle stocking rate of any system, produced approximately 10 times more nest-broods than did pastures managed in a deferred rotation system. During the 5-yr study, grouse followed the grazing rotation seeking the best herbaceous cover for nesting and rearing broods. Grouse preferred rested pastures for nesting that were at times 4.0 km from breeding sites.

In the south-central United States (e.g., Texas and Oklahoma), grazing management can be prescribed to benefit bobwhite habitat, but a large part of potential quail range in the Rolling Plains has been overgrazed and excessively treated for brush control (Rollins 2007). Today, more landowners are tempering traditional land management goals with more quail-friendly practices, including reduced stocking rates (Rollins 2007). Adequate nesting cover is a key consideration for quail managers

Sage-grouse (*Centrocercus urophasianus*) are an important species of concern in western rangelands. (Photo: Brett Billings)

(Slater et al. 2001) because food is rarely the limiting factor for bobwhites in Texas (Guthery 2000). Livestock grazing can be an effective tool for managing quail habitat, especially in manipulating plant succession (Guthery 1986). But across most of Texas, bobwhite abundance declines as cattle density increases (Lusk et al. 2002). Light to moderate stocking rates that provide 50% grass and 20% to 30% woody vegetation result in adequate bobwhite nesting habitat in western Oklahoma (Townsend et al. 2001). Guthery (1986) emphasizes flexibility in grazing prescriptions to allow "slack" (Guthery 1999) in the system to account for variability in brush cover and short- and long-term precipitation patterns.

Other than the examples mentioned above, little experimental data are available to identify beneficial grazing practices that increase bird populations levels (e.g., greater sage-grouse [Connelly et al. 2000] and lesser prairie-chicken [Pitman et al. 2005]) because mechanisms are poorly understood (Beck and Mitchell 2000; Hagen et al. 2004). Effects of livestock grazing vary regionally because, unlike the Great Plains where bison (*Bos bison* H. Smith 1827) once flourished (Sanderson et al. 2008), many semiarid sagebrush and arid desert ecosystems evolved with substantially less grazing (Connelly et al. 2000; Knick et al. 2003). Wildlife managers in the Great Plains readily acknowledge the importance of livestock grazing to conservation because ranchers whose operations remain profitable are less likely to convert native prairie to cropland (Licht 1997; Higgins et al. 2002). Conversely, wildlife managers in sagebrush and desert grasslands see grazing as detrimental because excessive stocking rates often results in severe habitat degradation (Mack and Thompson 1982; Knick et al. 2003). We need more experimental studies like those in Europe showing how managed grazing was used to recover a declining population of black grouse (*Tetrao tetrix* L. 1758) in northern England (Calladine et al. 2002). Black grouse numbers averaged 6.3% higher per year, and brood survival was 22% higher at sites with reduced grazing than in overgrazed reference sites.



Western populations of painted buntings (*Passerina ciris*) breed in the shrublands of northern Mexico and Texas. (Photo: Tim Fulbright)

Vegetation Manipulations Detrimental to Populations. A host of vegetation manipulations that detrimentally impact game birds include agricultural tillage, herbicide application, mechanical sagebrush removal, and overprescription of fire in xeric landscapes. Tillage agriculture directly reduces the amount of habitat available and fragments remaining grasslands to the detriment of wildlife populations (Swenson et al. 1987). Various means of mechanical and herbicidal removal of sagebrush (*Artemisia* spp.) directly reduce the abundance of shrub and herbaceous vegetation that sage-grouse rely on for food and cover (Wallestad 1975; Braun and Beck 1996). Periodic fire may rejuvenate grasslands in the Great Plains (Reinking 2005; but see Patten et al. 2007), but widespread burning of sagebrush landscapes is not warranted in xeric environments farther west (Beck et al. 2008). Similarly, lesser prairie-chickens in southeastern New Mexico shrublands selected sand shinnery oak (*Quercus harardii* Rydb.) landscapes for thermal refugia and protective overhead cover; selection for these landscapes suggests no justification for shrub control for prairie-chicken conservation in these landscapes (Bell et al. 2010).

“A science-based approach is the key to implementing the right practices in the right places and then documenting outcomes to wildlife populations.”

Exotic and Woody Plant Invasions. Activities that enable proliferation of exotic herbaceous and woody plants (e.g., tree/shrub establishment) in rangelands should be avoided, but those that reduce or remove unwanted invasive species are encouraged (Flanders et al. 2006). Game bird populations have suffered from human fire suppression that promotes tree and shrub invasions and establishment of exotic plants that eventually results in catastrophic wildfire. An increase in tree abundance is associated with lower persistence of lesser prairie-chicken populations in Oklahoma and Texas (Fuhlendorf et al. 2002). Sage-grouse do not use mountain big sagebrush (*Artemisia tridentata* Nutt. *vaseyama* [Rybd.] Bettle) landscapes that are invaded by pinyon (*Pinus* spp.)–juniper (*Juniperus* spp.) woodlands at higher elevations in the intermountain West (Miller et al. 2000; Crawford et al. 2004); the exact mechanism is unknown, but birds either experience higher predation rates or avoid tall structures in otherwise suitable habitats. Similary, scaled quail avoid grasslands invaded by trees in the desert Southwest (Van Auken 2000; Bristow and Ockenfels 2006). Another major problem throughout much of the West is proliferation of cheatgrass (*Bromus tectorum* L.), which reduces viability of game bird populations. Invasion of rangelands by cheatgrass has led to a cycle in which increasing abundance of this annual grass promotes large fires that allow cheatgrass to increase further, causing the loss of perennial bunchgrasses and low-elevation communities of Wyoming big sagebrush (*Artemisia tridentata wyomingensis* Beetle and Young) (Knick 1999; Baker 2006). This phenomenon is particularly troubling because no large-scale restoration techniques are currently available to restore the millions of ha of sagebrush-dominated rangelands that have been lost to wildfire.

Brush Management. South-central Great Plains rangelands have changed greatly over the past century as mesquite (*Prosopis glandulosa* Torr.) savannas become increasingly dense because of a lack of prescribed fire and regrowth from chemical and mechanical brush management. Light to moderate stocking rates usually provide the proper proportions of bare ground, herbaceous quail foods, and woody cover required to sustain bobwhite populations in Oklahoma (Townsend et al. 2001). Grazing intensity will range relative to how much brush is present; lighter stocking rates are required to maintain more herbaceous cover if little brush is present, but heavier stocking rates are possible if more brush canopy is present (Guthery 2002). In the Rolling Plains of Texas, bobwhites selected rangelands containing higher brush canopy cover and overall visual obstruction over those with more bare ground (Ransom et al. 2008). Weather has a tremendous influence on the amount of cattle forage available, leading Lusk et al. (2007) to conclude that reducing livestock stocking rates during dry periods likely will foster ground cover more similar to that available during wet periods. The main factors influencing bobwhite numbers in southern Texas were rainfall during the previous growing season and type of range, with treatments to reduce brush only nominally affecting bird abundance (Cooper et al. 2009). In the same areas of Texas, application of prescribed fire at large spatial scales was deemed a neutral practice for managing bobwhite habitat in semiarid rangelands (Ransom and Schulz 2007).

Strategic Approach to Implementing Beneficial Practices. Implementing practices that are beneficial to game birds is often challenging because many of the critical experiments have not been done to document positive population responses to management. A science-based approach is the key to implementing the right practices in the right places and then documenting outcomes to populations to identify and replicate our successes, manage adaptively to improve delivery, and provide accountability to all our audiences. Implementation of conservation practices should be linked with field-based experimental research to identify the most effective and least expensive ways to benefit wildlife populations. Many birds use habitats at a spatial scale that is larger than that of an individual pasture or ranch. Therefore, our scientific assessments should reflect appropriately large scales at which game bird populations use habitat resources year-round and transcend that of an individual ranch to encompass multiple and nearby ranches enrolled in conservation programs.

The USDA is trying new and innovative ways to link science with implementation to document the benefits of NRCS conservation practices. For example, the USDA launched its new and exciting Sage-grouse Initiative (SGI) in March 2010 to provide a holistic approach to conserving sage-grouse and sustaining working ranches in the West. In its inaugural year, the SGI has quickly become one of the largest and most recent conservation success stories in the West. The SGI's success is in capitalizing on the strong link between conditions required to support sustainable ranching operations and habitats that support healthy sage-grouse populations. The SGI is a science-based initiative with evaluations carried out by reputable, independent scientists to measure the biological response of sage-grouse populations to conservation practices, to assess SGI effectiveness, and to adaptively improve program delivery.

The SGI follows three primary steps in evaluating the benefits of conservation practices that may serve as a model for others dealing with uncertainty in their implementation effectiveness. First, the NRCS worked with the Bureau of Land Management to map rangewide sage-grouse population centers, or "core areas," to refine SGI delivery ensuring that practices benefit large numbers of birds (Doherty et al. 2010). Targeting practices within core areas ensures that enough of the right conservation practices are implemented in the right locations to anticipate a positive population response. Similar guidance is emerging for targeting conservation practices to benefit sustainable bobwhite quail populations in the West Gulf Coastal Plain (Twedt et al. 2007). Second, SGI-sponsored studies are under way in six states across the West to assess benefits of grazing systems and removal of encroached conifer. Assessments incorporate before–after control–impact designs using radio-marked birds across appropriately large time and space scales to quantify the biological and population-level response of birds to conservation practices. Third, the NRCS completed a conference report with the U.S. Fish and Wildlife Service (USFWS) that proactively amends a suite of 40 conservation practices to ensure they are either benign or beneficial to sage-grouse, including upland habitat management, prescribed grazing, and brush management for juniper removal. By conditioning NRCS conservation practices, private landowners enrolled in SGI can rest assured that they can continue normal ranching operations even if USFWS lists sage-grouse as a federally threatened or endangered species. Collectively, these three steps offer an approach for implementing conservation practices while documenting their success and adaptively improving them when necessary.

Rangeland Activities That Improve Habitat for Nongame Birds

Rangeland management has great potential to improve nongame bird habitat (Haufler and Ganguli 2007). To date, most studies address management effects, not necessarily benefits, on focal species or avian communities. This is logical because biologists must first understand the nature of the effects (e.g., positive, negative, or neutral) to effectively use a given management practice as a tool. However, the science has not progressed much beyond this preliminary phase, and experimental studies designed specifically to evaluate management actions to benefit wildlife are rare. We approach the review of conservation practices to improve habitat for nongame birds with a brief mention of key effects papers and then review papers that evaluate the efficacy of management with the primary objective to improve nongame bird habitat.

Effects Papers. By far, the focus of most research has been to address effects of livestock grazing on nongame birds (Fleischner 1994; Saab et al. 1995; Zimmerman 1997). Research

Young white-tailed deer (*Odocoileus virginianus*) in eastern Wyoming. (Photo: David Briske)

Pronghorn antelope (*Antilocapra americana*) on the Charles M. Russell National Wildlife Refuge, Montana. (Photo: Jeffrey Wright)

has been conducted also to understand effects of fire, mowing, and exotic flora and fauna (Herkert et al. 1996; Zimmerman 1997; Askins et al. 2007). Effects are attributed primarily to changes in habitat structure and composition (Bock and Webb 1984), although trampling of ground nests occur. Indirect effects are ascribed to changes in ecosystem structure that can influence ecological relationships among species. The focus of much attention here concerns brown-headed cowbirds (*Molothrus ater* Boddaert 1783) that parasitize nests of many cup-nesting species.

Given that the focus of this chapter is not to review these studies, suffice it to say that effects on species vary from positive to negative. Perhaps the relevance of these effects studies is that they indicate management activities that are benign, beneficial, or detrimental to species, which is a critical first step in developing proactive management prescriptions.

Grazing. Various studies evaluate grazing as a tool to enhance nongame bird habitat. Grazing is not restricted to exotic domestic herbivores but also includes native species, such as bison and elk (*Cervus canadensis* L. 1758). Indeed, many, if not most, ecosystems rely on grazing by native ungulates to influence vegetation structure and composition (Stebbins 1981); thus, some form and level of grazing may be compatible with natural ecosystems processes. Grazing variables that can be manipulated to achieve nongame bird goals include stocking rates, seasonality, duration, and livestock species. A premise of prescribed grazing is that, if done correctly, it will enhance horizontal heterogeneity and provide a mosaic of landscape conditions to meet a wide range of bird preferences (Herkert et al. 1996; Derner et al. 2009).

Wetland Birds. Grazing improved habitat for wading birds in Austria (Kohler and Rauer 1991). Two factors led to degraded habitat: conversion of pastureland to agriculture and the cessation of grazing that allowed for encroachment of common reeds (*Phragmites* spp.) and rushes (*Juncus* spp.) into pastureland. Cattle were introduced to control the encroachment of reeds and rushes, but Kohler and Rauer (1991) noted no tangible increases of wading bird populations. Tichet et al. (2005) evaluated grazing regimes (stocking levels and seasonality) on use by wading birds in French wetlands. They found that grazing intensity affected species responses differently, depending on their habitat requirements. Curlews (*Numenius arquarta* L. 1758) used areas with greater spring grazing intensity, whereas redshank (*Tringa tetanus* L. 1758) occupancy declined. In autumn, lapwings (*Vanellus vanellus* L. 1758) showed a positive relation to grazing, whereas responses by black-tailed godwits (*Limosa limosa* L. 1758) were negative.

Grassland Birds. Paine et al. (1997) compared three grazing regimes in Wisconsin: grass farms, continuously grazed pastures, and "bird-friendly" rotational systems whereby grazing was deferred to create nesting refuges during the breeding season. They reported that refuges attracted 11% more nesting birds than grass farms and that grass farms attracted 65% more nesting birds than continuously grazed pastures. Nest success for grass farms ranged from 6% to 24% and from 30% to 39% for refuges and was 25% for continuous grazing during both years of study. Most nest mortalities for grass farms and refuges were from mowing. Overall avian productivity within refuges was greater than that for grass farms, which were greater than continuously grazed pastures. Productivity is defined as the number of birds fledged from nests. Temple et al. (1999) also compared grazing regimes in Wisconsin and reported that diversity, density, nest success, and productivity of grassland birds was greatest on ungrazed lands. Continuously grazed pastures had the lowest diversity and densities but were intermediate for nest success and productivity. Rotationally grazed pastures had intermediate diversity and densities but the lowest nest success and productivity. They recommended a mosaic of ungrazed and rotationally grazed areas to increase productivity of grassland birds above that found on a mosaic of continuously and rotationally grazed pastures (Temple et al. 1999).

Derner et al. (2009) suggested that livestock could be used as "ecosystem engineers" to modify vegetation structure within and among pastures and provide for habitat needs of grassland birds of the Great Plains. Grazing is often used in combination with patch burning to provide the desired vegetation structure. For example, localized grazing and fire could be used to reduce vegetation cover and provide feeding sites for mountain plover (*Charadrius montanus* Townsend 1853) or nest sites for the long-billed curlew (*Numenius americanus* Bechstein 1812). For the upland sandpiper (*Bartramia longicauda* Bechstein 1812), reduced grazing could be used to provide tall vegetation required for nesting, whereas more intensive grazing could increase food availability and enhance foraging habitat.

Riparian Birds. Livestock grazing can have positive and negative effects on habitats for different species of birds riparian systems. Although grazing removes lower vegetation layers, it also influences seedling establishment and regeneration of shrubs and trees. Indeed, dramatic changes in vegetation structure can be seen shortly after livestock are removed from riparian areas (Krueper et al. 2003). In the Northwest, vegetation recovery following livestock removal in a riparian meadow was complex, given interactions with precipitation (Dobkin et al. 1998). Cattle removal resulted in a more diverse and abundant avian community that was even greater in wet years than in dry years. The northern harrier (*Circus cyaneus* L. 1766), common snipe (*Gallinago gallinago* L. 1758), short-eared owl (*Asio flammeus* Pantoppidan 1763), song sparrow (*Melospiza melodia* Wilson 1810), and yellow-headed blackbird (*Xanthocephalus xanthocephalus* Bonaparte 1826) were found only within the cattle-excluded area. In southeastern Arizona, density of herbaceous vegetation increased four- to sixfold following removal of cattle (Krueper et al. 2003). Mean numbers of detections during bird surveys increased for 42 species (26 significantly) and decreased for 19 species (8 significantly) 3 yr following the removal of cattle. Number of individuals detected per kilometer more than doubled. Detections of open cup-nesting species increased the most and Neotropical migratory birds more than others.

Brown-Headed Cowbird Control.
Reductions in cattle stocking by 86% (752 animal units [AUM] to 103) were made to decrease nest parasitism on the endangered black-capped vireo (*Vireo atricapilla* Woodhouse 1852) in Texas (Kostecke et al. 2003). Rates of cowbird parasitism decreased by 13 times after cattle were removed. Further, cowbirds needed to travel further to breed, resulting in greater energetic costs and reductions in numbers of eggs laid. There was no evidence of cowbird nest parasitism following removal of cattle from a riparian area in southeastern Oregon, even though nest parasitism was prevalent in nearby riparian habitats where cattle remained (Dobkin et al. 1998).

Multiple Range Activities. Walk and Warner (2000) compared burned, mowed, hayed, grazed, and undisturbed management regimes on areas of introduced cool-season grasses, native warm-season grasses, and annual forbs. Eastern meadowlark (*Sturnella magna* L. 1758) and dickcissels (*Spiza americana* Gmelin 1789) were detected most often among grazed warm-season grasses. Henlow's sparrows (*Ammodramus henslowii* Audubon 1829) and field sparrows (*Spizella pusilla* Wilson 1810) were detected more often among undisturbed warm-season grasses where eastern meadowlarks and grasshopper sparrows (*Ammodramus savannarum* Gmelin 1789) were least abundant. Grasshopper sparrows were most abundant among annual weeds where Henlow's sparrows and field sparrows were not observed. Overall abundance was least among recently burned cool-season grasses. Low-intensity late-season grazing was

Mule deer (*Odocoileus hemionus*) on the Theodore Roosevelt Memorial Ranch, Dupuyer, Montana. (Photo: Sonja Smith)

“…cattle removal or reduction seems to be an effective tool to reduce brown-headed cowbird numbers and nest parasitism on open cup-nesting birds.”

important for creating a heterogeneous mosaic to accommodate many of the grassland birds studied.

Griebel et al. (1998) evaluated bird use of two different grazing treatments: 1) bison grazing (year-round; 1.2 AUM · ha^{-1} · yr^{-1}) combined with prescribed fire and 2) cattle grazing (15 May–15 November; 1.0 AUM · ha^{-1} · yr^{-1}). Few differences were reported in bird species richness or relative abundance of species between grazing treatments, vegetation density, and height. During 1 of the 2 yr of study, bird species richness was greater in the bison-fire enclosure than in the cattle enclosure; abundances of lark sparrows (*Chondestes grammacus* Say 1823) and mourning doves (*Zenaida macroura* L. 1758) were higher and grasshopper sparrow lower in bison-fire enclosures. Within the bison-fire enclosures, differences existed between burned and unburned transects, with grasshopper sparrow abundance higher in unburned areas and mourning dove and lark sparrow abundances higher in burned areas.

Danley et al. (2004) reported few differences in bird species diversity or abundance between areas that were burned and grazed versus areas only burned in North Dakota. The notable exception was the brown-headed cowbird, which occurred 2.4 times more frequently on burned and grazed plots.

LaPointe et al. (2003) evaluated use of a rest–rotation grazing system targeted to improve plant cover for nesting ducks and grassland birds along the St. Lawrence River, Quebec. They evaluated four methods: cattle removal, grazing augmented with seeding of forage plants, seeding with no grazing, and seasonal grazing after duck nesting. Overall abundance of birds exhibited no change 2 yr posttreatment. However, bobolink (*Dolichonyx oryzivorus* L. 1758) were more abundant in areas that were seeded with no grazing and where cattle grazed after ducks had nested, and red-winged blackbirds (*Agelaius phoeniceus* L. 1756) were more abundant in the two treatments with no grazing.

Rangeland Restoration. Ecological restoration is a management paradigm whose objective is to return conditions to those that existed in the past, typically those that occurred prior to European settlement of North America. Implicit is that, in doing so, avian community structure and composition will be restored also. At this point, results of the few studies that have evaluated effects of restoration of birds are equivocal.

Fletcher and Koford (2003) evaluated effects of restoring native grasslands from former agricultural (e.g., hay land and row crops) land and reported that 16 of 54 species detected increased with restoration. Only killdeer (*Charadrius vociferus* L. 1758) and cowbird responded negatively to restoration. Five of the species that increased are of broad regional concern because populations are declining. In contrast, Van Dyke et al. (2004) found no bird responses, positive or negative, to the use of fire and mowing to restore tallgrass prairie in Iowa. Results may have been influenced by the small scale (< 10 ha) of treatments.

In southeastern Arizona, Malcolm and Radke (2008) evaluated effects of active wetland and riparian restoration following passive restoration (e.g., cattle removal) on bird density and diversity. Cattle removal occurred in 1980 and was followed by active restoration in 2005. Active restoration consisted of installation of erosion control gabions to create two wetlands that were then used to irrigate a desert scrub plot. Bird densities increased by 2.3 birds · ha^{-1} in 2006 and 8.4 · ha^{-1} in 2007 following active restoration treatments. Species richness showed a marginal difference.

Kennedy et al. (2008) compared cover by native versus nonnative plants and the resulting influence on nest productivity of passerine birds. They reported no association between the percentage of nonnative plant cover and nest densities, clutch size, productivity, nest survival, and nestling size.

Overall, studies evaluating effects of range management directed at improving nongame bird habitat are rare. Many studies are essentially case studies whose results apply largely to the place and time of study. As a result, generalizations are difficult at best. Some trends that emerged from the papers reviewed are that continuously grazed pastures appear to have fewer birds and fewer species than areas grazed using a rotational system, grazed after the breeding season, or where cattle were removed entirely. Additionally, cattle removal or reduction seems to be an effective tool to reduce brown-headed cowbird numbers and nest parasitism on open cup-nesting birds.

Rangeland Activities and Habitat for Carnivores

References regarding influences of rangeland activities on carnivores are notably sparse and are rarely considered by the NRCS. However, we include them in this review because of their importance to functioning ecosystems. We considered 14 taxa to be representative of western rangeland habitats: coyote (*Canis latrans* Say 1823), wolf (*Canis lupus* L. 1758), kit fox (*Vulpes macrotis* Merriam 1888), swift fox (*Vulpes velox* Say 1823), red fox (*Vulpes vulpes* L. 1758), grey fox (*Urocyon cinereoargenteus* Schreber 1775), black bear (*Ursus americanus* Pallos 1780), grizzly bear (*Ursus arctos horribilis* L. 1758), mountain lion (*Puma concolor* L. 1771), bobcat (*Lynx rufus* Schreber 1777), raccoon (*Procyon lotor* L. 1758), striped skunk (*Mephitis mephitis* Schreber 1776), spotted skunk (*Spilogale* spp.), and black-footed ferret (*Mustela negripes* Audubon and Bachman 1851).

Based on references in the bibliography of Maderik et al. (2006), rangeland activities appear to influence habitat for spotted skunks and striped skunks (Neiswenter and Dowler 2007). Spotted skunks use areas with more large mesquites than striped skunks, and striped skunks did not select any habitat relative to its availability, but both species appeared to avoid agricultural areas. Conservation of western spotted skunks may be enhanced by limiting brush control for management of livestock on mesquite dominated rangelands (Neiswenter and Dowler 2007).

Others reported that the distribution and shape of grassland patches, woodland patches, pastureland, and farmsteads influenced detections of striped skunks, raccoons, and red fox. Kuehl and Clark (2002) determined that evidence of striped skunks decreased as distance from grassland patches increased but, in contrast to Neiswenter and Dowler (2007), was positively associated with the number of farmsteads in their study area. Raccoon presence was positively related to presence of woody cover, and red fox presence increased with greater area of pastureland and greater isolation from farmsteads but decreased with increasing amounts of habitat arranged in strips across the landscape. Ivan et al. (2002) reported that alteration of prairie landscapes through increases in planted trees, woody cover, rock piles, and junk piles enhanced conditions for striped skunks and raccoons by providing denning habitat. Maestas et al. (2003) concluded that ranchlands supported relatively more coyotes than exurban developments and that ranches are important for protecting biodiversity, suggesting that future conservation efforts may require less reliance on reserves and a greater focus on private lands. In ecologically similar areas of Arizona, Horejsi (1982) reported that coyotes were relatively more abundant on ungrazed than on grazed rangelands; however, the ungrazed area had been closed to predator control for an extended period of time prior to the initiation of his research, and the other had not. These results have implications for predicting the influence of rangeland management practices on specific species of carnivores and, through their affects on

Rio Grande wild turkeys (*Meleagris gallopavo intermedia*) are found along riparian areas and in shrublands. (Photo: Tim Fulbright)

"...influences of rangeland activities on large native carnivores have nearly all been negative."

landscape configuration, on conservation of biodiversity in general.

Hilty and Merenlender (2004) emphasized that wide, well-vegetated riparian corridors are important in maintaining the connectivity of native predator populations to ensure their long-term survival. In a similar riparian system, Ammon and Stacey (1997) concluded that livestock grazing reduced streamside vegetation and that grazing could influence predator assemblages and, thereby, affect bird populations directly and indirectly. Cattle grazing did not affect vegetation height or density along edges of pasturelands compared to the interior of pasturelands. Raccoons and other predators may move more freely in pasturelands when compared with edges of pasturelands, thereby explaining an absence of differences in predation risk for nesting grassland birds in those habitats (Renfrew et al. 2005). Conversion of rangelands to irrigated agriculture (i.e., alfalfa, mint, and sugar beets) may have a positive effect on burrowing owls (*Athene cunicularia* Molina 1782) where those small strigids use burrows abandoned by badgers (*Taxidea taxus* Schreker 1777; Belthoff and King 2002); presumably, such practices have a negative affect on badgers although not explicitly stated.

Numerous references included in the bibliography by Maderik et al. (2006) (Beck and Mitchell 2000; Townsend et al. 2001; Herkert et al. 2003; Cox et al. 2005; Miller and Guthery 2005; Renfrew et al. 2005; Shochat et al. 2005; Sutter and Ritchison 2005; Grant et al. 2006) make inferences about onerous affects of grazing on predator assemblages or the ability of predators in general to better detect and prey on the nests of ground-nesting birds. Results of these investigations address primarily changes in predation risk to ground-nesting birds as a result of modifications to habitat structure or composition rather than changes in carnivore populations themselves.

Generalizations About Overall Effects of Management on Carnivores

Habitat alteration and loss and harvesting for sustenance, sport, and profit have resulted in substantial declines in top predators in a wide variety of habitats (Bruno and Cardinale 2008), including rangelands of western North America (Laliberte and Ripple 2004). Overgrazing of rangelands by domestic livestock, sometimes combined with other practices, has influenced the structure and composition of rangeland habitats, with resultant impacts to biodiversity and ecosystem function (Blaum et al. 2007). Additionally, efforts to enhance livestock production on those rangelands have included attempts to eliminate carnivores viewed largely as predators of livestock. As a result, influences of rangeland activities on large native carnivores have nearly all been negative. Nevertheless, some medium-sized carnivores (e.g., coyotes, skunks, and raccoons) have experienced increases in populations and distribution, in part resulting from an enhanced food base associated with human presence or the absence of predators that no longer compete with or prey on those carnivores.

Four of the taxa (i.e., wolf, grizzly bear, black-footed ferret, and San Joaquin kit fox [*Vulpes macrotis mutica* Merriam 1902]) have been impacted by activities associated with rangeland management (i.e., predator control activities, habitat modification, and conversion) to the extent that they have been afforded federal protection under the Endangered Species Act. Two others (i.e., mountain lion and swift fox) have suffered substantial reductions in distribution and numbers.

Throughout much of the history of western North America, ranchers and other livestock producers have viewed large carnivores as incompatible with production objectives. Ranchers and other rangeland managers viewed predator management as an augmentation of the efficacy of other practices, and, as such, it has become a widespread and accepted practice throughout much of the United States. Although predator control is not explicitly one of the NRCS rangeland management practices currently in place, it has been (and in some cases likely will continue to be) an activity that occurs in conjunction with current NRCS practices that place an emphasis on habitat quality and enhancement. As such, a brief history of predator management and its impacts on species and ecosystems is warranted in this chapter. Moreover, some carnivores have benefited from implementation of selected NRCS management practices and warrant recognition.

Widespread efforts to eliminate wolves and grizzly bears from rangelands in the 48 contiguous states were largely successful (Young 1944; Storer and Tevis 1955; Mech 1970; Brown 1985), and the use of a variety of techniques, including widespread campaigns of poisoning, trapping, and shooting, ultimately resulted in the previously mentioned classification of those large carnivores as endangered taxa. Another large carnivore, the mountain lion, also was the object of less successful but still intensive (Bruce 1953; Hert and McMillin 1955) efforts to reduce impacts to livestock operations.

Gray wolves once ranged throughout much of North America but were systematically eliminated from the majority of historical habitats in part because of the threat to livestock (Musiani and Paquet 2004). Indeed, it is estimated that wolves had been eliminated from greater than 85% of their former range in rangeland habitats prior to restoration efforts (Laliberte and Ripple 2004). Nevertheless, federal protection, combined with efforts to manage wolves in the north-central United States (Mech 1970) and efforts to restore them within historical ranges in the northern Rocky Mountains (USFWS 1987), has been successful. Wolves remain important predators of livestock, but current management strategies include provisions for removal of offending individuals.

Grizzly bears once occupied suitable habitat across a wide expanse of the continental United States, but their geographic range has been reduced by 91% in temperate grasslands, savannas, and shrublands and by 100% in desert and xeric shrublands (Laliberte and Ripple 2004), largely a result of efforts to eliminate historic conflicts with livestock grazing and other human activities. Grizzly bears were afforded protection under the Endangered Species Act in 1975, and an initial recovery plan was completed in 1982 and revised in 1993 (USFWS 1982, 1993). Currently, grizzly bears are categorized as 1) an experimental, nonessential population segment in parts of Idaho and Montana and 2) a recovered distinct population segment in the Greater Yellowstone Ecosystem of Idaho, Montana, and Wyoming. Elsewhere in the continental United States, grizzly bears remain listed as threatened, but the status of populations inhabiting the Cabinet-Yaak Recovery Zone, the Selkirk Recovery Zone, and the North Cascades Ecosystem Recovery Zone are under review (USFWS 2008). Recovery of grizzly bears is dependent on the maintenance of suitable habitat in occupied areas and judicious management of individuals that prey on livestock.

Populations of swift foxes and kit foxes declined substantially as a result of rangeland activities, and their influences, including habitat loss through conversion of native prairies, trapping, predator control, shooting, collisions, and use of rodenticides to control prey populations, likely contributed to the decline of swift foxes (Carbyn 1995; Meaney et al. 2006). Further, unanticipated trophic cascades due to widespread removal of wolves and subsequent increases in coyotes and, potentially, red foxes, which prey on or compete with these small canids, likely have contributed to the decline of swift and kit foxes (Carbyn 1995; Cypher et al. 2001; Meaney et al. 2006). Alteration of native prairies due to grazing and agricultural practices has been especially problematic for these foxes, and losses were exacerbated by poisoning, trapping, and other efforts to manage larger predators, including coyotes and wolves (USFWS 1983, 1995).

Mountain lions can be important predators of livestock, particularly domestic sheep, which are grazed widely on western rangelands. Efforts to reduce mountain lion populations were intense during the early 20th century (Bruce 1953; Hert and McMillin 1955), but those activities declined substantially in most of the western states by the 1970s. Nevertheless, it is estimated that the geographic range of mountain lions occupying western rangelands has been reduced by 49%; distribution of those large felids in desert and xeric shrublands has, however, remained unchanged (Laliberte and Ripple 2004). Although mountain lions were successfully eliminated from a substantial proportion of their historical distribution, they remain the most widely distributed large carnivore in North America (Pierce and Bleich 2003). In some areas of the southwestern United States, mountain lion populations have been subsidized by increased food supplies in the form of domestic livestock that allow mountain lions to persist at higher densities, and, as a result, the effects of predation on native ungulates have been exacerbated (Rominger et al. 2004). Increased shrub cover on rangelands often is associated with overgrazing (Blaum et al. 2004), with resultant influences on biodiversity of mammalian carnivores (Blaum et al. 2007) that may enhance hunting efficiency of mountain lions. Reduction of shrub cover on rangelands may decrease hunting efficiency of mountain lions, and conversion of cow–calf operations to steer operations may decrease the benefits of livestock operations to mountain lions and, thereby, reduce their impacts on native ungulates (Rominger et al. 2004). Currently, mountain lions are managed as a game species in the majority of western states, but exceptions occur (Pierce and Bleich 2003; Bleich and Pierce 2005).

Black-footed ferrets have declined substantially in distribution and once were thought to be extinct in the wild. Widespread poisoning campaigns to eliminate prairie dogs (*Cynomys* spp., a principal prey of these endangered mustelids) from rangelands were implicated in the near extinction of that species, as has conversion of rangeland to cropland (USFWS 1988). As a result of a captive breeding program, black-footed ferrets have been translocated to appropriate habitats in several states but remain one of the most critically endangered mammals in North America.

Coyotes have been an unanticipated beneficiary of widespread efforts to reduce wolves, and the distribution and range of coyotes have increased substantially as a result. Although direct mortality of coyotes due to wolf predation was low, results of recent research are consistent with the hypothesis that coyote abundance is limited by competition with wolves (Berger and Gese 2007). Trophic cascades involving wolf removal and resultant expansion of the distribution of coyotes, a generalist predator, have resulted in further impacts to smaller canids, including swift fox and kit fox (Cypher et al. 2001). Coyote control is an important rangeland activity, and substantial research on control efficacy and methodology has been conducted (Knowlton et al. 1985, 1999; Shivik 2006). Coyotes remain an important predator of livestock, particularly domestic sheep, but government-subsidized predator control alone has failed to prevent a decline of the sheep industry (Berger 2006). Coyote control to benefit livestock production can, however, have a positive effect on native ungulates, including mule deer (*Odocoileus hemionus* Rafinesque 1817) and pronghorn (*Antilocapra americana* Ord 1815; Harrington and Conover 2007). Similarly, interference competition by wolves with coyotes has a positive influence on survival of pronghorn fawns (Berger et al. 2008).

In general, mammalian carnivores have benefited little from rangeland management activities. An exception is the coyote, a generalist predator that has expanded its distribution substantially as a result of the extirpation of the wolf from the majority of its historical range. Such shifts have, however, had detrimental affects on other native carnivores. It is well established that predators play a vital role in maintaining structure and stability of communities and that removal of predators can have a variety of cascading, indirect effects (Terborgh et al. 2001; Duffy 2003). Indeed, impacts of rangeland activities that have targeted predators for reduction to enhance livestock productivity extend far beyond the anticipated outcomes. Further, current investigations of trophic cascades resulting from the elimination of top predators can have implications beyond the immediate ecosystems occupied by those carnivores (Berger et al. 2001). Moreover, reduction of top carnivores can lead to unanticipated detrimental impacts to species that may otherwise not have been preyed on as a result of mesocarnivore release, whereby midsized carnivores benefit from a reduction in the numbers or densities of top carnivores (Berger et al. 2008). Thus, a consequence of the elimination of many carnivores from rangelands in North America has resulted in indirect impacts to other species and other than the rangeland ecosystems from which the carnivores in question were eliminated.

Rangeland Activities and Habitat for Native Ungulates

Because livestock and wild ungulates share rangelands, managers have examined the influence of cattle and domestic sheep on the vegetation used by white-tailed deer (*Odocoileus* *virginianus* Zimmerman 1780), mule deer, elk, bighorn sheep (*Ovis canadensis* Shaw 1804), and pronghorn. In general, livestock using ranges shared with wildlife have historically had more negative than positive influences on ungulates, and grazing is not always considered an important conservation practice with beneficial outcomes. However, some studies examined how livestock influenced vegetation but did not present data related to how those influences altered productivity and recruitment of ungulates. Below are examples of studies that examined the use of prescribed grazing as a conservation practice for several ungulate species.

Pronghorn. Pronghorn populations have declined on the Anderson Mesa, Arizona, and cattle were considered a key factor in altering habitat. Five years after cattle were removed from Anderson Mesa, hiding cover (for fawns) increased by 8% at a distance of 5 m, but no differences were reported at 10 or 25 m (Loeser et al. 2005). Forb richness decreased in the fifth year after cattle removal by 16% but not in the following year, and canopy cover was unaffected. It will likely take longer than 5 yr of cattle absence to reverse damage that has occurred to this fragile environment, or some mechanism other than grazing was involved. However, pastures grazed by livestock conservatively or moderately were not used by pronghorn in New Mexico (Jamus et al. 2003).

In the Desert Experimental Range, Utah, pronghorn distribution was related to domestic sheep grazing, black sagebrush (*Artemisia nova* Beetle and Young), and topographic characteristics. Pronghorn-selected areas ungrazed by cattle and areas used moderately by sheep during dormant periods were not favorable for pronghorn (Clary and Beale 1983). Nevertheless, Mosley (1994) suggests that grazing rangelands by domestic sheep can be beneficial to wildlife habitat. However, Schwartz et al. (1977) suggest that pronghorn coexist on rangelands more successfully with cattle than with sheep.

White-Tailed Deer. Most of the studies examining livestock interactions with white-tailed deer have documented how deer respond to livestock under different grazing systems. From these data, conservation practices have been recommended. In general, white-tailed deer avoid livestock, and livestock operations are more profitable when deer are not considered in the operation (Bernardo et al. 1994). Conversely, returns from livestock were maximized when wildlife was not considered; however, small reductions in net gains (from livestock) can improve wildlife habitat (Bernardo et al. 1994).

The diets of white-tailed deer and cattle are different (i.e., deer consume forbs, and cattle consume more grass), and deer are more sensitive to grazing treatments than cattle. To enhance forage for white-tailed, cattle should be stocked at moderate rates with continuous grazing (or even less intensive grazing) to create environments where deer can select more forbs (Ortega et al. 1997a, 1997b). Dietary protein for growth and lactation of white-tailed deer was not met with short-duration or continuous grazing. However, the latter system may provide deer with more diversity and greater nutrition (Ortega et al. 1997b). Deer avoided concentrations of cattle and travel farther under short-duration than continuous grazing systems (Cohen et al. 1989). However, home ranges of white-tailed deer were not significantly different under short-duration or continuous grazing systems (Kohl et al. 1987). They also avoid anthropogenic water sources in short-duration grazing systems because of disturbance from humans, fences, and livestock (Kie 1991). Anthropogenic water sources for white-tailed deer should be on the periphery of short-duration grazing systems if it needs to be supplied (Prasad and Guthery 1986; Kie et al. 1991).

There are fewer studies examining how prescribed grazing by domestic sheep influenced white-tailed deer (Ekblad et al. 1993). In Texas, Darr and Kelebenow (1975) reported a negative relationship between domestic sheep and white-tailed deer due to removal of cover by the former.

Mule Deer. Overall, the best practice related to grazing for mule deer is to minimize cattle numbers on deer ranges. Moderate to heavy use of deer ranges by cattle reduced hiding cover (Loft et al. 1987), caused shifts in habitat (Loft et al. 1991, 1993), increased competition for forage (especially at high stocking rates and in

"livestock using ranges shared with wildlife have historically had more negative than positive influences on ungulates, and grazing is not always considered an important conservation practice with beneficial outcomes."

Elk (*Cervus canadensis*) in Rocky Mountain National Park, CO. (Photo: David Briske)

dry years; Smith 1949; Kie et al. 1991; Yeo et al. 1993), and influenced foraging behavior (Loft et al. 1993; Kie 1996). Mule deer avoided pastures occupied by cattle (Wallace and Krausman 1987; Austin et al. 1983; Austin and Urness 1986; Bailey and Rogotzkie 1991).

However, several investigators examined how forage removal influenced mule deer and reported that mowing at 50% removal can increase grass and total biomass the following spring but that fall cattle grazing leaves more nutritious plants available in summer (Taylor et al. 2004). According to some, spring and summer deer ranges can be grazed by cattle an average of 70% (relative utilization) to enhance the ranges the following year (Short and Knight 2003). Burning can also enhance mule deer habitat (Williams et al. 1980).

Domestic sheep grazing deer ranges often benefit deer by improving forage quality in fall and increasing quantity in spring (Rhodes and Sharrow 1990). The degree of range improvement due to grazing by domestic sheep depends on the intensity of grazing and weather. Browse quality will improve with moderate grazing (40% to 55%) that ends by June (Alpe et al. 1999).

Guidelines to improve the quality of winter range for mule deer in the Great Basin were developed by Austin (2000) based on a review of grazing studies. The following guidelines were established to maintain or increase browse production on winter range.

1. Graze livestock between 1 May and 30 June.
2. Alternate grazing by class of livestock.
3. Use rest–rotation with yearly grazing 66% of the total rangeland.
4. Graze livestock to remove 50% of understory grasses and forbs.
5. Balance deer browsing in winter and livestock grazing in spring.
6. Monitor utilization using permanent plots.

Elk. Studies examining how livestock influence elk were similar to other ungulates examined; most work concentrated on the influence of livestock on forage and did not directly examine population effects. Overall, cattle use of elk ranges had little influence on forage quality when stocked at 3.7 ha · AUM^{-1}, but it did influence the quantity of forage available for elk (Dragt and Havstad 1987). Others (Wambolt et al. 1997) reported similar results when the nutritional values of forage were measured.

Understanding forage use by wildlife and livestock is important for wildlife and livestock management. Most studies of elk and cattle interactions examined use of pastures under different conditions. Because of the varied management plans for livestock, managers should address multiple herbivore species in relation to environmental and climatic variation (Werner and Urness 1998). For example, in Utah, elk did not influence available forage for cattle in June and August 1994, but use by cattle was greater in areas not used by elk in two of three rested pastures in June–August 1995. Cattle grazing reduced preferred winter elk forage in the initial growing season in Montana, but by the second season, the standing crop was similar to the ungrazed control (Jourdonnais and Bedunah 1990). Intensive cattle grazing in Washington decreased elk use of ranges in 1 of 3 yr by 28% (Skovlin et al. 1983).

Limited research has demonstrated how livestock grazing can improve elk forage and increase elk numbers. The Bridge Creek Wildlife Management area in northwestern Oregon was grazed by cattle without a prescribed grazing system and supported 120 elk during winter over 13 yr. When a livestock grazing plan was initiated that incorporated rotational grazing, water distribution, properly located fences, salt placement, creation of a wildlife sanctuary, and closing roads, forage quality improved for elk and cattle, the elk population increased to nearly 1 200 animals, and AUM months for cattle grazing increased by 2.6 times (Anderson and Scherzinger 1975).

In other studies, elk shifted habitats when cattle were introduced (Wallace and Krausman 1987) and selected rested pastures over those used by cattle temporally (Frisina 1986; Yeo et al. 1993), even though fall cattle grazing and mowing (70% and 50% removal, respectively) can increase green vegetation the following spring (Frisina 1986; Short and Knight 2003; Taylor et al. 2004).

Impacts to elk range from domestic sheep depend on climatic conditions and grazing intensity. The quality of browse may improve with moderate grazing of sheep (40% to 55%) that ends by June (Alpe et al. 1999). Others (Rhodes and Sharrow 1990) suggest that at a stocking rate of 125 to 143 female-days · ha^{-1}, domestic sheep can improve forage quality in fall and forage quantity in spring. Carefully managed late-spring sheep grazing can improve winter forage quality on elk winter range (Clark et al. 2000).

Bighorn Sheep. Ranges used by bighorn sheep and cattle usually do not overlap spatially, but interactions have been documented (Halloran and Blanchard 1950; King and Workman 1984; Dodd and Brady 1986; Steinkamp 1990). Early reports (Halloran and Blanchard 1950) simply documented the occurrence of both animals, but later reports evaluated the relationships between them. Earlier studies of cattle and bighorn sheep (Spencer 1943; Halloran 1949; Matthews 1960; Arellano 1961) did not demonstrate competition. Habitat preferences for steeper slopes by bighorn sheep and gentler slopes by cattle precluded competition because there was no range overlap. However, Barmore (1962) argued that cattle grazing on gentle slopes has precluded the use of those areas by bighorn sheep, and Bleich et al. (1997) cautioned that extensive use of such areas could affect forage availability for male bighorn sheep in particular. Blood (1961) examined competition between cattle and bighorn sheep in Canada, where 70% of bighorn sheep winter range was used by cattle. He concluded that cattle grazing prevented increases in the bighorn sheep population.

King and Workman (1984) reported different associations between cattle and bighorn sheep in southeastern Utah. They reported bighorn sheep in higher, steeper, and more rugged talus slopes than cattle, which selected lower, gentler slopes and valleys close to roads and developed water sources. In addition, diets of the ungulates were different; cattle diets were dominated by grass, but bighorn sheep were browsers. King and Workman (1984) did not demonstrate that cattle and bighorn sheep competed for space or resources; however, they argued that the spatial separation they observed may result from a "social intolerance—avoidance factor." McCann (1956), Barmore (1962), McCullough and Schneegas (1966), Follows (1969), Ferrier and Bradley (1970), Dean (1975), Wilson (1975), Gallizioli (1977), and Albrechtsen and Reese (1979) argued that bighorn sheep avoid areas used by cattle. Steinkamp (1990) demonstrated that a translocated population of bighorn sheep clearly avoided cattle. As cattle moved into core areas used by bighorn sheep, sheep moved away. Additionally, the closer cattle grazed to sheep, the closer sheep remained near escape cover.

Social intolerance (Geist 1971) can have serious implications because cattle now graze most rangelands that historically supported bighorn sheep (Mackie 1978); 70% of public lands in the 11 most contiguous western states are grazed at least seasonally (US Department of the Interior 1986). Livestock grazing, even seasonally, appears to result in habitat fragmentation (Temple 1984), resulting in the exclusion of sheep from what is otherwise acceptable habitat. Bissonette and Steinkamp (1996) demonstrated that social intolerance can be a potent force influencing habitat use by sheep. Steinkamp's (1990) and Bissonette and Steinkamp's (1996) results pertain, however, to groups newly translocated into unoccupied habitat. Whether social intolerance between cattle and bighorn sheep is universal remains equivocal. Resolution of the dispute is clouded by the almost universal disregard for spatial scale. Lack of consideration of scale effects can have profound implications for management. For example, in 1988–1989, the bighorn sheep population in Aravaipa Canyon, Arizona, was reduced by 52%. Mouton et al. (1991) examined the causes of mortality and concluded they were “probably the result of livestock related viral diseases compounded by nutritional stress.” Because range overlap has been documented to result in sheep mortality by disease transmission, determination of overlap and the scale at which it occurs is most important. Overlap at the level of home ranges may have very different consequences from overlap on specific slopes or valley floor areas. Additionally, temporal overlap at different scales (e.g., seasonal and annual) would appear to have important ramifications for management.

In other areas of the Southwest, grazing by cattle has damaged bighorn sheep habitats (Gordon 1957; McColm 1963; Riegelhuth 1965; Gallizioli 1977). Low precipitation levels ensure that recovery of ranges will take many years, and in some areas damage from livestock grazing may be irreversible. Grazing by cattle has also influenced bighorn sheep habitat in less arid areas (Buechner 1960; Crump 1971; Geist 1971; Brown 1974) by converting grasslands to shrublands (Demarchi 1970).

Bighorn sheep do not do well when they share ranges with cattle. Following the population declines of bighorn sheep of the late 1800s and early 1900s, they did not recover as well as other native ungulates (e.g., mule deer). Bighorn sheep are not as tolerant as other native North American ungulates to poor range conditions, intraspecific competition, overhunting, and habitat alteration. In addition they are much more susceptible to diseases of livestock than other rangeland wildlife, especially diseases of domestic sheep.

Diseases of cattle that influence bighorn sheep are poorly documented, but diseases contacted from domestic sheep have played an important role in bighorn sheep mortality. Throughout the western United States, die-offs of bighorn sheep and population declines have occurred following the introduction of domestic sheep. Mortality was the result of competition for forage and space and shared diseases (Goodson 1982). According to Goodson (1982), “Co-use of ranges by domestic and bighorn sheep has been consistently linked with declines, dieoffs, and extinctions of bighorn populations from historic to recent times. While much of the evidence for competition between domestic sheep and bighorn sheep is circumstantial, it is sufficiently strong to have prompted management decisions against co-use of ranges by bighorn and domestic sheep by federal land management agencies and state wildlife departments.” The Technical Staff of the Desert Bighorn Council (1990) reviewed 24 interactions between bighorn sheep and domestic sheep and found that bighorn sheep died as a result of all interactions. Recent experimental studies confirmed field observations; when bighorn sheep are exposed to domestic sheep, bighorns die from *Pasteurella haemolytica* (Foreyt 1989, 1990, 1992; Silflow et al. 1993; Foreyt et al. 1994).

The actual mechanisms that kill bighorn sheep after they come in contact with domestic sheep are poorly documented (Jessup 1985), but two trends appear clear (Technical Staff of the Desert Bighorn Council 1990): 1) a large portion of the bighorn sheep population dies, and (2) domestic sheep do not suffer ill effects because of their contact with bighorn sheep. Bighorn sheep are more susceptible to diseases they share with livestock. Domestic animals have been selectively bred for disease resistance, but bighorn sheep have not evolved with resistance to the complement of diseases they are now exposed in the presence of domestic stock. As a result, they have not developed effective immunity against livestock diseases. Silflow et al. (1991) examined domestic and bighorn sheep and concluded that they had different control mechanisms for lung metabolism, and differences in the metabolites released led to different regulation of lung defense mechanisms.

Disease. Biologists are not aware of all the factors creating negative interactions between domestic stock and bighorn sheep, but scabies, chronic frontal sinusitis, nematode parasites, pneumophilic bacteria, foot rot, parainfluenza III, bluetongue, sore mouth, paratuberculosis, and pinkeye are documented decimating factors to bighorn sheep (Jessup 1985).

Bighorn sheep have coexisted with humans for ≥ 30 000 years but now face a precarious future. They are an ecologically fragile species, adapted to habitats that are increasingly fragmented. Fragmentation of habitats increases when cattle share the same rangelands as bighorn sheep. Domestic sheep pose an even greater threat to bighorn sheep.

Small Mammals, Reptiles, and Amphibians

There are few studies that address specific conservation practices for small mammals, reptiles, and amphibians (i.e., access control, fences, closing mine shafts, and ponds). While limiting human access has positive effects on big game survival (Rowland et al. 2000), direct data are lacking for effects on small mammals, reptiles, and amphibians that are not directly harvested by humans. There is evidence that trampling caused by high amounts of human access (e.g., hiking and off-road vehicles) does affect the occurrence of small mammal species in montane (Liddle 1975) and urban habitats (Dickman and Doncaster 1987). Off-road vehicle use has been directly attributed to desert tortoise (*Gopherus agassizi* Cooper 1863) and Couch's spadefoot (*Scaphiopus couchi* Baird 1854) declines in California (Berry 1986). Such data suggest that controlling human use by limiting access may be effective in enhancing habitats for small mammals, reptiles, and amphibians.

In rangelands, fences often provide added vegetative cover resulting from different microclimates and seed deposition by birds (Holthuijzen and Sharik 1985). There is some research examining the effects of fences on small mammals. Merriam and Lanoue (1990) used radiotelemetry and showed that white-footed mice (*Peromyscus leucopus* Rafinesque 1818) in farmlands preferentially traveled along fencelines, even when associated vegetative structure was less than 1 m wide.

A primary concern should be for bat maternity roosts and hibernacula. Mohr (1972) provided data on the importance of these cave resources for bats, and Jagnow (1998) reviewed an example of effective closure to restrict human access but to leave openings for ingress and egress by bats.

Decreased vegetative cover at the edge of stock ponds resulting from cattle grazing was correlated with decreased abundance of Columbia spotted frogs (*Rana luteiventris* Thompson 1913) in Oregon (Bull and Hayes 2000). Livestock effects on water quality were correlated with decreased larval diversity and abundance of amphibians in Tennessee (Schmutzer et al. 2008). However, the effect of cattle on terrestrial habitat quality and postmetamorphic survival of amphibians is yet to be quantified.

Prescribed Grazing and Upland Wildlife Habitat Management

Small Mammals. The greatest proportion of literature documenting effects of grazing on small mammals has focused on rangeland and riparian areas in the western United States. The density of aboveground biomass is important in structuring small mammal communities. Grant et al. (1982) indicated that small mammal communities and their response to grazing varied widely. Tallgrass communities tend to occur in areas of reliably high soil moisture and provide a high ratio of vegetation to seed with large accumulations of litter. These communities support highly variable populations of herbivorous litter-dwelling small mammals with high reproductive rates that can consume large amounts of vegetation. Grasslands of intermediate productivity have low biomass and low diversity of omnivorous and primarily surface-living small mammals, but both forage consumption and reproductive output are somewhat lower than in tallgrass prairie. Shortgrass prairie supports high biomass and high diversity of relatively long-lived omnivorous or granivorous species that reproduce opportunistically with precipitation and that use available resources (seeds and insects) intensively. Communities that differ in species composition, niches, and trophic dynamics are expected to differ in their responses to grazing. Land managers should anticipate that small mammals associated with herbaceous or shrub cover will decline when cattle remove this cover (Moulton et al. 1981; Giuliano and Homyack 2004; Johnston and Anthony 2008). Livestock grazing removes standing plant biomass but also prevents accumulation of ground litter that may influence small mammal community

Black-tailed prairie dogs (*Cynomys ludovicianus*) are common throughout the Great Plains and grazing facilitates colony expansion. (Photo: David Briske)

Rocky Mountain bighorn sheep (*Ovis canadensis*), Yellowstone National Park. (Photo: Jerod Merkle)

composition, plant species growth, and seedling establishment via shading and changes in soil temperature and moisture (Fowler 1988). In southwestern grasslands and shrublands, grazing and fire result in rodent communities dominated by heteromyids (family Heteromyidae; pocket mice, kangaroo rats, and kangaroo mice) instead of murids (family Muridae; rats, mice, hamsters, voles, lemmings, and gerbils) on mesic sites. In more arid sites, grazing and fire favor kangaroo rats (*Dipodomys* spp.) over pocket mice (*Perognathus* spp.; Jones et al. 2003).

Most studies demonstrating negative impacts on small mammal populations have attributed those effects to changes in vegetation cover and perceived predation risk (Grant et al. 1982; Uresk and Bjugstad 1983; Heske and Campbell 1991; Hayward et al. 1997) or to long-term changes in plant species diversity (Jones and Longland 1999). However, Steen et al. (2005) provided evidence that forage competition occurs between livestock and voles, herbivores of greatly differing size. Grazing can either increase or decrease plant community heterogeneity (Adler et al. 2001). Detling (2006) provided the most extensive review of our state of knowledge concerning livestock and prairie dog interactions and concluded that we still cannot accurately determine the effect of prairie dogs on domestic livestock production. However, there is evidence that heavy livestock grazing can facilitate prairie dog colony expansion. Lomolino and Smith (2004) reported that prairie dog colonies had similar species richness of nonvolant mammals, reptiles, and amphibians as adjacent landscapes in Oklahoma but harbored different and more rare and imperiled species. Milchunas et al. (1998) suggested that livestock grazing impacts on other grassland herbivores may depend, in part, on temporally variable short-term trade-offs between plant quantity and plant nutrient quality. Habitat productivity and herbivore densities may mediate shifts from facilitative to competitive interactions between different-sized herbivores (Krueger 1986; Cheng and Ritchie 2006). Field voles (*Microtus agrestis* L. 1761) in Denmark showed a skewed quadratic response to grazing intensity (Schmidt et al. 2005) with population biomass and productivity at light to intermediate grazing intensity slightly greater than ungrazed and much greater than heavily grazed sites. Grazing on these sites reduced thick vegetative cover and promoted more nutritional regrowth, and this species of vole responded much the way livestock do. Steen et al. (2005) reported that field voles in Norway responded similarly but that bank voles (*Clethrionomys glareolus* Schreber 1780), whose diet differs, did not respond to sheep grazing.

Reptiles and Amphibians. Kazmaier et al. (2001) detected no differences in survival or demography of Texas tortoises (*Gopherus berlandieri* Agassoz 1857) between moderately grazed (short-duration, winter–spring rotational grazing regime; 6–28 AUM d · ha^{-1} · yr^{-1}) and ungrazed sites in the Western Rio Grande Plains, Texas. Brodie (2001) examined freshwater turtles across North America and suggested that increased siltation and soil compaction resulting from overgrazing in riparian areas could impact reproduction of freshwater turtles.

Smith and Ballinger's (2001) review indicated that lizards that sit and wait in open habitats (e.g., collared lizard [*Crotaphytus collaris* Say 1823], lesser earless lizard [*Holbrookia maculate* Girad 1851], and side-blotched lizard [*Uta stansburiana* Baird and Girard 1852]) tend to be positively affected at the population level by livestock grazing, whereas active foragers that need vegetative cover (e.g., western whiptail [*Cnemidophorus tigris* Baird and Girard 1852], western stone gecko [*Diplodactylus granariensis* Starr 1879], fine faced gecko [*Diplodactylus pulcher* Steindachner 1870], desert spiny lizard [*Sceloporus magister* Hallowell 1854], bunch grass lizard [*Sceloporus scalaris* Weigmann 1828], and Baja California bush lizard [*Urosaurus nigricaudus* Cops 1854]) tend to be negatively affected. Fair and Henke (1997) indicated that Texas horned lizards (*Phrynosoma cornutum* Harlan 1825) selected for burned plots and did not select for grazed plots in southern Texas. Lizard community composition in Arizona and desertified arid grasslands (Castellano and Valone 2006) was significantly different between inside and outside a grazing exclosure. Analysis of tail-break frequencies suggested that higher predation rates outside the exclosure may have contributed to increased abundance of eastern fence lizard (*Sceloporus undulutes* Bosc and Daudin 1801) and side-blotched lizards following livestock removal. In contrast, the round-tailed horned lizard (*Phrynosoma modestum* Girard 1852) was significantly less abundant inside the exclosure.

Knutson et al. (2004) reported that small agricultural ponds in southeastern Minnesota provided breeding habitat for at least 10 species of amphibians. Gray et al. (2004) reported that relative abundance (i.e., average daily capture) of New Mexico and plains spadefoot toads (*Spea multiplicata* Cope 1863 and *S. bombifrons* Cope 1863) was greater at cropland than at grassland playas but that the abundance of other species and diversity of the amphibian assemblage was not affected by surrounding land use. However, Gray and Smith (2005) reported that mass and length of amphibians from playas surrounded by grasslands were greater than those from agricultural playas. They attributed this to altered hydroperiod in playas surrounded by agriculture. Body size is positively related to the probability of survival, reproduction, and evolutionary fitness in amphibians (Gray et al. 2004). Thus, if cultivation of landscapes surrounding wetlands negatively influences postmetamorphic body size of amphibians, restoration of native grasslands surrounding playa wetlands may help prevent local amphibian declines.

Restoration and Management of Rare or Declining Habitats

Manipulating riparian herbaceous cover and stream habitats are conservation practices that have influenced small mammals, reptiles, and amphibians. Endangered Columbia Basin pygmy rabbits (*Brachylagus idahoensis* Merriam 1891) avoided grazed areas with fewer burrows than ungrazed areas (Thines et al. 2004). Grazing and mowing have been used effectively in specific cases to improve habitat for small mammal and reptile species that prefer reduced vegetative cover. Grazing reduced herbaceous and woody cover for the endangered Stephen's kangaroo rat (*Dipodomys stephensi* Merriam 1907) in California (Kelt et al. 2005) and reduced rhizomatous plant growth to facilitate burrowing while increasing sunning spots for threatened bog turtles (*Clemmys muhlenbergii* Schoepff 1801) in New Jersey (Tesauro 2007).

Riparian Herbaceous Cover. Medin and Clary (1989) reported that, after 11 yr of grazing exclusion, small mammal biomass “there are common research needs and recommendations that need to be considered if administrators, managers, biologists, and the public are to better understand how the conservation practices of NRCS apply to upland wildlife on western rangelands in the United States”

was three times greater on an ungrazed aspen (*Populus tremuloides* Mrchx.) and willow (*Salix* spp.) riparian site. Chapman and Ribic (2002) reported that ungrazed stream bank buffer strips supported more small mammals and species than similar grazed areas and that rotational grazing was not different from continuous grazing as applied to small mammal responses. Grazing in wet meadows can have indirect effects on small mammals also. Whitaker et al. (1983) reported that ground-dwelling and fossorial invertebrates in diets of vagrant shrews (*Sorex vagrans* Baird 1857) were replaced primarily by volant species on grazed sites. Klaus et al. (1999) reported that grazing in Wyoming and Montana did not affect reproductive activity but did affect survival of young water voles (*Microtus richardsonii* DeKay 1842), a species of management concern in alpine riparian habitats. Land managers should anticipate that small mammals associated with herbaceous or shrub cover in riparian areas will decline when cattle remove this cover (Moulton et al. 1981; Giuliano and Homyack 2004; Johnston and Anthony 2008). Frog community response to grazing intensity was positively correlated with grazing reduction of palustrine vegetation in an Australian floodplain (Jansen and Healey 2003).

Stream Habitat Improvement and Management. Homyack and Giuliano (2002) reported that northern queen snakes (*Regina septemvittata* Say 1825) and eastern garter snakes (*Thamnophis sirtalis* L. 1758) were more abundant on fenced than unfenced stream banks but that most herptofauna may require longer than their 4-yr study to respond to exclusion from grazing. As with other conservation practices, little is available on how they influence wildlife populations.

RECOMMENDATIONS

Little peer-reviewed research exists that examines the effects of conservation practices on habitat heterogeneity and diversity of wildlife. Most studies that we reviewed failed to collect pretreatment data, lacked experimental controls, had limited or no replication, or were too short in duration. Implications may often be extrapolated too broadly because results are frequently derived from studies of local management actions.

Research needs and recommendations for the different groups of fauna vary. However, there are common research needs and recommendations that apply to all categories that need to be considered if administrators, land use planners and managers, biologists, and the public are to better understand how the conservation practices of NRCS apply to upland wildlife on western rangelands in the United States.

1. Experimentally designed studies with replicates and controls are necessary. These studies need to be conducted so that scientifically reliable data can be collected.
2. Studies have not been designed to understand how NRCS conservation practices apply to wildlife. This can be acquired only through targeted research. Specific studies should be designed to determine how specific NRCS conservation practices influence wildlife and the habitat they depend on, including (but not limited to) access control, access road, brush management, clearing and snagging, conservation cover, diversion, early successional habitat development/management, fence, hedgerow planting, herbaceous weed control, land clearing, reclamation, mine shaft and adit closing, pond, range planting, restoration activities, spring development, tree and shrub establishment, and upland wildlife habitat management.
3. Carnivore management is not an aspect of NRCS conservation practices for upland wildlife, but because of their role in the ecosystem, they need to be considered and managed.
4. One common theme that is constantly emphasized in management theory is the importance of monitoring. Unfortunately, funds are not provided for these important activities. As a result, projects and practices are put in place, management plans are developed, and short-term research is conducted with little or no follow-up. This lack of efficient monitoring creates numerous information gaps that otherwise may have been filled. It is critical that monitoring be included in local, regional, and national management efforts so that the results of those efforts can be determined.

KNOWLEDGE GAPS

Game Birds

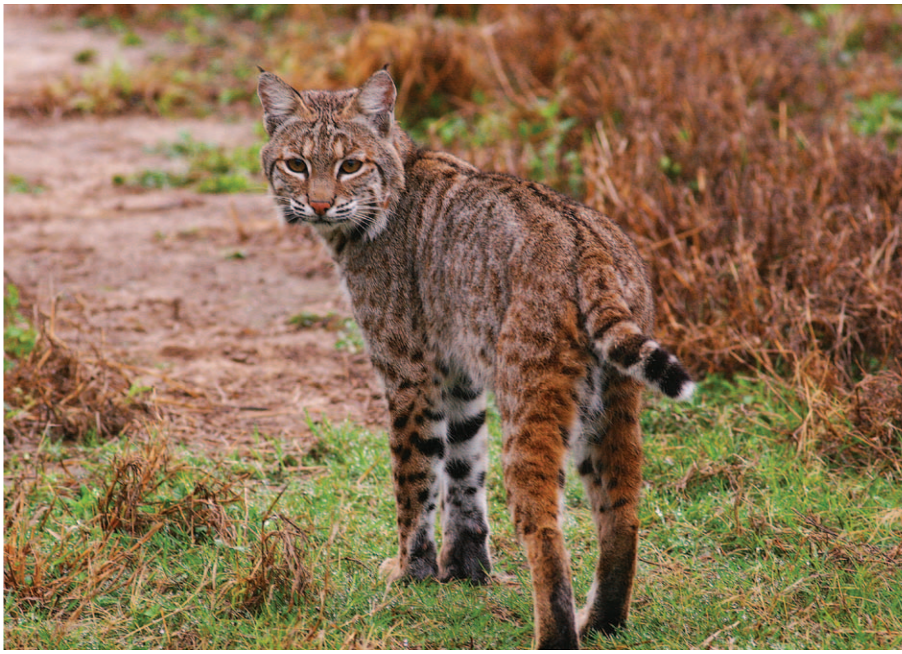
1. Experimental evidence of grazing practices beneficial to game birds is largely lacking. Before–after control–impact field experiments are needed to determine widespread, relative effects of grazing treatments and stocking intensities on nesting success and female and chick survival (Beck et al. 2000). Investigations also are needed to evaluate effects of grazing, use levels, and stocking rates on abundances of important forbs and insects in brood-rearing habitat because these responses are poorly understood. Experiments should be well replicated and of a sufficient time to understand short- and long-term effects on populations.
2. Similarly, investigations are needed to understand how to reduce and mitigate impacts of energy development and other significant sources of human disturbance over large landscapes as they relate to conservation practices. Recent studies show the large-scale and population-level impacts of oil and gas development on wildlife, including mule deer (Sawyer et al. 2009), sage-grouse (e.g., Walker et al. 2007), and songbirds (Ingelfinger and Anderson 2004). Wind energy will reduce our carbon footprint, but impacts to wildlife resulting from roads, noise, tall turbines, and additional power lines are poorly understood (e.g., lesser prairie-chicken; Pruett et al. 2009). These studies also will require strong statistical designs that include treatments and controls at spatial and temporal scales relevant to landscape-scale impacts (Johnson and St-Laurent 2011).
3. A multitude of local-scale questions should be addressed as part of larger investigations. For example, we should determine whether the addition of anthropogenic water sources benefits quail (and other wildlife) populations in the desert Southwest (Western Quail Management Plan 2008) and whether mortality from fence collisions places a role in population dynamics, and, if so, we should develop recommendations on type and placement of fencing to reduce mortality (Wolfe et al. 2007). Studies should be conducted long enough to capture the short- and long-term influences that impact the practice being examined.

Researchers should collaborate with management agencies to develop large and experimental projects as part of treatment projects planned by state and federal partners. In response, researchers and agencies can commit to monitoring at appropriate scales to evaluate treatment effects and to provide a basis for adaptive management.

Nongame Birds

As noted above, few experimental studies have specifically evaluated the use of rangeland management to benefit nongame birds. Most efforts have been from the midwestern United States in the series of studies conducted by Herkert et al. (1996, 2003). The degree to which their results apply to western ecosystems is unknown. To date, most studies conducted in the West have consisted of "fence-line" observational studies whereby investigators compare adjoining pastures with and without cattle grazing. Questions concerning grazing regime, timing (both longevity and season of grazing), stocking levels, and related variables have yet to be addressed. To do so will require well-designed, replicated studies that can determine various sources of variation to understand cause–effect relationships.

Bobcats (*Lynx rufus*) are a common rangeland predator that subsist primarily on rodents, rabbits, and birds. (Photo: Tim Fulbright)



Carnivores

Fruitful areas of research will include further evaluations of the role that top carnivores play in ecosystem structure and function (Hebbelwhite et al. 2005) and understanding the benefits or consequences of restoring those predators to historically occupied distributions. Additionally, better understanding of conditions that result in conflicts between humans and large carnivores (Wilson et al. 2006) may provide opportunities to lessen conflicts in the future. Continued efforts to improve methods of reducing human–carnivore impacts and the implementation of those methodologies on rangelands is desirable and necessary to conserve large carnivores (Shivik 2006). Further, responses of small carnivores to conservation practices should be explored more explicitly because of their importance as predators of ground-nesting birds; currently, much of the literature addresses risk of predation to avian species associated with rangeland management practices rather than demographic or habitat shifts in small carnivores that may result in those shifts in predation risk.

Ungulates

Much of the peer-reviewed literature documents the influence of livestock and wildlife on range flora, but the studies are usually not replicated, are conducted on a small scale, and do not indicate how associations with livestock influence productivity and recruitment of wildlife.

Additional research is needed to address these issues. In addition, because of the fragmentation of bighorn sheep habitat by livestock (Steinkamp 1990; Bissonette and Steinkamp 1996), social intolerance (Geist 1971), and disease transmission (Jessup 1985), most researchers argue that prescribed livestock grazing should not occur in bighorn sheep habitat. To minimize avoidance of livestock by bighorn sheep and, hence, avoidance of habitat, livestock and bighorn sheep should not be close to each other (Steinkamp 1990; Bissonette and Steinkamp 1996). When separation is not possible, efforts should be made to minimize contact (e.g., placement of anthropogenic water sources or fencing critical areas), monitor distribution, monitor range conditions, and carefully watch for incidences of disease outbreaks (Goodson 1982; McCullough et al. 1980; Technical Staff of the Desert Bighorn Sheep Council 1990).

Small Mammals, Reptiles, and Amphibians

Responses of small mammals, reptiles, and amphibians to grazing and other range management practices is species and often species-habitat specific. Few general trends have been identified, as studies have not been adequately designed to understand the underlying processes responsible because of the highly variable population dynamics of these groups of organisms and poor experimental designs (Johnson 1982). Experiments need to be of sufficient duration (perhaps on the order of decades in some ecosystems) and sufficient replication (over broad regional ranges) to isolate effects of interacting environmental factors that are usually not subject to experimental control from the effects of rangeland "treatments" (Rosenstock 1996). At least four avenues would assist in better data:

1. Experimental evidence of conservation practices beneficial to small mammals, reptiles, and amphibians is largely lacking. Experiments designed with pre- and posttreatment data and controls are needed to determine relative effects of treatments on abundance and reproductive success of wildlife species. Experiments must include regional replications and be of sufficient duration to account for the variable nature of small animal populations to enable managers to understand short- and long-term population effects attributable to conservation practices at regional levels.
2. Monitoring the distribution of various land uses in different landscapes (e.g., clumped or dispersed) and at what scale they occur are crucial for assessing long-term population persistence of small mammals, reptiles, and amphibians in fragmented landscapes.
3. When examining the effects of a management practice, comprehensive analyses, including the impacts of type, frequency, timing, and extent of disturbances (e.g., mowing, burning, or grazing) of vegetation, are necessary to understand the species and species-site-specific effects of such practices on species

abundance and reproductive success.

4. Researchers should collaborate with management agencies to develop large-scale, cost-effective experimental projects in an adaptive resource management strategy as part of conservation projects planned by state and federal partners. Commitments need to be made for monitoring at appropriate scales to evaluate treatment effects and to provide sound scientific data of sufficient scope and scale for assessing the true effects of conservation practices.

CONCLUSIONS

Very few of the 167 conservation practices listed by the NRCS have been evaluated in the peer-reviewed literature to determine their influence on upland wildlife. Activities associated with those conservation practices, particularly those efforts to enhance livestock production by limiting predation, have not been adequately investigated with respect to their overall impacts to rangeland ecosystems. Nevertheless, rangelands are important for protecting biodiversity, suggesting that future conservation efforts may require less reliance on reserves and a greater focus on private lands (Maestas et al. 2003). Grazing by livestock has received more attention in the literature than other conservation practices, but even then, studies often fail to distinguish between the different types, seasons, and intensities of grazing. Peer-reviewed literature evaluating how conservation practices influence upland wildlife habitat management has not received high priority, and their complex influences on wildlife and its habitat are largely unknown. Furthermore, other uses of rangelands (e.g., energy development) result in broad-scale loss and degradation of habitat that overwhelms other types of management (e.g., conservation practices) by increasing predation rates, promoting the spread of invasive plants, and facilitating disease transmission. However, the use of rangelands for sustainable livestock production has the potential to ensure the maintenance of wildlife habitat, especially when compared to energy development and urbanization, which will ensure that wildlife habitat will persist into the future.

Studies will need to be designed as targeted research, with adequate replicates and controls, for outcome-based science if managers and scientists are to better understand how NRCS conservation practices influence wildlife on western rangelands. Future studies should also follow rigorous before–after control–impact designs, be implemented at the landscape level, and be conducted for a sufficient amount of time to understand how NRCS conservation practices influence ecosystem dynamics.

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