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The Effects of Management Practices on Grassland Birds—An Introduction to North American Grasslands and the Practices Used to Manage Grasslands and Grassland Birds

Chapter A of
The Effects of Management Practices on Grassland Birds

Professional Paper 1842–A
Version 1.1, March 2022

U.S. Department of the Interior
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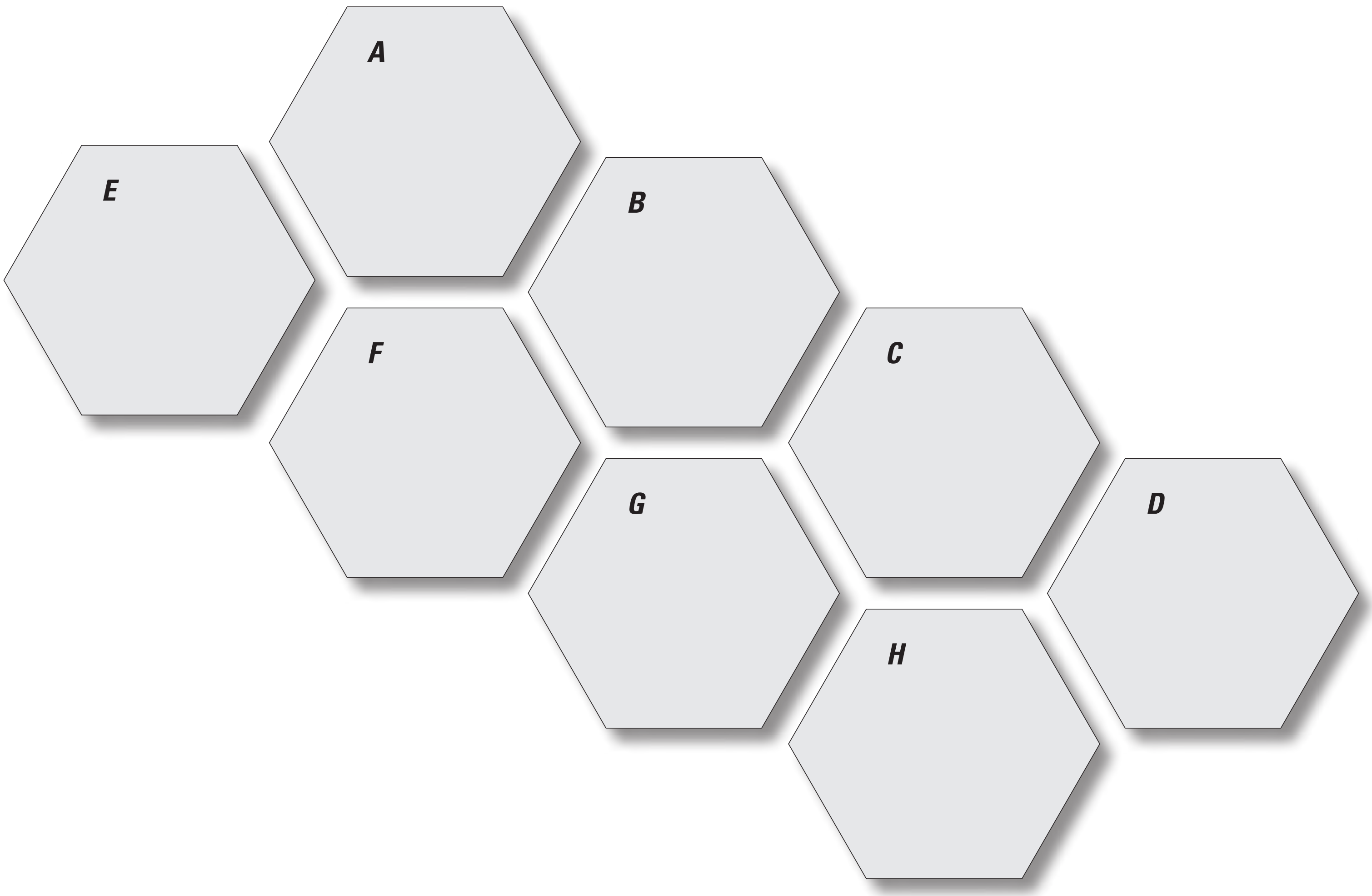

Cover. *A*, Upland Sandpiper, by Rick Bohn, used with permission. *B*, Burrowing Owl, by David O. Lambeth, used with permission. *C*, Greater Sage-Grouse, by Tom Koerner, U.S. Fish and Wildlife Service. *D*, Baird's Sparrow, by Rick Bohn, used with permission. *E*, Chestnut-collared Longspur, by Rick Bohn, used with permission. *F*, Cattle grazing in McPherson County, South Dakota, by Lawrence D. Igl, U.S. Geological Survey. *G*, Hay baling and raking, by Rick Bohn, used with permission. *H*, Prescribed burn, by Jennifer Jewett, U.S. Fish and Wildlife Service.
Background photograph: Northern mixed-grass prairie in North Dakota, by Rick Bohn, used with permission.

The Effects of Management Practices on Grassland Birds—An Introduction to North American Grasslands and the Practices Used to Manage Grasslands and Grassland Birds

By Jill A. Shaffer[1] and John P. DeLong[1,2]

Chapter A of

The Effects of Management Practices on Grassland Birds

Edited by Douglas H. Johnson,[1] Lawrence D. Igl,[1] Jill A. Shaffer,[1] and John P. DeLong[1,2]

[1]U.S. Geological Survey.

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Contents

Figure

Conversion Factors

U.S. customary units to International System of Units

Multiply	By	To obtain
	Length	
mile (mi)	1.609	kilometer (km)

International System of Units to U.S. customary units

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
	Area	
hectare (ha)	2.471	acre
square kilometer (km^2)	247.1	acre
hectare (ha)	0.003861	square mile (mi^2)
square kilometer (km^2)	0.3861	square mile (mi^2)
	Mass	
kilogram (kg)	2.205	pound (lb)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as

$$°F = (1.8 \times °C) + 32.$$

Abbreviations

CRP	Conservation Reserve Program
DNC	dense nesting cover
FWS	U.S. Fish and Wildlife Service
NGO	non-governmental organization
PCP	Permanent Cover Program
spp.	species (applies to two or more species within the genus)
ssp.	subspecies
USDA	U.S. Department of Agriculture
WPA	Waterfowl Production Area

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Northern mixed-grass prairie. Photograph by Rick Bohn, used with permission.

The Effects of Management Practices on Grassland Birds—An Introduction to North American Grasslands and the Practices Used to Manage Grasslands and Grassland Birds

By Jill A. Shaffer[1] and John P. DeLong[1,2]

North American Grassland and Wetland Habitats

The grasslands of North America can be divided into several major biogeographic regions, including the tallgrass, mixed-grass, and shortgrass prairies of the Great Plains; the desert grasslands of the southwestern United States and Mexico; the California grasslands; the Palouse prairie in the Intermountain Region (that is, the area between the Rocky Mountains and the Cascade and Sierra mountain ranges) of northwestern United States and British Columbia; the fescue prairie of northern Montana, southern Alberta, and central Saskatchewan; and the coastal grasslands of the Gulf Coast (Sims and Risser, 2000).

Characteristics of the North American Great Plains

The boundaries of the Great Plains have been described by numerous authors since the term was first popularized in the mid-1800s to describe the western plains of North America (Fenneman, 1931; Lewis, 1966). We adopt the definition of the term Great Plains, as defined by Lauenroth and others (1994), as the land mass that encompasses the entire central portion of the North American continent that was an unbroken expanse of primarily herbaceous vegetation at the time of European settlement and that extended from central Saskatchewan and Alberta to central Mexico and from Indiana to the Rocky Mountains (Clements, 1920; Weaver, 1954; Sims and Risser, 2000). The Great Plains was formed between 70 and 25 million years ago by the uplift of both the continental interior and the present-day Rocky Mountains, which displaced shallow seas, created a warmer climate, and deposited sediments that initiated soil building (Dix, 1964; Risser and others, 1981; Trimble, 1990). A renewal of the Rocky Mountain uplift during the Tertiary Period and glaciation events that occurred about 10,000 years ago in the northern Great Plains fostered the replacement of forests by herbaceous vegetation, to the extent of about 1.5 million square kilometers (km^2) (Weaver, 1954; Risser and others, 1981; Axelrod, 1985; Trimble, 1990; Samson and others, 1998). Periodic drought, recurrent fires, and extensive browsing and grazing by large mammals also played pivotal roles in determining the distribution of grasslands and forests prior to European settlement (Sauer, 1950; Axelrod, 1985).

The word *prairie* is often used to refer to the North American grasslands; its use is ascribed to French explorers of the 1680s to describe the tall grasslands west of the Mississippi River (Risser and others, 1981). The term is now broadly used to refer to any expanse of native grassland (Risser and others, 1981). Joern and Keeler (1995, p. 15) defined *prairie* as "grasslands maintained by naturally occurring forces representing years of interplay among countervailing pressures." People unfamiliar with the Great Plains often perceive this region as a homogeneous and monotonous landscape. Quite the opposite, the Great Plains harbors a diverse array of grassland, wetland, and woodland plant and animal communities that are uniquely adapted to the natural forces of the region. Despite local and regional differences, North American grasslands share the characteristics of a general uniformity in vegetation structure, dominance by grasses and forbs, a near absence of trees and shrubs (Weaver, 1954), annual precipitation ranging from 25 to 100 centimeters (cm), extreme intra-annual fluctuations in temperature and precipitation (Risser and others, 1981; Sims and Risser, 2000), and a flat to rolling topography over which fires can spread (Sauer, 1950). The dominance by grasses and forbs is, in part, a response to the high summer temperatures in the air and soil, soil moisture and precipitation that are not adequate to support tree growth, and groundwater sources beyond the reach of tree roots (Bailey, 1980). Classification of grasslands has been aided by readily identifiable climatic and soil features that help to distinguish vegetation types (Joern and Keeler, 1995).

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The simplest classification of grasses in the Great Plains places species into one of three broad categories based on the height attained at flowering (Weaver, 1954). Tallgrass species typically attain heights of 100–300 cm, mixed-grass species of 60–122 cm, and shortgrass species of 15–60 cm (Risser and others, 1981). Tallgrass species are most prevalent in the eastern prairies, although they may occupy moist lowlands and deep ravines elsewhere in the Great Plains (Weaver, 1954). Mixed-grass species predominate where the climate is drier, such as in the central Great Plains, or where rainfall is not supplemented by runoff, such as on slopes. Shortgrass species are more prevalent in very dry places, such as in the western Great Plains, or on hill crests and ridges where evapotranspiration is high owing to strong winds. Within the height classification of grasses, grass species also may be classified as cool season or warm season, depending on the timing of their emergence and growth; as sod forming or bunch forming, depending on their growth form; and as drought or grazing resistant, depending on their response to these disturbances.

The close relationship between grass height and precipitation nicely lends itself to another broad classification, which divides the Great Plains into tallgrass, mixed-grass, and shortgrass prairie types (Risser and others, 1981) (fig. A1; not all geographic places mentioned in report are shown on figure). The location of these prairie types generally follows an east-west gradient in declining precipitation. Precipitation in the tallgrass prairie region falls primarily during the spring and summer months and ranges from 64 to 102 cm annually (Bailey, 1980). Tallgrass prairie has the greatest plant species diversity of the three prairie types (Risser and others, 1981). Some of the dominant tallgrass species are big bluestem (*Andropogon gerardii*), Indiangrass (*Sorghastrum nutans*), switchgrass (*Panicum virgatum*), western wheatgrass (*Pascopyrum smithii*), rough dropseed (*Sporobolus clandestinus*), and green needlegrass (*Nassella viridula*) (Bailey, 1980; Risser and others, 1981; Steinauer and Collins, 1996; Samson and others, 1998); vernacular and scientific names of plants and animals follow the Integrated Taxonomic Information System (https://www.itis.gov).

Mixed-grass prairie contains plant species from both tallgrass and shortgrass prairie, with considerable intergrading of grassland types towards the peripheries (Risser and others, 1981; Samson and others, 1998). Precipitation falls primarily during the summer months, ranging from about 35–50 cm, with considerable variation depending on location (Joern and Keeler, 1995). Although mixed-grass prairie has few endemic plant species (Axelrod, 1985; Bragg and Steuter, 1996; Sims and Risser, 2000), distinct differences in species composition, plant community structure, and climate lend themselves to the subdivisions of northern mixed prairie, sandhills prairie, and southern mixed prairie (Risser and others, 1981; Bragg and Steuter, 1996). Plant communities of northern mixed prairie include the wheatgrass-bluestem-needlegrass (formerly *Agropyron* species [spp.], *Andropogon* spp., *Schizachyrium* spp., *Stipa* spp., *Hesperostipa* spp., *Nassella viridula*) and the wheatgrass-needlegrass associations of Küchler (1964;

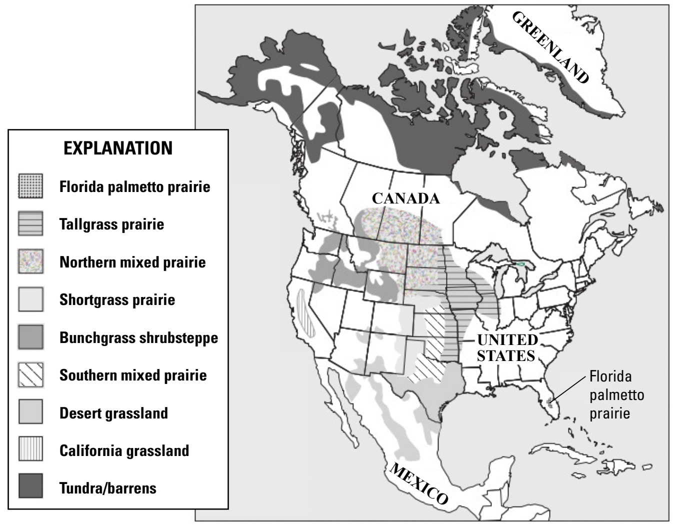
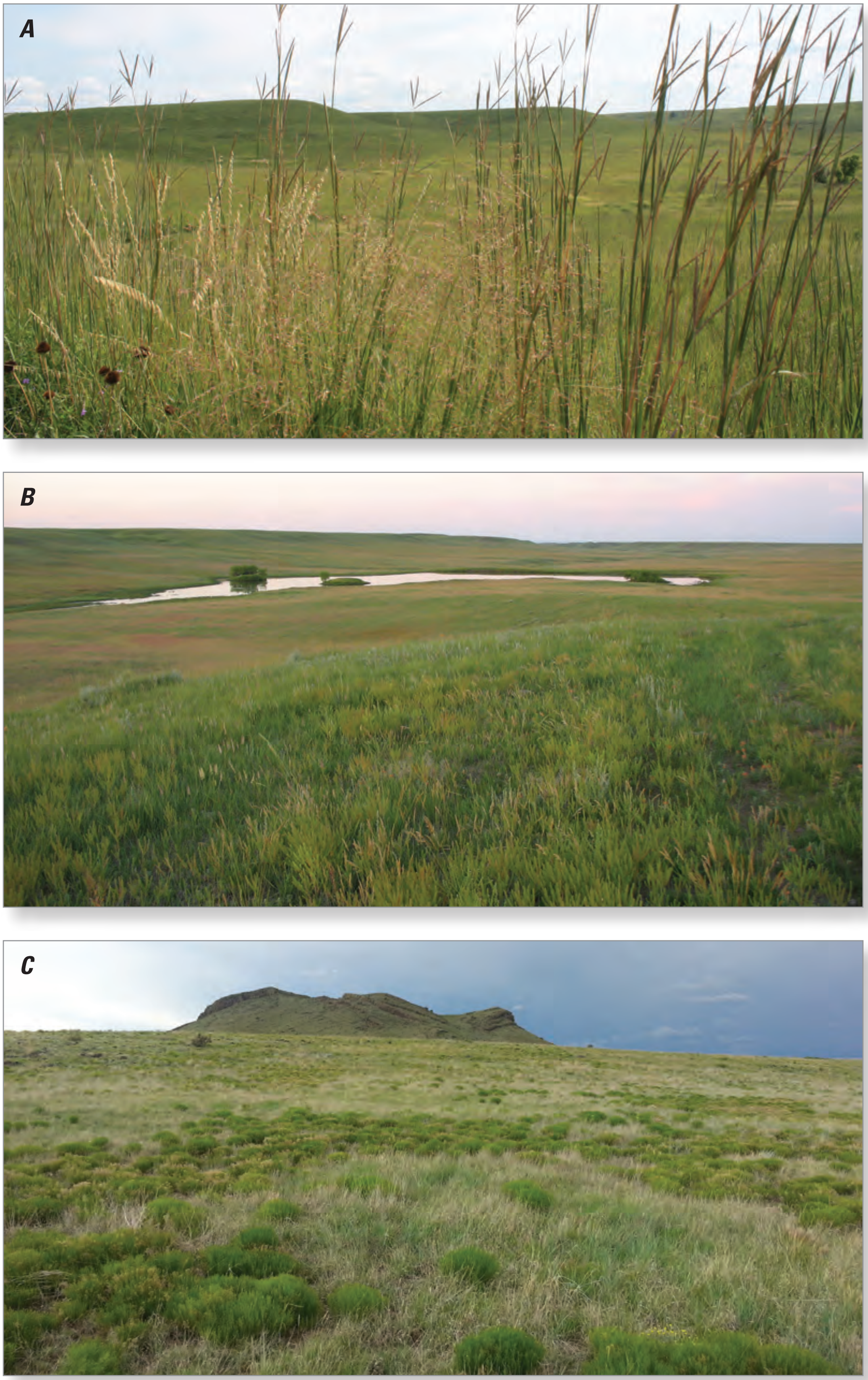


Figure A1. Distribution of major grassland ecosystems in North America prior to European settlement. Modified from Vickery and others (1999) and used with permission.

see also Risser and others, 1981; Bragg and Steuter, 1996). Common grass species of northern mixed prairie include blue grama (*Bouteloua gracilis*); buffalograss (*Bouteloua dactyloides*); and various wheatgrasses, needlegrasses, and fescues (*Festuca* spp.) (Bailey, 1980; Risser and others, 1981; Bragg and Steuter, 1996). Dominant grasses of sandhills prairie include prairie sandreed (*Calamovilfa longifolia*), sand bluestem (*Andropogon gerardii* spp. *hallii*), big bluestem, little bluestem (*Schizachyrium scoparium*), blue grama, hairy grama (*Bouteloua hirsuta*), needle and thread (*Hesperostipa comata*), and sand dropseed (*Sporobolus cryptandrus*) (Weaver, 1965). Southern mixed prairie includes the bluestem-grama (*Bouteloua* spp.) and mesquite-buffalograss (*Prosopis* spp.) associations of Küchler (1964; see also Bragg and Steuter, 1996).

Shortgrass prairie occurs primarily in the western Great Plains. Shortgrass prairie is dominated by blue grama and buffalograss, both of which are adapted to xeric conditions (Risser and others, 1981). Most precipitation in the shortgrass prairie falls during the summer. Annual precipitation ranges from 25 to 64 cm, and evapotranspiration usually exceeds precipitation (Bailey, 1980). Precipitation in this region is unpredictable, and the region often experiences periodic, sometimes severe, droughts (Knopf, 1988).

Various authors have described other divisions in vegetation within these three broad categories of prairie types in the Great Plains (Sims and Risser, 2000), including the prairie associations of Clements (1920), the vegetation associations of Küchler (1964), and the ecoregions of Bailey (1980), all of which are identified mainly by the dominant grass species and soil types. Ryan (1990) modeled the array of habitat types within a prairie ecosystem through the use of a "prairie continuum model," which uses gradients of soil moisture and fire and grazing frequency and intensity to portray grassland habitats along a two-dimensional continuum. This continuum



A, Tallgrass prairie at Konza Prairie Biological Station, Flint Hills, Kansas; photograph by Jill Haukos, Kansas State University, used with permission. *B*, Mixed-grass prairie in Valley County, Montana; photograph by Melissa Wolfe Welsch, U.S. Geological Survey. *C*, Shortgrass prairie at Two Buttes, Colorado; photograph by Dale W. Stahlecker, used with permission.

can be used on a large geographic scale to describe regional variation in shortgrass prairie, or at smaller scales to describe differences in habitats between dry ridgetops and wet valleys.

Wetlands are integral to the Great Plains landscape. The Great Plains are home to five major wetland regions: Prairie Pothole, Nebraska Sandhills, Rainwater Basin, Cheyenne Bottoms, and Playa Lakes (Batt, 1996). Each wetland region has had a unique hydrological evolution that occurred during the Pleistocene (Batt, 1996; Samson and others, 1998). The wetlands within each region play critical roles in the structure and functioning of the upland prairie community through flood attenuation, nutrient storage, groundwater storage and recharge, and provisioning of wildlife habitat (Johnson and others, 1997; Knutsen and Euliss, 2001; Euliss and others, 2004). Small wetlands provide important habitat for many species of prairie fauna because the wetlands produce an abundant source of aquatic insects and other invertebrates (Kantrud and Stewart, 1984; Johnson and others, 1997; Larson and others, 1998).

Wetlands and mixed-grass prairie in the South Dakota portion of the Prairie Pothole Region of North America; photograph by U.S. Fish and Wildlife Service.

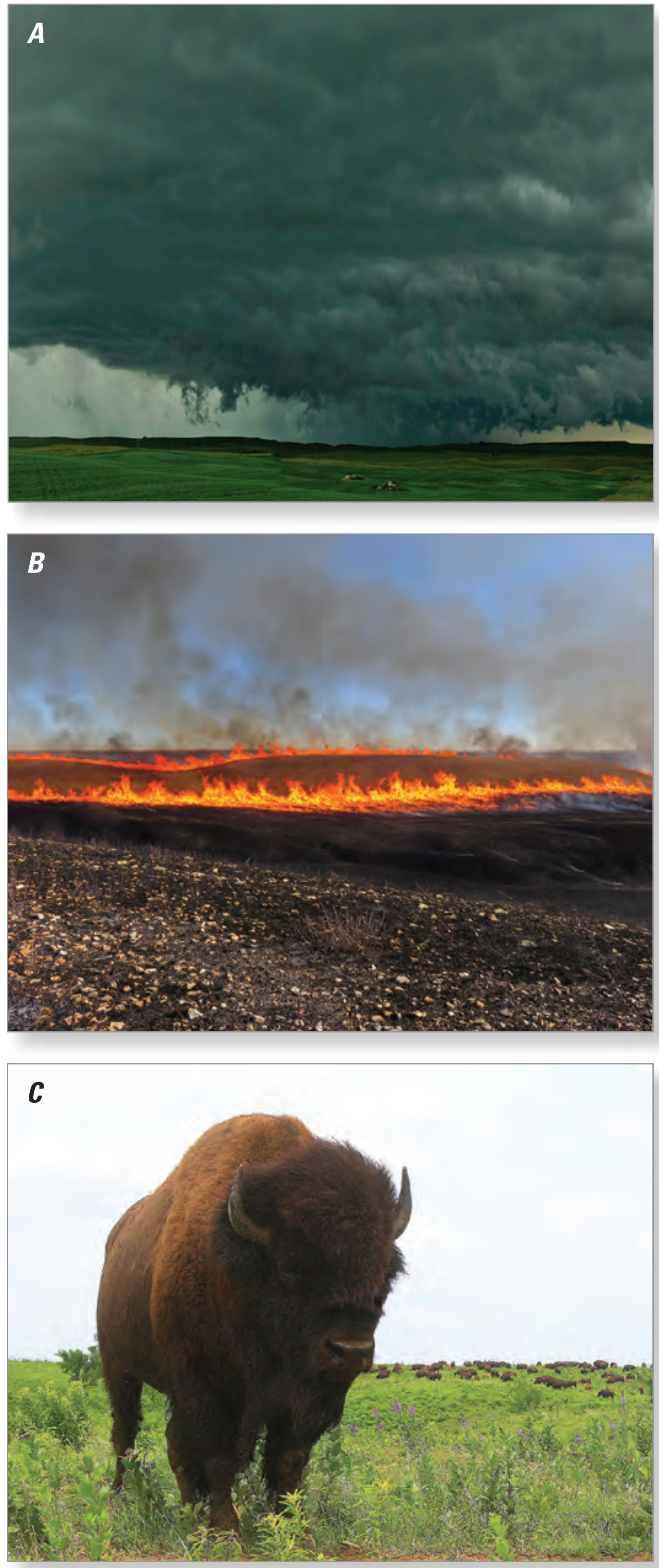
Woodlands and shrub-dominated habitats persist in the Great Plains in areas that were protected from fire, such as on buttes and in riparian areas, on river bluffs, along slopes of hills, and in isolated thickets within grasslands (Stewart, 1975; Bragg and Steuter, 1996). Prairie-forest ecotones occur at the periphery of the Great Plains where grassland habitats transition into forest or shrubland communities. In the northern Great Plains, prairie parkland forms a transitional habitat between grasslands and northern peatlands of the boreal forest (McNicholl, 1988; Chapman and others, 1998). In prairie parklands, stands of aspen (*Populus* spp.) are intermixed in grasslands. Oak (*Quercus* spp.) savannas are transitional habitats that occur between eastern oak forests and prairies and are characterized by a grassy understory and scattered oaks (Henderson and Epstein, 1995; McPherson, 1997). Canopy coverage in oak savannas varies considerably, and savanna types vary regionally and by soil type. Juniper (*Juniperus* spp.) savanna is a similar type of habitat, transitioning between the prairie and the coniferous woodlands of higher-elevation areas in the West. Shrubsteppe habitats occur in the western Great Plains grasslands and are dominated by sagebrush (*Artemisia* spp.) and grasses (Paige and Ritter, 1999). Shrubsteppe habitats vary from dry shrublands with sparse grass cover to patchy mixes of shrubs and grasses.

A, Wooded riparian area in Dickey County, North Dakota; photograph by Jill A. Shaffer, U.S. Geological Survey.
B, Oak savanna in the Sheyenne National Grassland, Richland County, North Dakota; photograph by Catherine Pohl, Vermont Institute of Natural Science, used with permission.

Major Ecological Forces in the Great Plains prior to European Settlement

Grassland plant communities of the Great Plains were formed and are maintained by the interactive forces of climate, fire, and grazing, and are influenced by soil type (Risser and others, 1981). These natural forces created a diversity that sometimes displays itself in obvious contrasts, such as those among tallgrass prairie in the northern Great Plains, sandhill prairie of Nebraska, and shortgrass prairie of the western Great Plains (Bragg, 1995). Other differences are more subtle, such as the intergradations between prairie types or between north- and south-facing slopes. Differences, both obvious and subtle, arise from interactions between the abiotic components of the environment, namely climate and soils, and the biotic components. Fire and grazing pressure also exert an influence. Within grasslands more so than other biomes, organisms are exposed to extremes of temperature, humidity, wind, and precipitation, as well as to daily, seasonal, and long-term variation in climatic factors on local and regional scales (Risser and others, 1981).

Geological processes and their effect on regional and continental air masses have a profound influence on climate in central North American grasslands. The uplift of the Rocky Mountains during the Tertiary Period created a subhumid climate in the interior of North America (that is, a climate in which evapotranspiration and precipitation are nearly equal on an annual basis; Bailey, 1980). Pacific, polar, and tropical air masses interact in the Great Plains to create east-west and north-south gradients of temperature and moisture, which in turn affect the development of prairie types across the region (Samson and others, 1998; Sims and Risser, 2000). As moist-air masses from the Pacific Ocean pass over the coastal mountain ranges and the Rocky Mountains, the air masses drop precipitation west of the mountains, causing a rain shadow effect that results in relatively little precipitation falling over the Great Plains, especially in the shortgrass prairie of the western plains (Weaver, 1954; Dix, 1964; Bragg, 1995). Air masses from the Gulf of Mexico move northward and spread high humidity and precipitation over the mixed-grass prairie of the central Great Plains and especially the tallgrass prairie of the eastern Great Plains (Risser and others, 1981; Bragg, 1995; Samson and others, 1998). Thus, from west to east, the amount of precipitation increases and the frequency of drought decreases (Sims and Risser, 2000). Most precipitation occurs during the growing season. Eastern grasslands receive much more precipitation (102–152 cm) than grasslands in the Intermountain Region or just east of the Rocky Mountains (25–38 cm) (Joern and Keeler, 1995). From south to north, a greater proportion of annual precipitation occurs as snow, the growing season becomes shorter, and average temperatures decrease (Sims and Risser, 2000). Over time, these gradients have strongly influenced the evolution of species and the species composition and distribution of grassland communities (Steinauer and Collins, 1996; Weaver and others, 1996). Climatic variability also was an important factor in the



Climate, fire, and grazing are natural forces that shaped the Great Plains. *A*, Storm gathering over the prairie; photograph by Rick Bohn, used with permission. *B*, Fire, and *C*, American bison (*Bison bison*); photographs by Jill Haukos, Kansas State University, used with permission.

evolution of species and grassland communities. For example, drought and flooding have been major ecological forces in the evolution of grassland biota (Bragg, 1995; Samson and others, 1998). These wet and dry cycles may occur over short and long time scales, and grassland species have adapted to these fluctuations (McNicholl, 1988).

As with climate, soil characteristics vary across grasslands and reflect differences in precipitation and other climatic factors, as well as in parent materials, biological activity, and topography (Kantrud and Kologiski, 1982; Brady, 1990; Samson and others, 1998). Prairie soils, or mollisols, have black, friable, organic surface horizons (Bailey, 1980). Grass roots penetrate deeply into mollisols, bringing chemical bases to the surface and creating fertile soils. Thus, mollisols are one of the most productive soil groups. Because grasslands typically receive less precipitation than do forests, grasslands experience less soil leaching. Therefore, calcification, or accumulation of carbonates in the lower layers, is the primary pedogenic process. Salinization occurs on poorly drained soils. Soils of the semidesert shrub, the aridisols, have little organic matter, clay horizons in some places, and accumulations of various salts.

Soils of the Great Plains are derived from parent materials deposited from seas during the Cretaceous Period; from the processes of erosion, deposition, and mountain building during the Tertiary Period; and from glaciation during the Pleistocene (Bragg, 1995). Glacial deposits and outwash sands and gravels are the primary parent materials east and north of the Missouri River, whereas soils derived from sandstone and shale are present south and west of the Missouri River (Sims and Risser, 2000). The central Great Plains contain loess and eolian sand deposits, and soils are deep, loamy sediments of loess, eolian sand, alluvium, and outwash. In the Texas Panhandle area, fine-textured soils were deposited.

Each grassland type in the Great Plains supports vegetation that is compositionally and structurally heterogeneous. Fuhlendorf and Engle (2001) expanded on the term *heterogeneous* to denote variability not only in vegetation stature and composition but in vegetation density and biomass as well. Before European settlement, species diversity in grasslands was maintained by climate, fire, and by grazing pressures at intensities and frequencies that varied by grassland type, creating shifting mosaics (Saab and others, 1995; Vickery and others, 2000; Johnsgard, 2001). Tallgrass prairies were maintained primarily by fire, whereas shortgrass prairies were maintained primarily by drought and grazing (Gibson and Hulbert, 1987; Collins, 1992; Vickery and others, 2000).

Historically, causes of fires were natural and anthropogenic (that is, those started by Native Americans) and were an important factor in maintaining native grasslands (Sauer, 1950; Axelrod, 1985; Bragg, 1995; Samson and others, 1998). Without fire, grasslands undergo succession to shrublands or forests (Sauer, 1950). A number of factors or conditions, acting individually or in concert, might influence the response of a particular grassland to a particular fire (Bragg, 1995). Important variables include fire frequency or interval (number of years between burns); season of burn; burn intensity; flammability of vegetation; and whether fires are headfires or backfires, which influences the speed and intensity of the fire. Flammability hinges upon biomass accumulation and dryness of plants, which is dependent on fire history, grazing pattern and intensity, moisture available to plants, season, and weather conditions. Fires set by native hunter-gatherers differed from fires set by lightning in terms of seasonality, frequency, and intensity (Lewis, 1985). Lightning typically caused infrequent, high-intensity fires, whereas Native Americans set frequent

Soil profile of a prairie mollisol showing the thick, dark, humus-rich upper soil layer with an intervening albic layer; photograph by U.S. Department of Agriculture Natural Resources Conservation Service.

but low-intensity fires (Kay, 1998). Thus, anthropogenic fires and lightning fires resulted in different vegetation mosaics, and in some cases, different plant communities (Blackburn and Anderson, 1993).

The grasslands of the Great Plains evolved under the influence of grazing pressure over millions of years. The current vegetation composition and physiognomy of grasslands and the ability to withstand grazing were shaped by selection pressures during the Pleistocene (Milchunas and others, 1988). The effect of the Pleistocene megafauna (mainly mammoths [*Mammuthus primigenius*], camels [*Camelus* spp.], bison [*Bison* spp.], and horses [*Equus caballus*]) on the evolution and coevolution of native flora and fauna in grasslands likely was immense but remains virtually unknown. Between 12,000 and 10,000 years ago, the Pleistocene megafauna had largely gone extinct, with the bison emerging as one of the few large herbivores to survive extinction. At the time of European settlement, important native herbivores in North American grasslands included American bison (*B. bison*), elk (*Cervus elaphus canadensis*), deer (*Odocoileus* spp.), pronghorn (*Antilocapra americana*), prairie dogs (*Cynomys* spp.), pocket gophers (*Geomyidae* spp.), and Rocky Mountain grasshopper (*Melanoplus spretus*) (Steinauer and Collins, 1996; Knapp and others, 1999; Vickery and others, 1999; Lockwood, 2004). Historically, unrestricted animal movements and a diverse herbivore community helped to maintain heterogeneity (for example, variability in vegetation stature, composition, density, and biomass) in vegetation structure (Bock and others, 1993; Steinauer and Collins, 1996; Fuhlendorf and Engle, 2001). Large herbivores selected plant species based on seasonal dietary requirements and forage quality (Steinauer and Collins, 1996). Bison were nomadic, moving in large herds in response to vegetation changes associated with precipitation and fire (Samson and others, 2004). Bison often did not return to previously grazed areas for 1–8 years, providing a natural rest interval that resulted in vegetation heterogeneity. Unlike bison, which roamed widely, the influence of prairie dogs was more localized. As many as 5 billion prairie dogs may have populated the Great Plains prior to European settlement (Samson and others, 1998; Johnsgard, 2005). Selective grazing of grasses by prairie dogs created large swaths of tender, green grasses, microhabitats for a diversity of plant and arthropod species, and improved soil fertility and nutrient cycling (Johnsgard, 2005). Prairie dog colonies were thus attractive to bison and other herbivores. The vegetative diversity, altered soil structure from burrowing activities, and rich prey base provided by the prairie dogs themselves provided resources for more than 100 species of vertebrates (Jones and Cushman, 2004). Rocky Mountain grasshoppers were irruptive and had major effects on vegetation in the Great Plains in some years (Lockwood, 2004).

In pre-modern times, fire intensity and coverage were influenced by ungulate grazing pressure, which in turn was influenced by the degree to which ungulates were hunted by Native Americans (Kay, 1998). Historical accounts of prairie fires that raged for days indicate that moderate numbers of ungulates roamed the prairie prior to European settlement, because heavy grazing by large numbers of ungulates would have slowed the spread and growth of large fires. In areas of high ungulate populations, standing plant biomass and litter accumulation were reduced by grazing, creating patches where fuel loads were insufficient to sustain fires. These remaining unburned patches then attracted grazers immediately after a fire. Once regrowth occurred on the burned sites, grazing was concentrated in burned patches because of the nutritive value of the plants that emerged after a burn (Risser, 1990; Fuhlendorf and Engle, 2001). Because grazing then shifted from unburned areas to burned areas, the unburned areas accumulated fuel loads capable of supporting fire. Overall, then, the interplay between the effects of Native Americans on the ungulate populations may have shifted the fire pattern from one of infrequent, high-intensity, naturally caused fires to one of frequent, low-intensity fires (Kay, 1998).

North American Grassland and Wetland Habitats after European Settlement

Anthropogenic Changes to the Major Ecological Forces of Grazing and Burning

The arrival of European settlers to North America brought profound change, including the establishment of permanent towns and cities, the proliferation of cropland-based agricultural systems, and the suppression of wildfires. Settlement of the Great Plains in the United States increased with the Homestead Act of 1862. The near extirpation of bison by the 1860s paved the way for dramatic changes in the dominant grazers on the Great Plains and a shift in the disturbance patterns that historically influenced the vegetation structure of grasslands. The bison population, which once numbered in tens of millions, dwindled to a few hundred individuals (Hornaday, 1889; Roe, 1951; Sandoz, 1954; Knopf, 1994). Native Americans were displaced from traditional hunting grounds and concentrated into reservations. By 1890, the number of cattle and sheep on the western range were estimated at 45 and 50 million, respectively (Fedkiw, 1989). Originally, free-ranging cattle grazed over wide areas on the open range. In the 1880s, the cattle industry experienced a fundamental shift in operations. In response to the difficulties of keeping livestock alive during harsh winters, cattle in many areas of the Great Plains and western rangelands were restricted to fenced pastures, where it was easier to provide supplemental feed during the winter.

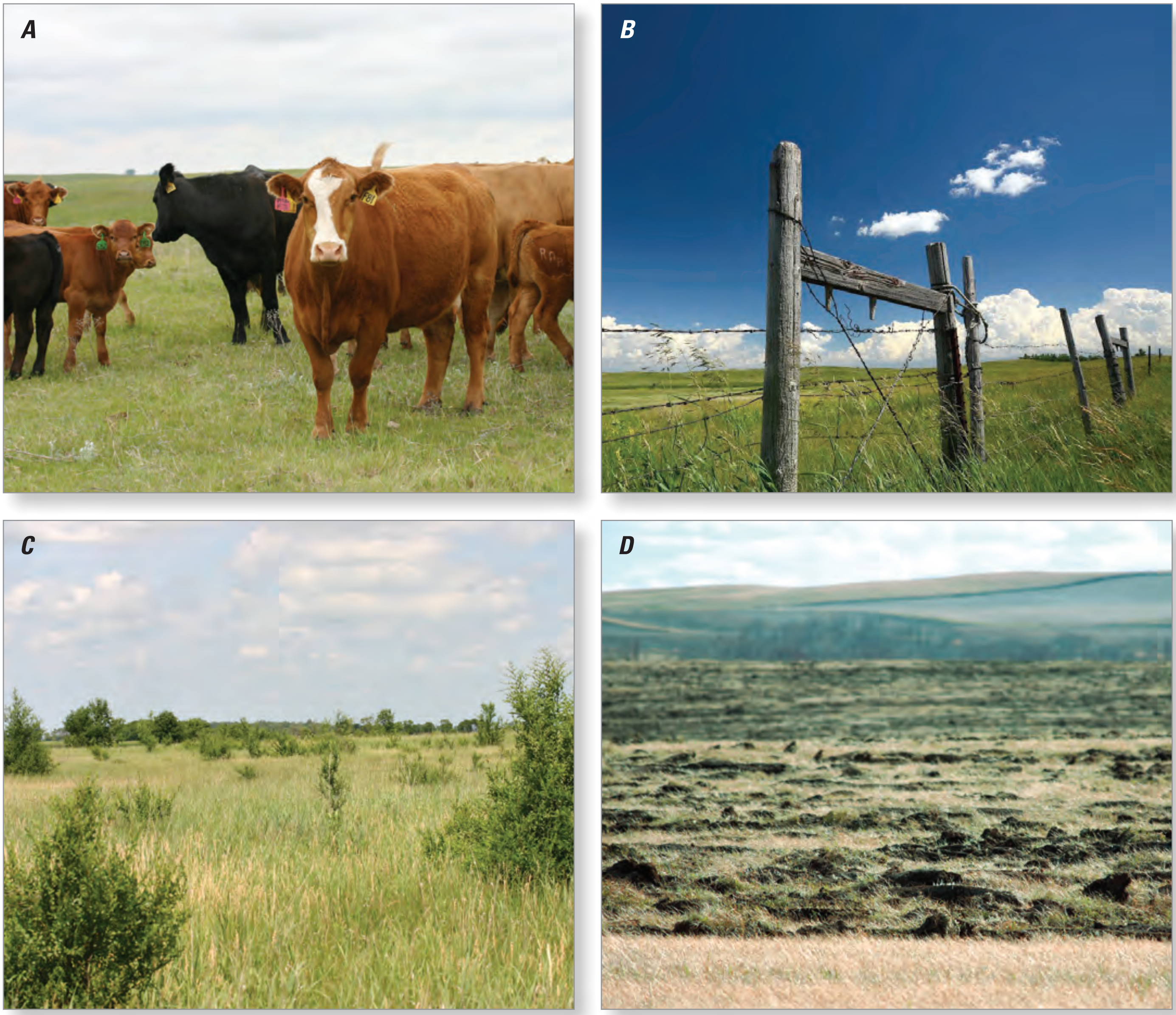
Compared to bison, domestic cattle and other livestock have different foraging patterns and behaviors, forage preferences, and effects on grassland vegetation (Johnsgard, 2001). Historically, American bison were migratory, moving through areas in large herds and remaining in areas until their preferred forage was gone; in contrast, domestic cattle typically are confined to fenced areas and continue to forage in the same area for longer periods. Different species of grazers vary in their preference of palatable plants, thus creating different impacts on plant composition (Peden and others, 1974; Schwartz and Ellis, 1981). For example, bison may eat about 90 percent graminoids and 10 percent forbs and browse, whereas cattle may eat about 75 percent graminoids and 25 percent forbs and browse, which can lead to a change in the diversity and abundance of remaining vegetation (Plumb and Dodd, 1993). Rangeland practices that have directly or indirectly promoted the growth or dominance of some plant species that are more palatable to domestic livestock may have caused a decline in the less-palatable species as well as a decline in biological diversity (Fuhlendorf and Engle, 2001). Alternatively, because domestic livestock typically graze particular patches of grassland for longer durations than bison did, livestock grazing may lead to elimination of plants that are highly palatable to domestic livestock, as well to soil compaction (Weaver, 1968; Johnsgard, 2001).

The area of rangeland in North America has been steadily declining. In the five States (that is, North Dakota, South Dakota, Nebraska, Minnesota, and Iowa) constituting the western Corn Belt, Wright and Wimberly (2013) estimated a net decline in grass-dominated land cover of 530,000 hectares (ha) from 2006 to 2011. Prior to this, from 1977 to 1997, 1.4 million ha of rangeland in South Dakota alone were converted to cropland and other developments (Higgins and others, 2002). Further exacerbating the degradation of grasslands has been the increased grazing intensity exerted on remaining grasslands. In recent decades, heightened consumer demand for beef and subsequent opportunity for greater profits has encouraged the livestock industry to produce heavier cattle in larger herds that are foraging over smaller areas (Higgins and others, 2002). In South Dakota, average slaughter weight of cattle increased from 427 kilograms (kg) in 1940 to 622 kg in 1999. During the same period, the number of cattle in the State increased from 1,632,000 to 3,850,000 (Higgins and others, 2002).

The practice of restricting livestock movements by constraining them to fenced pastures has reduced variation in grazing pressure across the Great Plains (Knopf, 1993). Fencing of pastures is a tool used by many land managers, including Federal agencies, to achieve standardized vegetative goals, but the practice may decrease biological diversity and viability (Samson and others, 2004). As Fuhlendorf and Engle (2001, p. 625) explained, “Most techniques of rangeland management were developed under the paradigm of increasing and sustaining livestock production by decreasing the inherent variability associated with rangelands and grazing.” Traditional rangeland management techniques have promoted the dominance of those few plant species that are most productive and most palatable to domestic livestock. Fuhlendorf and Engle (2001) advocated a new rangeland management paradigm that focuses not only on livestock production but also on biological diversity. That approach is based on focal patches that receive fire and grazing disturbances that change through time, creating shifting mosaics of burned and grazed patches.

The near extermination of bison in North America was followed by an eradication effort of another major herbivore, the prairie dog (Knopf, 1994). Prairie dog numbers have declined by about 98 percent since European settlement, primarily owing to eradication measures intended to reduce presumed competition for forage with domestic livestock or to prevent damage to nearby agricultural crops (Summers and Linder, 1978; Marsh, 1984; Miller and others, 1994). The grazing and fossorial activities of prairie dogs have played an important role in the maintenance and composition of grassland plants and animals. For example, prairie dog colonies may increase forb and shrub coverage and decrease grass coverage compared with noncolony areas (Coppock and others, 1983; Fahnestock and others, 2003). In addition, prairie dogs play an important role in nutrient cycling and soil formation in grasslands (Coppock and others, 1983; Samson and Knopf, 1994).

Fire frequency or suppression may substantially influence biodiversity in grasslands. Historically, fire frequency estimates on native prairie ranged from nearly every year in tallgrass prairies to every 3–5 years in mixed-grass prairies (Samson and others, 2004). Suppression of wildfires and the near-total loss of fire as a natural disturbance agent have dramatically changed vegetation patterns on the Great Plains. Prior to settlement of the Great Plains, woodlands largely were restricted to riparian areas, ravines, and canyons, where conditions hampered fire frequency and intensity (Anderson, 1982; Grant and others, 2004a; Grant and Murphy, 2005). Reduced fire frequency and the extirpation of bison contributed to the spread of juniper, aspen, and other woody vegetation into grassland areas in the prairie parklands and prairies of the Great Plains (McNicholl, 1988; Coppedge and others, 2001; Grant and Murphy, 2005). Changes in the timing, intensity, size, or frequency of fire and other disturbances may have profound influences on grasslands. For example, long-term idling or periods without fire may facilitate encroachment of trees and shrubs and thereby the conversion of grasslands to woodlands or shrublands (Hobbs and Huenneke, 1992; Vickery and others, 1999, 2000; Grant and others, 2004a). However, too-frequent burning also can result in a change in species composition and loss of biodiversity (Fuhlendorf and Engle, 2001; Powell, 2006). In the Flint Hills of Kansas, annually burned grasslands exhibited lower plant species diversity than did unburned grasslands or grasslands burned every 4 years (Collins, 1992). A grassland community’s response to burning may depend on community composition and productivity, evolutionary history, and the type and frequency of disturbance. Historically, different grasslands evolved under different disturbance regimes. A change in the disturbance regime can profoundly influence the vegetation within those grasslands. In Arizona, for example, the shift from fire to grazing as the dominant tool for maintaining shortgrass prairies altered plant species composition and canopy coverage of the area (Bock and Bock, 1993). Grazing reduced grass coverage and changed grass species composition, which in turn altered fire regimes.



Major changes to the native prairie ecosystem wrought by the arrival of Europeans to North America included the near-extirpation of American bison (*Bison bison*) and their replacement with domestic cattle (*A,* photograph by Lawrence D. Igl, U.S. Geological Survey), which precipitated the fencing of the Great Plains (*B*, photograph by Rick Bohn, used with permission), suppression of fire which led to woody encroachment (*C*, photograph by Lawrence D. Igl, U.S. Geological Survey), and the breaking of prairie sod for cropland agriculture (*D*, photograph by Krista Lundgren, U.S. Fish and Wildlife Service), which continues with ever more intense agricultural practices in modern times (*E*, photograph by Krista Lundgren, U.S. Fish and Wildlife Service).

Factors Contributing to the Loss and Degradation of Grassland and Wetland Habitats

The two major threats to grassland habitats are grassland loss and degradation in the quality of those grasslands that remain. These factors mirror the greatest threats to biodiversity worldwide (Vitousek and others, 1997). The two biomes at greatest risk of extensive habitat loss and underprotection are temperate grasslands and savannas; in these biomes, the extent of habitat conversion exceeds that of habitat protection by a factor greater than eight (Hoekstra and others, 2005).

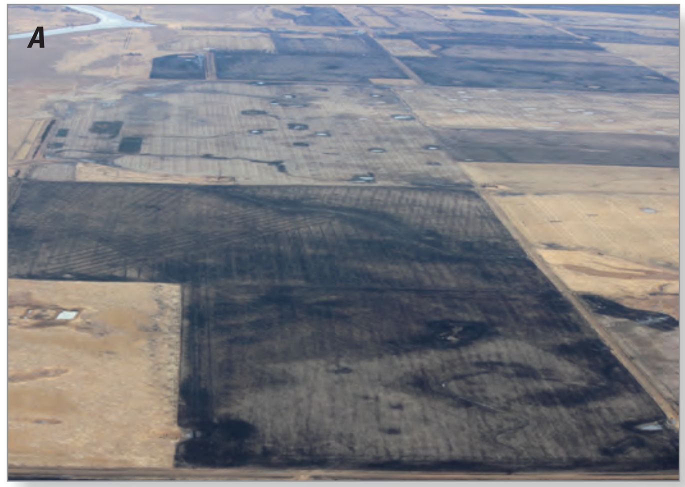
Historically, agricultural practices have been the greatest causes of grassland and wetland loss in North America (Knopf, 1994; Dahl, 2011). Urban development and sprawl in exurban areas have caused further loss, fragmentation, and isolation (Blair, 1996; Marzluff and Ewing, 2001; Dahl, 2014). The increase of cropland agriculture led to the widespread loss of native grasslands in North America, which continues into the present (Knopf, 1988; Noss and others, 1995; Stephens and others, 2008; Rashford and others, 2011a, 2011b; Wright and Wimberly, 2013; Lark and others, 2015). In Canada, about 70–75 percent of native prairie has been converted to non-native cover (Gauthier and Wiken, 2003).

Of the three main types of native prairie in the Great Plains, tallgrass prairie has suffered the most severe loss: less than 5 percent of original tallgrass prairie remains (Samson and others, 2004). Losses of tallgrass prairie in individual States or Provinces range from 82.6 to 99.9 percent (Samson and others, 1998). Loss of mixed-grass prairie ranges from 30 percent to more than 99 percent, and loss of shortgrass prairie ranges from 20 to 86 percent (Samson and Knopf, 1994; Samson and others, 1998, 2004). Most remaining native grasslands are managed as rangeland for domestic livestock. The management priority on these private rangelands is usually that of increasing livestock production rather than protecting biological diversity or ecosystem functions (Fuhlendorf and Engle, 2001; Derner and others, 2009).

Agricultural-induced losses have occurred in all three major grassland types of the Great Plains, with losses increasing from west to east. Areas previously dominated by small-grain production and conservation grasslands and thought to be unsuitable for cropland are now being reevaluated as potential areas to plant annual crops (Mushet and others, 2014). Lark and others (2015) estimated that more than 2.3 million ha of native and planted grasslands were converted to cropland from 2008 to 2012, with around 647,000 ha of that being grasslands with a high likelihood of not having been planted, plowed, or hayed for at least 20 years. Lark and others (2015) further estimated that the cultivation of corn (*Zea mays*) and soybeans (*Glycine max*) reached record high levels following the biofuels boom of the 2000s. In South Dakota, as in other parts of the United States, the recent development of drought-resistant, genetically modified soybeans has accelerated the conversion of native grasslands to cropland in areas once too dry to grow soybeans (Higgins and others, 2002). Similarly, new corn varieties have been developed that are drought resistant, cold tolerant, and pesticide tolerant and that mature earlier than existing varieties; these new varieties have allowed the geographic range of corn to expand westward and northward into the mixed-grass prairies of North America, threatening remaining grasslands and wetlands (Ringelman, 2007). Recent grassland losses have been attributed to economic and political forces that have stimulated increased planting of corn for the production of ethanol (Kriz, 2007; Ringelman, 2007). The popularity of the herbicide glyphosate also has hastened conversion of grasslands. Transgenic crop plants that are genetically designed to resist glyphosate do not succumb to the herbicide, whereas glyphosate is lethal to nontransgenic plants (Service, 2007). Glyphosate-resistant crops allow farmers to drill crop seeds directly into native prairie, wait until the crop has emerged, and then apply glyphosate to kill all species but the crop species, without the need for plowing.

As with grasslands, oak savannas and wetlands have been altered by agricultural operations. Oak savannas also are subject to tree removal operations and may undergo succession to woodland habitats when fire-return intervals are altered owing to human activities; less than 1 percent of the historical extent of oak savannas remains (Nuzzo, 1986; Henderson and Epstein, 1995; Noss and others, 1995; McPherson, 1997). Most of the remaining oak savannas in North America occur in isolated small patches (McPherson, 1997). As for wetlands, Dahl (1990) estimated that the continental United States contained 89 million ha of wetlands in the 1780s but lost 53 percent of them within the past 200 years. Most loss is attributed to agricultural conversion, with 22 States having lost 50 percent or more of their original wetlands (Dahl, 1990). At the time of Dahl's (1990) writing, he estimated that the continental United States lost more than 24 ha of wetlands for every hour between the 1780s and the 1980s. Within the Prairie Pothole Region of Montana, North Dakota, South Dakota, Minnesota, and Iowa, Dahl (2014) estimated that about 65 percent of the 17 million wetlands on the landscape around 1850 had been drained by the mid-1980s.

The trend of wetland loss since European settlement (Dahl, 1990) continues in the Great Plains (Knutsen and Euliss, 2001; Johnston, 2013; U.S. Fish and Wildlife Service [FWS], 2017). Dahl (2014) estimated that emergent and farmed wetlands in the Prairie Pothole Region declined by 38,600 ha between 1997 and 2009. More than one-half of the emergent wetlands that are drained are small (average size of 0.4 ha) (Dahl, 2006), but these wetlands are invaluable as wildlife habitat (Reynolds and others, 2006). Wetlands have been drained for many reasons, but especially to facilitate cultivation and development of human settlements (Dahl, 2011). Both cultivation and human settlements affect the integrity of the prairie ecosystem by altering the hydrology, groundwater, and floral and faunal relationships between the grassland and wetland areas (McNicholl, 1988; Batt, 1996; Gleason and others, 2008). Agriculture is the largest source of wetland loss, because the demand for corn ethanol, expiration of agricultural conservation programs, and commodity prices have all increased demand for arable land (Johnston, 2013). Owing to Federal legislation, very few private wetlands in the Prairie Pothole Region are conferred Federal protection under either the Clean Water Act or the wetland conservation (or Swampbuster) provision of Farm Bill legislation (Dahl, 2014). A landowner's perception of wetlands and their value is strongly influenced by the landscape context within which wetlands are located (Higgins and others, 2002). Wetlands within a native prairie landscape provide water and forage not only to wildlife but also to livestock, and so are at low risk of drainage. Wetlands within a cropland matrix, however, are more likely to be drained by farmers who tire of farming around them. As new advances in biotechnology and economic forces entice farmers to till native and conservation grasslands, existing wetlands will be subjected to increased

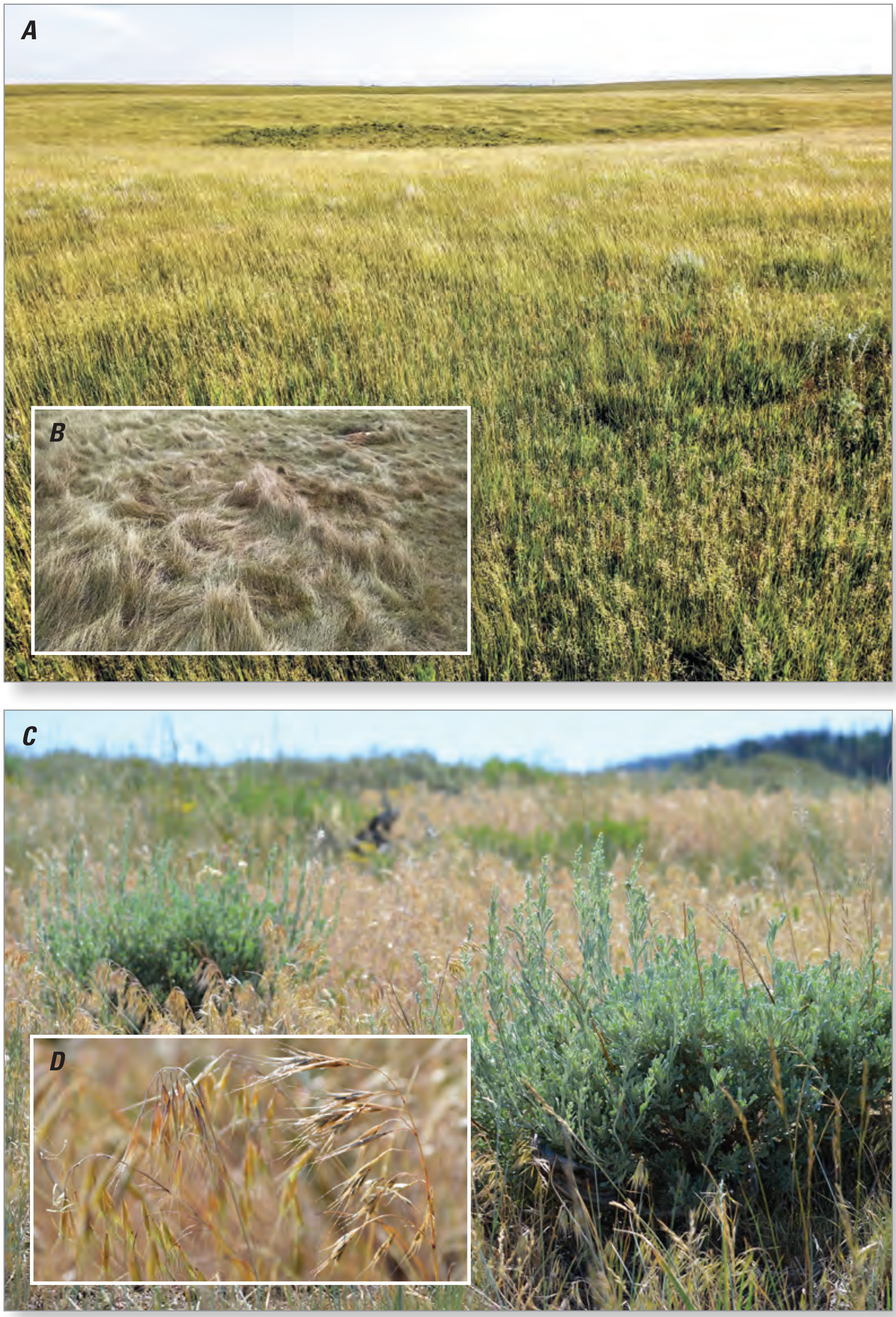


Conversion of native prairie to agricultural uses is the primary cause of grassland loss in North America and has occurred at such a scale that temperate grasslands are one of the most endangered ecosystems on Earth. *A*, Aerial view of the extent of converted grasslands and drained wetlands in one portion of the Prairie Pothole Region of North America, North Dakota; photograph by Krista Lundgren, U.S. Fish and Wildlife Service. *B*, Before and after shots of mixed-grass prairie hayland plowed up for cropland production, Kidder County, North Dakota; photograph by Rick Bohn, used with permission. *C*, Highly erodible cropland that was formerly planted to perennial grass cover in a conservation program but now has been plowed in preparation for seeding back to cropland; photograph by U.S. Geological Survey.

As with grasslands, conversion of wetlands to agricultural uses is the primary cause of wetland loss in the Great Plains. The practice of pattern tile drainage, in which plastic tubing is placed below the surface of the ground, has accelerated the draining and subsequent farming of wetlands. *A*, Installation of tile drainage; photograph by Charles Dahl, U.S. Geological Survey. *B*, Aerial view of a tile-drained field; photograph by Krista Lundgren, U.S. Fish and Wildlife Service. *C*, Wetlands also can be drained through the practice of ditching, as indicated in the middle field by the squiggly lines, as opposed to the undrained wetlands in the field in the foreground; photograph by Krista Lundgren, U.S. Fish and Wildlife Service. *D*, Subsurface tile drainage and ditching allow wetlands to be farmed; photograph by Rick Bohn, used with permission).

drainage pressure (Blann and others, 2009; Werner and others, 2016; Tangen and Finocchiaro, 2017). In the upper Midwest, agricultural producers have increasingly opted to remove land formerly enrolled in conservation programs, many of which included wetlands, and convert them to corn and soybean fields to take advantage of high commodity prices (Miller, 2008). In South Dakota, Wright and Wimberly (2013) estimated that nearly 100,000 ha of grassland conversion occurred within a 100-meter (m) buffer surrounding wetlands, with a similar pattern occurring in North Dakota.

After habitat loss, the second largest threat to biodiversity worldwide is habitat degradation, which refers to the loss of balance among the major influences that maintained biological diversity and ecosystem health (Vitousek and others, 1997; Ricketts and others, 1999). Habitat degradation can be caused through loss of quality, such as by the encroachment of invasive or woody plants, or by fragmentation of remaining expanses of habitat. Non-native, or exotic, invasive plant species encroach into grasslands and outcompete

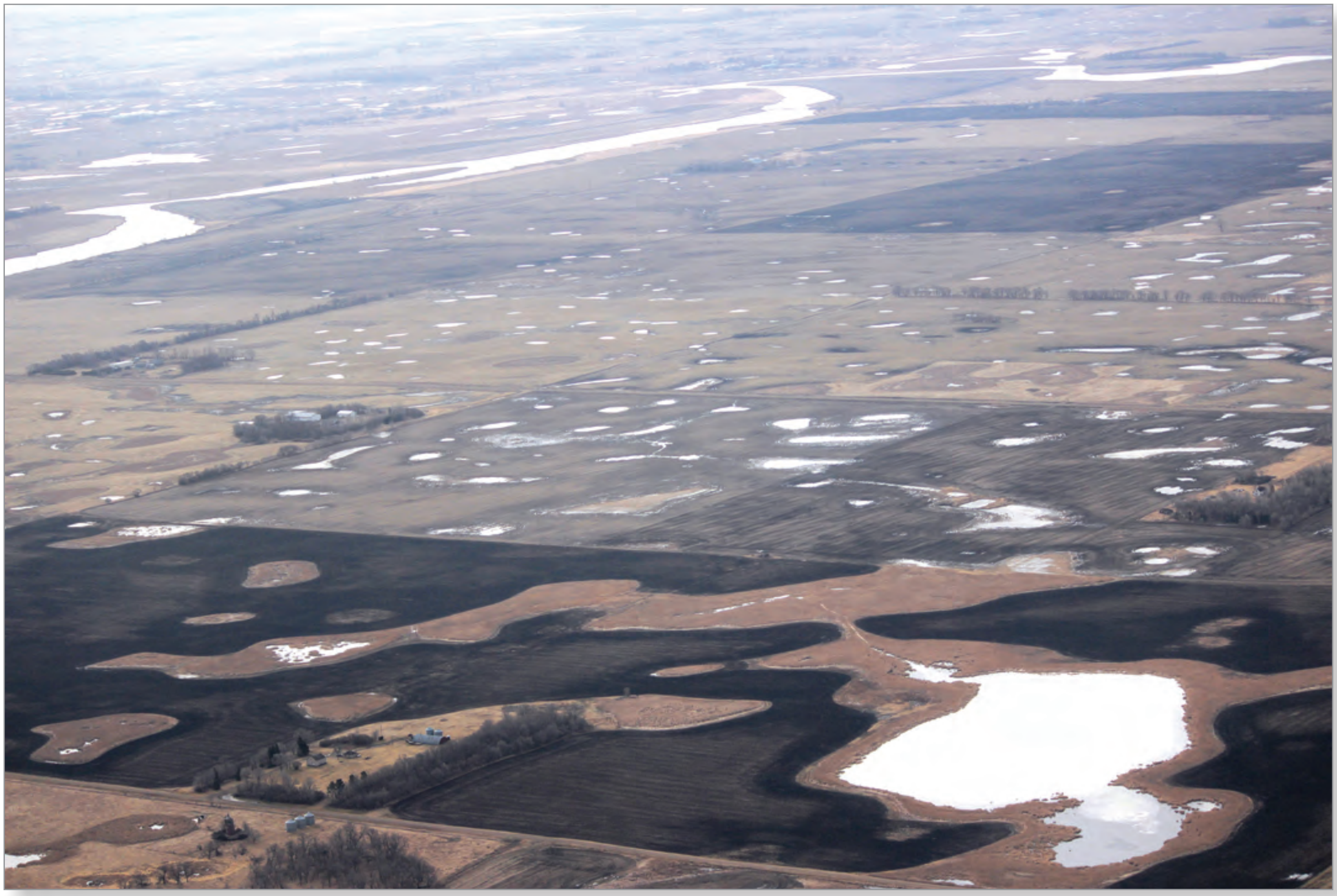


After habitat loss, the second largest threat to biodiversity worldwide is habitat degradation, such as through the encroachment of invasive plant species into native ecosystems. *A* and *B*, In temperate grasslands, Kentucky bluegrass (*Poa pratensis*) is an aggressive invasive species that crowds out native plant species by forming thick stands of residual cover, pictured here invading mixed-grass prairie in North Dakota; photographs by Rick Bohn, used with permission. *C* and *D*, In sagebrush ecosystems, cheatgrass (*Bromus tectorum*) is an aggressive invasive species; photographs by Jennifer Strickland, U.S. Fish and Wildlife Service.

native grassland plant species, thus altering the vegetation structure and ecosystem functions of grassland communities. Woody plant species, either non-native or native, may naturally encroach or may be intentionally planted into grasslands. Degradation also may result from certain management practices, such as rangeland practices that promote the dominance of a few plant species to the detriment of an area's biodiversity (Fuhlendorf and Engle, 2001; Fuhlendorf and others, 2006). Within the United States, 45 percent of the undesirable plant species within pastures are non-native species (Pimental, 1993; Pimental and others, 2005). Samson and others (1998) estimated that 13–30 percent of plant species in the Great Plains are non-native species. Monetary losses to forage crops owing to non-native weeds are nearly $1 billion annually (Pimental, 1993). About $5 billion is spent annually trying to control invasive weeds in pastures and rangelands (Babbitt, 1998). Some non-native plant species were introduced intentionally for agricultural or horticultural purposes and had a competitive advantage over native plant species, especially in disturbed systems. For example, to counteract erosion during the droughts of the 1920s and 1930s, the U.S. Department of Agriculture (USDA) "rehabilitated" rangelands by seeding crested wheatgrass (*Agropyron cristatum*), a Eurasian species that is now a serious threat to the biological integrity of grasslands in western North America and that covers an estimated 25 million ha of North America (Lesica and DeLuca, 1996; Samson and Knopf, 1994). Lehmann lovegrass (*Eragrostis lehmanniana*) and buffelgrass (*Cenchrus ciliaris*), which are native to South Africa, were planted during the 1940s to restore overgrazed rangelands and now dominate millions of hectares of rangeland in the southwestern United States (Flanders and others, 2006). Two highly invasive species, smooth brome (*Bromus inermis*) and cheatgrass (downy brome, *Bromus tectorum*), are responsible for marked changes to grasslands of the Great Plains and shrubsteppe communities of the Intermountain Region (Mack, 1981; Murphy and Grant, 2005; Miller and others, 2011). Cheatgrass outcompetes native species; increases fire frequency that in turn kills and eliminates sagebrush; reduces water filtration into soils; and alters the availability and distribution of nutrients, soil organic matter, and water (Miller and others, 2011). Natural or anthropogenic disturbances also may play a role in creating an opening for introduced species to spread. For instance, fire has the potential to increase the likelihood of invasion by non-native plants (Hobbs and Huenneke, 1992; Miller and others, 2011), and overgrazed pastures may be susceptible to plant invasions (Weaver, 1968; Brown and Archer, 1989). Invasive species can colonize disturbed areas rapidly and gain footholds into native prairie by way of road or railroad rights-of-ways, especially those planted to non-native species (Parker and others, 1993).

Habitat fragmentation refers to the reduction in area of some original habitat, a change in spatial configuration (that is, spatial arrangement), and an increasing distance between patches of what remains, through the subdivision of continuous habitat into smaller pieces (Andrén, 1994; Villard, 2002). The effects of fragmentation on organisms are difficult to isolate experimentally and difficult to summarize into concise management guidelines (Haila, 2002; McGarigal and Cushman, 2002; Schmiegelow and Monkkonen, 2002; Villard, 2002). Villard (2002) and Haila (2002) stressed that fragmentation effects are highly specific to taxa, to spatial scales, and to the ecological processes under consideration; vary according to landscape type and structure; and their influence on species distribution and abundance is obscured by local or regional effects. Fragmentation causes a loss of habitat heterogeneity, and with it, a loss of biodiversity; fragmentation also lowers habitat quality because of edge effects, such as lower avian reproductive success near the edge than interior of remaining habitat (Ribic and others, 2009). The importance of understanding the ecological impacts of grassland size is discussed further in the section below titled "Considerations in Grassland Reserve Design."

Since settlement, there has been a persistent effort to plant trees and shrubs in the open habitats of the Great Plains (McNicholl, 1988). The introduction of woody vegetation into grasslands creates conditions of habitat degradation and fragmentation. In the 1870s, States and territories offered cash rewards or land titles to settlers who planted trees (Griffith, 1976). Beginning in the 1930s, in response to the devastating effects of the Dust Bowl years, Federal initiatives, such as the U.S. Forest Service's Prairie States Forestry Project, encouraged tree plantings in the Great Plains to reduce soil erosion; ameliorate the dessicating and destructive conditions produced by strong winds that affected crops, livestock, and homesteads; reduce fuel costs of heating homes; supply wood for fuel and lumber; function as living snow fences; and provide food and cover for wildlife (Tinus, 1976; Baer, 1989). In the United States, Hanks (1976, p. 2) wrote, "Between 1935 and 1942, more than 200 million trees and shrubs were planted on 30,000 farms in windbreak strips totaling 18,600 miles (mi) in length. The planting zone extended from the Canadian border to the Texas Panhandle." Besides reducing the area of grassland, the establishment of woodlots, shelterbelts, and windbreaks within the prairie has facilitated changes in the vertebrate community in the Great Plains, sometimes to the detriment of grassland-obligate species (Knopf, 1986; McNicholl, 1988; Samson and Knopf, 1994; Igl and Johnson, 1997).



As native habitats are lost to conversion, the parcels that remain are beset by low biodiversity, high amounts of habitat edge, and increasing distances to other parcels, all factors that lower their habitat quality. Aerial view of a fragmented portion of the Prairie Pothole Region of North America, North Dakota; photograph by Krista Lundgren, U.S. Fish and Wildlife Service.

Conservation of Grassland and Wetland Habitats

Management and conservation of native grasslands has occurred at several scales, by governmental and private entities, and at various durations from temporary to permanent protection. The size of grassland management units ranges from several hectares administered by one of the more than 1,900 private land trusts in the United States (National Land Trust Alliance, 2015) to more than 1.5 million ha in the 20 national grasslands administered by the U.S. Forest Service (Olson, 1997). In addition to the national grasslands in the United States, grasslands are permanently protected by other Federal agencies, such as the FWS, which manages national wildlife refuges, waterfowl production areas, and other fee-title lands (Niemuth and others, 2008); Bureau of Land Management, Bureau of Reclamation, U.S. Army Corps of Engineers, and National Park Service (Kirby and others, 1992; U.S. Department of the Interior, 2019). State agencies also protect grasslands in State-owned wildlife management areas.

Waterfowl Production Areas, such as this one at Long Lake National Wildlife Refuge in North Dakota, are administered by the U.S. Fish and Wildlife Service for the protection of grasslands, wetlands, and wildlife; photograph by U.S. Fish and Wildlife Service.

Of course, Federal and State agencies and private entities manage grasslands for a variety of purposes, not exclusively for grassland birds (Ryan, 1990). Protection through private means may occur through the actions of individual landowners or through local and State land trusts. Non-government organizations (NGOs), such as The Nature Conservancy and Ducks Unlimited, and State and local land trusts had protected nearly 14 million ha as of 2005 (National Land Trust Alliance, 2015). These privately owned grasslands are becoming increasingly important because of the many constraints (for example, increasing bureaucracy, shrinking budgets and staff) inherent to Federal and State agencies.

In Canada, wetlands and uplands are protected by the Canadian Wildlife Service, which administers Federal Migratory Bird Sanctuaries, National Wildlife Areas, the National Parks network, grasslands rehabilitated through the Prairie Farm Rehabilitation Act, and other habitats protected by Provincial agencies and NGOs (Beyersbergen and others, 2004). Groups such as The Nature Conservancy and Ducks Unlimited work across national boundaries to protect grasslands or other habitats in the United States, Canada, and many other countries (Ducks Unlimited, 2019; The Nature Conservancy, 2019).

Other forms of grassland protection are conferred through cost-sharing programs or conservation easements between private landowners and the Federal, State, or local agencies or private organizations administering the programs. States vary in the types of programs and the length of conservation protection that they offer. One example is the Private Lands Initiative of North Dakota offered by the North Dakota Game and Fish Department (North Dakota Game and Fish Department, 2016). The programs under this initiative offer cost-sharing assistance to landowners who, in return, provide habitat for wildlife and allow walk-in hunting opportunities for the public. The initiative also includes incentives to landowners to limit haying and grazing on their land, and the program will match money from Federal grants for the maintenance, enhancement, and restoration of wetlands and grasslands.

As with State programs, Federal easement initiatives vary in the types of programs and length of protection. The easement program within the FWS was established from a strong foundation and history of land protection and acquisition. The Migratory Bird Hunting and Stamp Act of 1934 provided a means to generate funds for land acquisition through the required purchase by adult waterfowl hunters of the Duck Stamp (FWS, 2017). In 1958, the Small Wetlands Acquisition Program was created; this legislation authorized the acquisition of Waterfowl Production Areas (WPAs) involving small wetlands and potholes (FWS, 2017). In 1962, Wetland Management Districts were formed. In 1989, the Small Wetlands Acquisition Program was expanded to include the acquisition of upland easements to improve the quality and availability of waterfowl nesting habitat. Beginning in the 1990s, the FWS began to purchase permanent grassland easements to augment existing or new wetland easements. As of 2017, nearly 1 million ha of habitat have been protected through the Small Wetlands Acquisition Program (FWS, 2017).

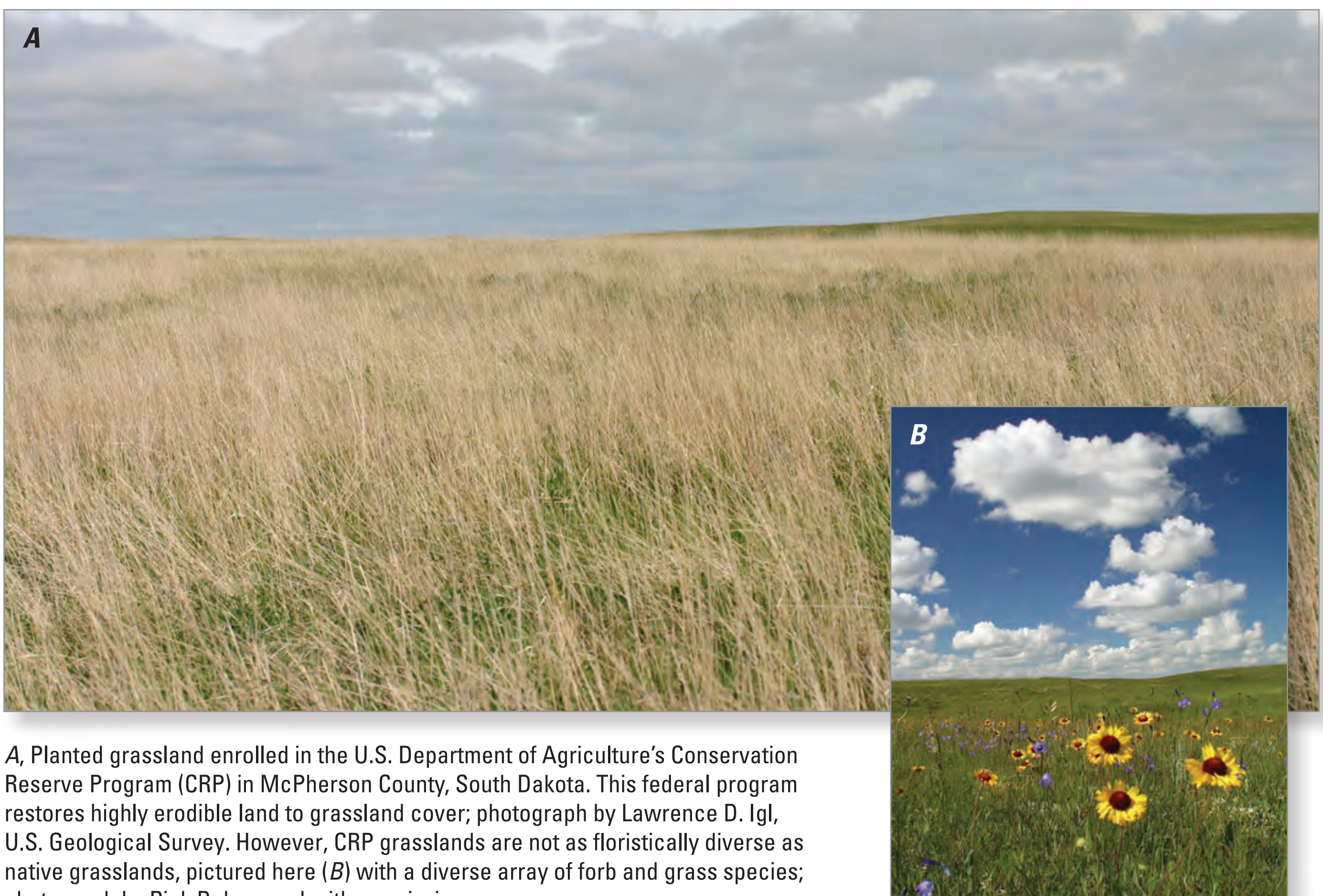
Neal D. Niemuth (FWS, Bismarck, North Dakota, written commun. [n.d.]) offered the following insights on easement programs:

> Easement programs offer many advantages and some disadvantages relative to other conservation strategies and are increasingly being used to conserve grasslands. Easements have low initial cost relative to fee-title acquisition, have no long-term management costs to agencies, and are typically better accepted by the public than fee-title acquisition in that lands stay on the tax roll and agricultural presence in the community is not diminished. Easements also are more attractive to landowners because easement payments can help pay debt, landowners retain control over the land, and land can still be used for livestock and hay production. Grazing is by far the largest land use on grassland easements. Livestock producers do not receive many of the considerable Federal subsidies received by row-crop producers, so an easement payment helps offset the financial incentive to plow grass and plant crops. One of the best things any grassland conservation program can do is keep ranchers on the land so the grass stays 'green side up.' Ranching and grazing also can be encouraged through assistance with cattle watering projects and development of grazing systems. In the United States, the FWS has extensive easement acquisition programs, funded primarily through sale of Federal Duck Stamps, to protect grassland habitat for waterfowl. These easements are perpetual and require that grasslands remain intact and undisturbed from plowing, disking, spraying, etc. Grazing is allowed year-round, but haying is only allowed after July 15 to reduce loss of nests and young. Compliance with easement requirements is monitored annually on all easement parcels. FWS easement programs have resulted in the perpetual conservation of more than 420,800 ha of grassland, primarily native prairie, in North Dakota and South Dakota. Although funded by waterfowl conservation programs, these grasslands benefit a host of other grassland species, including native prairie specialists such as Thick-billed Longspur (*Rhynchophanes mccownii*), Baird's Sparrow (*Centronyx bairdii*), and Sprague's Pipit (*Anthus spragueii*).

FWS easement wetlands account for about 8.5 percent of the remaining wetland area in the Prairie Pothole Region, and about 70 percent of the remaining wetlands are in private ownership and unprotected by Federal legislation (Dahl, 2014). Easement programs vary considerably in the length of time that they offer conservation benefits. The programs also vary in the restrictions placed on landowners. The programs also differ in their effect on taxable value of the land and management costs, which affect participant interest.

Other Federal programs also confer protection. The Partners for Fish and Wildlife program administered by the FWS assists private landowners with habitat restoration, development, and management on their property and protects grasslands and wetlands under term leases (Beyersbergen and others, 2004). The USDA's Natural Resources Conservation Service administers the Agricultural Conservation Easement Program that provides financial and technical assistance to help conserve agricultural lands and wetlands; the Wetlands Reserve Easements component restores, protects, and enhances enrolled wetlands (USDA, 2018). The USDA formerly offered three easement programs that protected extant native grasslands or provided incentives for creating grassland habitat (USDA, 2018). The Wetlands Reserve Program established grasslands of seeded native plant species on land that was formerly cropland with associated degraded wetlands. The Farm and Ranch Land Protection Program protected land for agricultural purposes including native grassland habitats. The Grassland Reserve Program restored and protected grassland, including rangeland and pastureland, while maintaining the area as grazing lands. These programs were eventually discontinued owing to lack of funding. Other conservation programs for private lands offered through the USDA included the Environmental Quality Incentive Program, the Conservation Reserve Enhancement Program, and the Wildlife Habitat Incentive Program. These programs did not protect grassland habitats through easements but provided payments to private landowners to restore and manage native or tame grasslands for 10–15 years (USDA, 2018).

One of the most effective and largest grassland conservation programs to date has been the Conservation Reserve Program (CRP), which is administered by the USDA's Farm Service Agency. This program has been effective at restoring highly erodible land to grassland cover and providing habitat for wildlife. Numerous studies have shown that grassland birds have benefitted from the millions of hectares of perennial grasslands established under the CRP (Johnson and Schwartz, 1993a, 1993b; Johnson and Igl, 1995, 2001; Rodenhouse and others, 1995; Patterson and Best, 1996; Ryan and others, 1998; Igl and Johnson, 1999; Heard and others, 2000; Coppedge and others, 2001); however, CRP contracts with landowners offer only short-term (usually 10–15 years) protection from tillage. Recent incentives to expand production of major field crops and the current demand to use crops for biofuel production has negatively influenced CRP contract renewals. For example, CRP enrollment peaked in 2007 at 14.9 million ha and then declined by more than 25 percent, with much of this land returning to agriculture (Morefield and others, 2016).



A, Planted grassland enrolled in the U.S. Department of Agriculture's Conservation Reserve Program (CRP) in McPherson County, South Dakota. This federal program restores highly erodible land to grassland cover; photograph by Lawrence D. Igl, U.S. Geological Survey. However, CRP grasslands are not as floristically diverse as native grasslands, pictured here (*B*) with a diverse array of forb and grass species; photograph by Rick Bohn, used with permission.

North American Sagebrush Habitats Before and After European Settlement

The original intent of this series, “Effects of Management Practices on Grassland Birds,” was to provide a literature review that would synthesize information on the habitat requirements and effects of habitat management on grassland birds, with primary emphasis on the northern Great Plains. Over time, the focus expanded to include other grassland communities of the Great Plains as well as sagebrush communities of the Great Basin and elsewhere. To that end, we provide a brief description of the sagebrush ecosystem and changes in habitat quality and quantity in this system from a variety of stressors.

Sagebrush communities in North America extend from British Columbia and Saskatchewan to northern Arizona and New Mexico and from the eastern slopes of the Sierra Nevada and Cascade mountain ranges to western South Dakota (Miller and others, 2011). The sagebrush biome can be divided into three main vegetation types, including two in the Intermountain Region and one in the northern Great Plains: (1) sagebrush steppe, dominated by big sagebrush (*Artemisia tridentata*) and perennial bunchgrasses; (2) Great Basin sagebrush, also dominated by sagebrush but with a sparse understory; and (3) mixed desert shrubland of the northern Great Plains, dominated by big sagebrush, prairie sagewort (*Artemisia frigida*), silver sagebrush (*Artemisia cana*), and sand sagebrush (*Artemisia filifolia*) (Küchler, 1964; Miller and others, 2011). Further subdivisions have been defined based on differences in climate, elevation, topography, floristics, geology, soils, and disturbance history (Miller and others, 2011).

The geologic history of sagebrush communities east of the Rocky Mountains is similar to that of the Great Plains. The uplift of mountains reduced the influence of maritime air from the Pacific Ocean and resulted in semi-arid conditions (Mack and Thompson, 1982). The drier climate, in combination with frequent large fires, allowed sagebrush and grasses to supplant forests (Miller and others, 2011). Unlike the Rocky Mountains, however, the Cascade and Sierra mountain ranges are not high enough to obstruct all maritime air (Mack and Thompson, 1982); therefore, the Intermountain Region does experience a moderating influence from the prevailing westerly winds. The peak of annual precipitation in this region occurs during autumn and winter, which differs from the early

Sagebrush (*Artemisia* spp.) in Sublette County, Wyoming; photograph by Mary Rowland, U.S. Forest Service.

summer peak in prairies east of the Rocky Mountains. The differences in the timing of precipitation between the two regions are reflected in differences in growth forms of the dominant grasses. East of the Rocky Mountains, the grasses are characterized by rhizomatous or stoloniferous grass species (Daubenmire, 1978; Mack and Thompson, 1982). In the Intermountain Region, the grass species grow in characteristically clumped (that is, caespitose) growth forms.

Based on fossil evidence, the biota of the Intermountain Region appears to have evolved over several million years, with grazing as a natural ecological driver (Burkhardt, 1996). Massive extinctions during the Pleistocene removed many large herbivores from this region about 10,000 years ago. Bison continued to be widely distributed in this region but were largely extirpated from the area just prior to the arrival of European settlers. In contrast to the eastern prairies, where large herbivores were nomadic grazers with few seasonal patterns, in the Intermountain Region, large herbivores developed seasonal grazing patterns to deal with the short growing season and the protein-deficient foraging environment (Mack and Thompson, 1982; Burkhardt, 1996).

Estimates of historical fire-return intervals for the sagebrush biome range from more than 200 years in little sagebrush (*Artemisia arbuscula*) to 200–350 years in Wyoming big sagebrush (*Artemisia tridentata* ssp. [subspecies] *wyomingensis*) and 150–300 years in mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) (Baker, 2011). This wide range reflects regional differences, variable responses to fire among taxa of sagebrush, and the quantity and quality of fuel loads as influenced by precipitation. However, in sagebrush communities invaded by cheatgrass (downy brome) or other exotic annual grasses, fire intervals are much shorter (that is, 5–10 years in Wyoming big sagebrush; Innes, 2016), and complete elimination of sagebrush has occurred following grass-fueled fires (Billings, 1994; Monsen, 1994; Crawford and others, 2004; Miller and others, 2011). Increased fire frequency eliminates shrubs, disturbs soils and microbiotic crusts, and releases nutrients, all actions that favor the invasion of annual exotic plant species and reduce the stability of the sagebrush ecosystem.

Miller and others (2011) estimated that 45 percent of the historical distribution of sagebrush in western North America has been lost to agricultural uses, urbanization, or degradation caused by the encroachment of woody vegetation or increased fire exacerbated by annual grasses. Prior to settlement, the sagebrush biome was dominated by sagebrush and bunchgrasses. After settlement, this biome became increasingly dominated by sagebrush, woodlands, and invasive annual plants. Two Eurasian annual grasses, cheatgrass and medusahead (*Taeniatherum caput-medusae*), are among the most aggressive invasive weeds degrading native sagebrush communities. These two species now dominate or have had a significant impact on 17.5 percent of the 400,000 km^2 of sagebrush on public land surveyed in five western States (Washington, Oregon, Nevada, Idaho, and Utah; Meinke and others, 2009; Miller and others, 2011). Invasive species change the structure and composition of the understory and support more frequent and more destructive fires, which results in fewer unburned patches and more widely dispersed sagebrush seed sources (Miller and others, 2011). Woodland species (primarily pinyon [*Pinus* spp.] and juniper) have encroached into 60–90 percent of the sagebrush biome. Miller and others (2011) estimated that about 12 percent of the current distribution of sagebrush will be replaced by other woody vegetation for each 1 degree Celsius (°C) increase in temperature that occurs with projected climate change.

Livestock grazing has occurred over virtually the entire sagebrush ecosystem and thus its influence is perhaps the most pervasive of any land management practice in this system (Knick, 2011; Knick and others, 2011; Boyd and others, 2014). Livestock grazing serves as a form of disturbance with diffuse effects from repeated pressure (Knick and others, 2011). Effects of livestock grazing on vegetation species composition and structure in sagebrush communities have been well documented (Vale, 1974; Owens and Norton, 1992; West, 1999; Belsky and Gelbard, 2000; Jones, 2000; Anderson and Inouye, 2001). Notably, grazing can exacerbate the dominance of cheatgrass in sagebrush systems (Reisner and others, 2013). Accurately quantifying effects of grazing on sagebrush at broad scales, however, is challenging owing to the lack of sufficiently large control areas (Knick and others, 2011). Interactions of livestock grazing with other factors, such as wildfire, are complex and not widely studied. However, Boyd and others (2014) modeled effects of livestock grazing and fire using state and transition models and concluded that carefully managed grazing at moderate intensities can be compatible with maintaining ecosystem function in sagebrush communities.

The remaining stands of sagebrush occur in landscapes that are increasingly dominated by agriculture and urbanization (Knick and others, 2011). Croplands are estimated to influence between 41 and 73 percent of sagebrush habitat in North America (Knick and others, 2011). Vander Haegen and others (2000, 2002) demonstrated that habitat fragmentation and degradation can negatively impact some sagebrush-obligate avian species through, for example, increased nest predation near habitat edges.

Grassland Birds

A grassland bird is a species that relies on grassland habitats to support some portion of its life cycle, including breeding, migration, or wintering needs (Mengel, 1970; Vickery and others, 1999). The vegetation structure of grassland habitats is an important determinant of abundance and nest-site selection in grassland birds (Wiens, 1969; Davis, 2003). Any process that alters that vegetation structure has the potential to reduce or enhance habitat quality for a grassland bird species, depending on the species' habitat needs and preferences. As illustrated in the series of species accounts that compose this compendium, "The Effects of Management Practices on Grassland Birds," and others (Rotenberry and Wiens, 1980; Kantrud, 1981; Cody, 1985), individual bird species have affinities for grassland habitats with specific structural characteristics. Bird populations are influenced by the degree of habitat heterogeneity within grasslands (Fuhlendorf and Engle, 2001; Wiens, 1974a, 1974b). The diversity of habitat requirements among grassland birds attests to the importance of providing heterogeneity within grasslands and landscapes to support the full spectrum of grassland birds in a region (Ryan, 1990; Fuhlendorf and Engle, 2001; Fuhlendorf and others, 2006). In many native grasslands, such as in the Prairie Pothole Region of northern North America, wetlands are an integral component of the grassland ecosystem, and grassland birds have evolved to use wetland habitats as well as grassland habitats, particularly those wetland types (temporary and seasonal) that function as grasslands part of the year. Land managers aiming to conserve the true character of grasslands and managing for high biological diversity recognize the importance of maintaining the ecological connectivity between grasslands and wetlands. For this reason, although grassland management is the primary focus of this section, wetlands will remain part of the management discussion where appropriate.

Anthropogenic changes to the ecological factors shaping grasslands have affected grassland birds to the extent that they are experiencing greater and more consistent patterns of decline than any other group of North American species (Droege and Sauer, 1994; Sauer and others, 2013). The two most important factors implicated in this decline are grassland loss and degradation (Askins, 1993; Wilcove and others, 1998), as discussed in the previous section, "Factors Contributing to the Loss and Degradation of Grassland and Wetland Habitats." Population declines will not stop or be reversed without the protection of remaining native grasslands and the establishment and maintenance of human-created grasslands to compensate for past losses of grassland habitat. Wetland drainage for agriculture and human developments directly affects wetland-dependent birds but also impacts upland-nesting species, such as grassland birds, through the loss of a water source and alteration of cover during the breeding and wintering seasons (McNicholl, 1988; Knopf, 1994; Igl and Johnson, 1999; Dugger and Dugger, 2002). Dry wetlands provide important nesting areas for some grassland birds during drought (Hubbard, 1982).

Use of Human-Created Grassland Habitats by Grassland Birds

Despite the many anthropogenic changes to North American grasslands, some grassland bird species are adaptable and opportunistic in their habitat selection and now utilize one or more human-created habitats (Vickery and others, 1999). Human-created grasslands include pastures, hayfields, agricultural terraces, crop buffer strips, field borders, grassed waterways, fencerows, road rights-of-way, airports, reclaimed coal mines, and planted wildlife cover. Fields of seeded grasslands enrolled in Federal long-term set-aside programs, such as the CRP in the United States and the Permanent Cover Program (PCP) in Canada, provide important nesting habitat for grassland birds (McMaster and Davis, 2001; Allen and Vandever, 2012). These programs were designed primarily to reduce soil erosion and crop surpluses but also featured the additional benefit of providing wildlife habitat. Although the types and frequencies of disturbances differ among the aforementioned human-created grassland habitat types, some of these habitats may be viewed as surrogates for native grasslands (Sample and Mossman, 1997). Pastures with domestic livestock are a common feature of rural areas in the Great Plains. Pastures may include unbroken native prairie, grasslands planted to a limited number of native or non-native species of grasses and forbs, and grasslands planted to a variety of native and

Some species of grassland birds have adapted to using human-created grassland habitats, such as terraces shown here in Shelby County, Iowa (*A*), and contoured buffer strips shown here in Tama County, Iowa (*B*), but these habitats are often constrained in size and are low in plant diversity and high in amount of habitat edge; photographs by U.S. Department of Agriculture.

non-native grass species, forbs, shrubs, and sedges (for example, Renfrew and Ribic, 2001, 2002). Depending on the vegetation structure and size of the pastures, these areas may be used as nesting habitat by grassland bird species (Renfrew and Ribic, 2001, 2002) and, to some extent, seeded hayfields and pastures may serve as suitable grassland habitat (Herkert and others, 1996). However, pastures and hayland habitats have declined by more than 50 percent during the past 100 years in the Midwest. Igl and Johnson (1997) determined that the area of hayland declined 52 percent between 1967 and 1993 in North Dakota. In the Midwest, populations of Eastern Meadowlarks (*Sturnella magna*), western Meadowlarks (*Sturnella neglecta*), Bobolinks (*Dolichonyx oryzivorus*), Grasshopper Sparrows (*Ammodramus savannarum*), Dickcissels (*Spiza americana*), and Savannah Sparrows (*Passerculus sandwichensis*) declined concurrently with the declines in pasture area, but generally not with hayfield area, suggesting that midwestern pastures are important for grassland birds and that their loss may have contributed to population declines of grassland birds (Herkert and others, 1996).

Several linear grassland habitats are common in agricultural landscapes, including habitats that function as part of the agricultural system and those that occur as edges between different habitat types. These areas include terraces, buffer strips, field borders, grassed waterways, and fencerows. Linear agricultural habitats may support grassland bird species that are not commonly found in cultivated fields, in part, because of the different management practices applied to the two different areas (Rodenhouse and others, 1995). Terraces are dirt embankments that have been seeded to grassland vegetation; terraces typically occur in agricultural fields with moderate-to-steep slopes and are designed to trap soil and reduce erosion (Hultquist and Best, 2001). In Iowa, birds used grassed terraces more than adjacent rowcrop fields but less than nearby grassed waterways and roadsides (Hultquist and Best, 2001). Field borders may be an important linear habitat for grassland birds in agricultural areas, but the number and size of field edges has been declining as cropland field sizes have been increasing over time with the development of large-scale agricultural practices and larger machinery (Rodenhouse and others, 1995; Higgins and others, 2002). In the central United States, field edges have declined by 30–80 percent since the 1930s (Rodenhouse and others, 1995). Grassed waterways are linear strips of grassland habitat in highly erodible areas in agricultural fields (Bryan and Best, 1991); these linear grassland habitats slow water movement and typically are planted to cool-season grasses to reduce erosion. In Iowa, more species and greater abundances of birds occurred in grassed waterways than in surrounding soybean and corn fields (Bryan and Best, 1991). Schulte and others (2016, 2017) determined that the number of bird species was 1.5 to 2.0 times higher in Iowa rowcrop fields that incorporated strips of native perennial grass species than fields without grass strips.

Road and transmission line rights-of-ways may provide remnant strips of grassland of varying vegetation structure that some birds may use for nesting (Camp and Best, 1994; Leston and Koper, 2017). In eastern North America, where native grassland habitats have diminished greatly in size, airport grasslands may serve as refugia for some grassland birds (Caccamise and others, 1996). For example, Snyder and others (1987) found Upland Sandpipers (*Bartramia longicauda*) in only five sites in Indiana, one of which was an airport. However, airports may not support all of the grassland bird species that historically occurred in an area. Small, rural airports in the Midwest may be population sinks for some grassland birds. For example, in Illinois, grassland birds, such as Eastern Meadowlark, Grasshopper Sparrow, Savannah Sparrow, and Horned Lark (*Eremophila alpestris*), nested on airports, but all species experienced nest destruction as a result of mowing operations (Kershner and Bollinger, 1996).

Areas that have been reclaimed from previous uses and planted or restored to grasslands or wetlands may provide important habitat for grassland birds. Inactive coal mines that have been reclaimed to grasslands provide large blocks of habitat for grassland birds (Bajema and others, 2001; DeVault and others, 2002; Ingold, 2002; Scott and others, 2002). Seeding areas with grassland vegetation has been the dominant reclamation approach since the 1960s and 1970s in the Illinois coal basin region owing to the ease, low cost, and quickness in reducing soil erosion as compared with planting trees (Brothers, 1990). Scott and others (2002) reported no difference in grassland bird use of reclaimed coal-mine grasslands and native prairie, even when exotic grasses were a dominant cover type in the reclaimed grasslands. Reclaimed grasslands may provide important nesting habitat for some declining populations of grassland birds. Henslow's Sparrows (*Centronyx henslowii*), for example, occupy reclaimed coal-mine grasslands in Indiana to a degree that may help stabilize the species' population in the area (Bajema and others, 2001). Reclaimed coal mines that have been restored to native grass species have some characteristics especially beneficial to grassland birds, such as large grassland size and single ownership that may be conducive to consistent management practices and that may lower the risk of conversion to nongrassland habitats (DeVault and others, 2002; Scott and others, 2002).

Grasslands managed by Federal and State agencies for wildlife often are planted to mixes of grass and forb species. The WPAs, managed by the FWS, are blocks of land that include both wetland and upland habitats, some of which have been reclaimed from agricultural production (Duebbert, 1981). In North Dakota, WPAs may include a mixture of grassland types, such as mixed-grass prairie and tame-grass pastures, and these areas provide important nesting habitat for many grassland bird species. Many WPAs and other seeded grasslands have been planted to dense nesting cover (DNC), a mixture of grasses and legumes intended to provide tall and dense wildlife cover (Duebbert and others, 1981). Although this habitat is specifically intended to create nesting sites for upland-nesting waterfowl, DNC also may provide nesting habitat for many species of birds, including upland gamebirds, shorebirds, waterbirds, and songbirds. For example, in North

Dakota, DNC grasslands that were seeded to alfalfa (*Medicago sativa*)-wheatgrass mixtures supported high breeding densities of Bobolink, Sedge Wren (*Cistothorus stellaris*), and Savannah Sparrow (Renken and Dinsmore, 1987). In Saskatchewan, DNC planted primarily to native grasses had avian species richness, abundance, and productivity indices that were similar to native grasslands (Hartley, 1994).

In the United States, Government set-aside programs have helped create wildlife-friendly, albeit temporary, grassland habitat on private lands (Duebbert and others, 1981; Sample and Mossman, 1997). The Soil Bank Program of the 1950s and 1960s enabled farmers to retire cropland from production and to plant introduced grasses and legumes as a cover crop (Duebbert and others, 1981). Other set-aside programs were included in subsequent Farm Bills. The Soil Bank Program was followed by the Cropland Adjustment Program, which was then succeeded by the CRP. The CRP was established in 1985 and paid landowners to plant grasses and other perennial cover on highly erodible agricultural land in an effort to reduce erosion, decrease crop surpluses, and provide wildlife habitat (Young and Osborn, 1990; Rodenhouse and others, 1995; Ryan and others, 1998; Heard and others, 2000). Although CRP grasslands are floristically less diverse than native prairie (Higgins and others, 2002), several declining grassland bird species occur in CRP fields during the breeding season, such as Dickcissel, Lark Bunting (*Calamospiza melanocorys*), Baird's Sparrow, Grasshopper Sparrow, Clay-colored Sparrow (*Spizella pallida*), and Bobolink (Johnson and Schwartz, 1993a; Johnson and Igl, 1995; Herkert, 1997b, 1998; Ryan and others, 1998). Ryan and others (1998) reviewed literature on bird use of CRP grasslands and determined that more than 90 species have been reported using CRP grasslands during the breeding season and that at least 42 species have nested in these habitats. In a long-term study (1990–2008) in the northern Great Plains, Igl (2009) reported 149 bird species using CRP grassland fields during the breeding seasons, including at least 66 species that have shown evidence of nesting. In the Midwest, CRP fields may support from 1.4 to 10.5 times as many birds as cropland supports (Ryan and others, 1998). In Indiana, Iowa, Kansas, Michigan, Missouri, and Nebraska, CRP fields supported 3 times the density of nesting bird species and 13 times the density of nests as rowcrop fields, but nesting success was similar between CRP and rowcrop fields (Best and others, 1997). In Oklahoma, populations of some grassland bird species have increased through time along with increasing coverage of CRP grasslands (Coppedge and others, 2001). Johnson and Igl (1995) estimated that a return of CRP acreage to cultivation would result in a 17-percent decline in populations of several grassland bird species in North Dakota. Moreover, the benefits of CRP grasslands may depend on the landscape context within which the fields are embedded. Coppedge and others (2001) determined that grassland birds showed a positive response to CRP grasslands in areas most affected by juniper invasion but did not respond in areas where native grasslands were abundant and structurally sound. Johnson and Igl (2001) concluded that locating a CRP field near existing grasslands, or establishing one large rather than several small CRP fields, would benefit more grassland bird species than would creating small, isolated CRP fields.

Despite the many obvious benefits of the CRP (Allen and Vandever, 2012), the program is not without its shortcomings. The benefits of CRP grasslands to breeding birds are largely temporary because enrollment is dependent on landowner interest, economic conditions, length of contracts (which generally are limited to 10–15-year periods), and periodic renewal of the program by the U.S. Congress in subsequent Farm Bills. CRP grasslands that are removed from the program often revert back to cropland. Moreover, the CRP alone may not be enough to stem the loss of native prairie or reverse the declines in all grassland bird populations (Vickery and Herkert, 2001). In some areas, the acreage of CRP grasslands has not been enough to offset continued losses of grassland habitat in recent times (Vickery and Herkert, 2001). The 7.3 million ha of CRP grassland in the northern Great Plains covers almost the same area of native prairie that had been converted to cropland since the 1960s (Higgins and others, 2002). In some regions, the attractiveness of the CRP and its financial incentives may have encouraged some landowners to convert native prairie to newly created croplands, making these fields eligible for CRP payments after a cropping history has been established. Since the inception of the CRP in 1985, more than 404,000 ha of native prairie were lost in South Dakota, North Dakota, and Montana (Higgins and others, 2002). In States with abundant CRP coverage, CRP fields may reduce habitat fragmentation that is typical of agricultural areas (Rodenhouse and others, 1995); however, in States with less abundant CRP coverage, CRP fields may be too small and too poorly configured to support some grassland birds (Vickery and Herkert, 2001). Although breeding bird densities often are higher in CRP grasslands than in the cropland that they replaced, in some regions, CRP grasslands may act as population sinks for some grassland bird species (McCoy and others, 1999).

Canada's PCP, established in 1989, encourages landowners to convert agricultural lands with poor soils to grass cover for at least 10 years (McMaster and Davis, 2001). As with the CRP, PCP habitats provide important alternative nesting habitat for many grassland species. In Alberta, Saskatchewan, and Manitoba, PCP sites were characterized by taller, denser vegetation and less bare ground than cropland sites. There were more avian species, and the abundances of nine of 10 common grassland bird species were greater on PCP fields than on agricultural fields (McMaster and Davis, 2001).

Use of Agricultural Lands by Grassland Birds

Any discussion of management effects on grassland bird populations is incomplete without a discussion of agricultural fields. Many grassland bird species use agricultural fields during the breeding season, including for nesting, foraging, and brood rearing (Rodenhouse and others, 1995). Small-grain cropland (for example, wheat [*Triticum* spp.], barley [*Hordeum* spp.], rye [*Secale* spp.]) may provide suitable nesting habitat because small grains closely resemble grasslands in height and structure and because small grains often are harvested late enough to provide suitable nesting habitat for some grassland birds (Rodenhouse and others, 1993; Sample and Mossman, 1997). However, avian diversity and density in small-grain cropland usually is low (Johnson and Igl, 1995; Best and others, 1997; Samson and others, 1998; Johnsgard, 2001). Rowcrops such as corn and soybeans, on the other hand, are harvested later than small grains but generally are poor surrogates for grassland habitats. Nonetheless, a few grassland species nest in rowcrop fields (for example, Vesper Sparrow [*Pooecetes gramineus*], and Horned Lark). Species such as Vesper Sparrow, Horned Lark, Upland Sandpiper, Chestnut-collared Longspur (*Calcarius ornatus*), and Killdeer (*Charadrius vociferus*) may be more common in cropland than in some seeded grasslands, whereas species such as Sedge Wren, Grasshopper Sparrow, and Savannah Sparrow may occur at lower densities or may not be present in cropland (Johnson and Igl, 1995).

Farming practices have changed dramatically during the past century (Rodenhouse and others, 1993, 1995; Higgins and others, 2002). Modern changes or patterns in agricultural production that are detrimental to bird populations include reduction in farmland devoted to pasture and hayland, increased production of corn and soybeans, larger farms and field sizes, lower crop and cover diversity, and increased use of agricultural chemicals (Farris and Cole, 1981; Rodenhouse and others, 1993; Higgins and others, 2002). In the northern Great Plains, less farmland is devoted to small grains, such as wheat and barley, which provide reasonably good cover for some nesting grassland birds, and more area is planted to soybeans and corn, which provide poor cover for grassland-nesting birds (Higgins and others, 2002; Lark and others, 2015). Modern farms maintain fewer grassy field edges or fencerows (Higgins and others, 2002). Modern changes in

Some species of grassland birds have adapted to using small-grain cropland fields such as wheat (*Triticum* spp.) fields, but these habitats have low plant and animal diversity and may be subjected to mechanical disturbances while birds are still nesting; photograph by Rick Bohn, used with permission.

agricultural patterns that are advantageous to bird populations are the application of precision agriculture technologies (for example, geospatial tools including global positioning systems, geographic information systems, digital landscape information, spatially explicit mathematical models, and computer analyses) to conservation management practices (Dosskey and others, 2005; McConnell and Burger, 2011).

The more-intensive agricultural practices of today (for example, increased pesticide treatments) may reduce the potential values that agricultural habitats once held for grassland birds (Best, 1986; Vickery and others, 1999; Mineau and Whiteside, 2013; Hill and others, 2014). Agricultural areas may be ecological traps, which Best (1986, p. 308) defined as “manmade areas that, on the basis of physical and/or vegetational characteristics, appear to be suitable habitats for nesting but which, by virtue of some confounding factor(s) (for example, brood parasitism, predation, human disturbance), result in population sinks rather than sources for species that settle there.” Avian population trends are linked strongly to changes in agricultural land use. Murphy (2003) determined that a decline in the amount of land managed as rangeland was associated with negative population trends for at least 12 avian species, whereas a decline in the area of land planted to cover crops (that is, land planted to legumes and grasses, which are not harvested or grazed for the purpose of improving soil) was associated with positive trends for 9 of 12 species. Wilcoxen and others (2018) reported higher abundances of grassland birds in corn and soybean fields planted with cover crops between growing seasons than fields without cover crops. Greenwood and others (1995) estimated that for every 10 percent of land area that was converted from grassland to cropland in southern Canada, a corresponding 4-percent decrease in duck (*Anas* spp.) nest success ensued. Conversion of native grasslands to agricultural areas may reduce prey abundance for some grassland raptors, such as the Ferruginous Hawk (*Buteo regalis*), which appears to have declined as a result of conversion (Houston and Bechard, 1984; Schmutz, 1984).

The use of pesticides is widespread in agricultural areas of North America and may have direct and indirect effects on grassland birds (Mineau and Whiteside, 2013). The effects of chemical exposure depend on the type of pesticide used and its concentration during application. For example, in Montana, Chestnut-collared Longspur densities were unaffected by low concentrations of phenylglyoxylonitrile oxime O,O-diethyl phosphorothioate, applied to control grasshoppers (Acrididae), but longspur densities were lower when higher concentrations were used (McEwen and others, 1972). In agricultural habitats in Saskatchewan, Burrowing Owl (*Athene cunicularia hypugaea*) brood size, nest success, and number of young fledged per nest were reduced by exposure to carbofuran but not by exposure to carbaryl (James and Fox, 1987; Fox and others, 1989). Pesticide applications may impact grassland birds by creating a reduction in food resources (Martin and others, 1998, 2000). Disturbances associated with spraying pesticides also may deter birds from using some areas (Rodenhouse and others, 1995). In general, managers should strive to use only rapidly degrading chemicals of low toxicity at the lowest rates possible (McEwen and others, 1972; Sample and Mossman, 1997). As with mowing, spraying of pesticides in CRP grasslands should be delayed until after July to avoid the peak nesting period (Patterson, 1994). Uncultivated areas such as field edges or CRP fields should not be sprayed (Rodenhouse and others, 1993). On grazed pastures, the use of pesticides may be avoided by maintaining range in good condition, because overgrazed and drought-affected areas tend to be more prone to insect outbreaks (McEwen and others, 1972). In contrast to conventional agricultural production, organic farming (that is, agriculture that does not use synthetic chemicals or fertilizers) may benefit some grassland birds (Rodenhouse and others, 1993; Lokemoen and Beiser, 1997; Freemark and Kirk, 2001; Beecher and others, 2002). For example, organic farms may have a higher insect prey base for nesting birds because organic farming does not use the synthetic fertilizers and pesticides that are used during conventional farming (Rodenhouse and others, 1993; Sample and Mossman, 1997). In Nebraska, organically managed corn fields supported more species and higher densities of birds, including several grassland bird species, than did nonorganic corn fields (Beecher and others, 2002). In a southern Ontario study, many bird species were more abundant on organic than conventional farms, but farming practices (tillage, amount of cover, nonharvested habitats) explained the most variance in bird abundance (Freemark and Kirk, 2001). However, organic farms are frequently small and therefore may not provide adequate nesting areas for some grassland birds (Sample and Mossman, 1997). Also, the use of mechanical techniques to control weeds instead of pesticides for controlling weeds may lead to high rates of nest destruction.

Agricultural tillage systems include conventional, minimum tillage, and no till. The latter two practices sometimes are referred to as conservation tillage practices (Best, 1985). Conventional tillage involves turning crop residues into the soil prior to planting, and there may be direct and indirect effects on grassland birds using those fields depending on the timing of the disturbance in relation to the nesting cycle (Best, 1985; Castrale, 1985; Rodenhouse and others, 1993, 1995). Direct effects include disturbance, destruction of nests, and the killing or injuring of incubating females or young (Rodenhouse and others, 1995). Indirect effects include alteration of vegetation structure that may reduce cover or reduce the abundance of litter or foliage-dwelling arthropods (Rodenhouse and others, 1995). The alternative to conventional tillage is to reduce the number of times that a field is tilled, and the options usually include no till (crops are planted directly into crop residues from the previous growing season) and minimum tillage (fields are tilled as little as possible) (Best, 1985).

The principal differences in fields managed with conventional and reduced-tillage practices are the quantity of crop residue, the presence or amount of waste grains, the number of mechanical disturbances associated with machinery, and how weeds are controlled (Best, 1985). Reduced-tillage fields may support greater food resources for grassland birds because fewer arthropods and seeds are plowed under the soil than during conventional tillage operations (Rodenhouse and others, 1995; Sample and Mossman, 1997). The effect of reduced-tillage on nesting birds depends on the timing of tilling operations and the cover type (Rodenhouse and others, 1995; Martin and Forsyth, 2003). For example, in Alberta, grassland sparrows were more abundant or had greater productivity in minimum-tillage fields than in conventionally tilled fields, depending on plant species and cover type (Martin and Forsyth, 2003). Although Horned Lark and Thick-billed Longspur were more abundant in conventionally tilled fields than in minimum-tillage fields, these species had greater productivity in minimum-tillage fields than in conventionally tilled fields for some cover types. Overall, minimum tillage appeared to have positive effects on the grassland bird community using cultivated fields. In a North Dakota study, passerines had higher nesting success in minimum-tillage fields than in conventionally tilled fields when nest loss due to predation was excluded (Lokemoen and Beiser, 1997). Similarly, in Iowa, there were more nesting species and greater nest densities on no-till fields than on tilled fields (Basore and others, 1986), and in Indiana, there were more bird species found in no-till fields than in conventionally tilled fields (Castrale, 1985). However, Best (1986) reviewed literature on bird use of minimum-tillage fields and cautioned that minimum-tillage fields might be an ecological trap wherein birds are attracted to the fields but still experience poor reproductive success because of the tilling and other mechanical disturbances. In addition, higher levels of herbicides may be needed on no-till fields than conventionally tilled fields because of the loss of weed control provided by tilling; increased use of pesticides may harm nesting birds through toxic effects (Best, 1985; Martin and others, 2000; Mineau and Whiteside, 2013). Other approaches, such as ridge till and integrated pest management, might be useful to reduce the need for additional pesticides on reduced-tillage fields (Rodenhouse and others, 1993; Sample and Mossman, 1997). In particular, integrated pest management may help retain nontarget arthropod populations that are an important food source for birds (Rodenhouse and others, 1993).

The timing of agricultural activities such as planting, cultivation, and harvesting has important implications for grassland birds nesting in agricultural habitats (Rodenhouse and others, 1993). Tilling, planting, cultivating, and harvesting may cause mechanical destruction of bird nests, whereas delaying some disturbances (for example, harvesting) may allow more nesting birds to fledge young (Best, 1985; McNicholl, 1988; Lokemoen and Beiser, 1997). Because the timing of harvest depends on latitude and crop type (Rodenhouse and others, 1995), consideration of these factors is important in areas where bird conservation is a priority. Delaying harvesting, avoiding night harvesting, and spacing harvests as far apart as possible may allow grassland birds to successfully nest in agricultural areas (Rodenhouse and others, 1993). Waste grain left in summer-harvested fields may be an important food source for some nesting birds (Rodenhouse and others, 1993), as well as migrants.

Maintaining and Managing Grasslands for Grassland Birds

Given the complexities of short- and long-term effects of management on vegetation and bird populations in grasslands, a universal approach to managing grasslands for the conservation of the entire suite of grassland bird species does not exist. Land or natural-resource managers (this terminology is used broadly for all resource managers, including private land owners) recognize that it will be impossible to manage for all grassland bird species simultaneously, especially on small management units. Management practices or treatments (the terms will be used interchangeably) that may support the habitat needs of one suite of species likely will not meet the habitat requirements of another suite of species. For example, it may be difficult to create habitat that supports species that require tall and dense vegetation while simultaneously supporting species that require short and sparse vegetation. Prairie ecosystems evolved under dynamic forces that created a diverse array, or mosaic, of habitats. The loss or alteration (such as a change in frequency or intensity) of those natural forces, and the accelerated loss of native grassland habitats through anthropogenic activities, means that natural habitat diversity is lost in many grasslands. Increasingly, managers are finding it necessary to prioritize their management efforts toward those bird species or habitats that the manager or management agency ranks highest for a specific region or management unit. For example, a manager might focus their management on one or a few rare species or habitats. Because some grassland bird species are more imperiled than others, additional attention to the species of highest conservation concern might be merited (Herkert and others, 1996). Alternately, management might focus on species that have limited continental breeding ranges but whose core breeding ranges occur within the land manager's jurisdiction. Management also could be based on an agency's preference for providing resources for one or a suite of species (for example, upland-nesting gamebirds or waterfowl), recognizing that other species also might benefit from this single- or few-species management approach. If two or more focal species have contrasting habitat requirements relative to other focal species, management practices may need to be rotated through the landscape to create a mosaic of habitats (Sample and Mossman, 1997; USDA, 1999a, 1999b; Fuhlendorf and Engle, 2001). Regardless of the basis for a prioritization scheme, the act of prioritizing will be just one in a string of necessary but complex decisions. Therefore, a management plan with clearly desired outcomes that can guide decision-making efforts will be beneficial to a manager.

Despite the thousands of studies that have been cited in this compendium on “The Effects of Management Practices on Grassland Birds” to document the habitat requirements or effects of particular management treatments on grassland birds, much remains unknown about the effects of management practices on grassland bird species. Realistically, there is no easy way to obtain a comprehensive understanding of the most effective management options for particular species. In addition, Herkert and others (1996) cautioned that land managers should acknowledge that different management practices might interact to produce unintended consequences. Site-specific experiences and knowledge of the biotic and abiotic environment in an area will prove invaluable to managers as they develop management or conservation plans for their particular management unit. The series of species accounts in this compendium review the current state of knowledge regarding management of grassland bird species in North America. These accounts summarize information on the effects of management practices on individual species. The accounts do not give definitive statements on the effects of management practices for any particular species, primarily because there are very few replicated studies in which identical management practices have been applied in the same geographical area with consistent results, which are elements necessary to provide concrete recommendations for the management of a particular species in a particular area. Documentation of the effects of different management treatments on individual species through statistically sound methods that incorporate multiple years and locations will further scientists’ and land managers’ knowledge far more than 1–2-year studies that are limited in scope as well as time (Grant and others, 2009), but studies of that scope and breadth are rare.

Factors to Consider when Choosing a Management Approach

There are several scales at which conservation measures are initiated, ranging from small-scale (for example, a grassland managed by a single land manager), to regional (for example, management of a biome), to international (for example, range-wide conservation strategies) planning efforts. Managers no longer work in isolation, because regional planning efforts exist for North America (for example, Fitzgerald and others, 1998; Beyersbergen and others, 2004), and indeed, the success of local efforts can be amplified by becoming integrated into larger-scale conservation planning efforts (Sample and Mossman, 1997). Many grassland birds exhibit low levels of philopatry and high levels of opportunism, and therefore focusing on the management of specific areas rather than whole landscapes may not properly protect grassland birds (McNicholl, 1988). Large fluctuations in grassland bird abundance and shifts in their distribution emphasize the importance of large-scale conservation efforts (Sauer and others, 2013). Regional planning and prioritization are important approaches for the conservation of grasslands and grassland birds, especially for those species that have limited breeding ranges (Ryan, 1986, 1990; Sample and Mossman, 1997; Samson and others, 1998; Vickery and others, 1999). Cooperative management across land-ownership and political boundaries with multiple stakeholders may be an efficient means to promote the conservation of grassland birds and habitat diversity (Johnson, 1996; Vickery and others, 2000). Noss and others (1995) and Samson and others (1998) contended that viable populations of individual grassland bird species may best be achieved through ecosystem-level efforts.

Numerous authors have produced management guidelines and recommendations for grassland management that were designed for particular States, Provinces, or ecosystems (for example, Ryan, 1986, 1990; Herkert and others, 1993; Sample and Mossman, 1997; Paige and Ritter, 1999; Gillihan and others, 2001; Prairie Conservation Action Plan, 2014). Several plans have been developed at national and international levels, including the North American Landbird Conservation Plan (Rosenberg and others, 2016), the North American Waterbird Conservation Plan (Kushlan and others, 2002), and the North American Waterfowl Management Plan (North American Waterfowl Management Plan, 2012). The goal of this compendium is not to repeat these expansive efforts, but rather to focus on the major topics that will serve to inform management decisions and conservation actions.

The extreme climatic fluctuations characteristic of the Great Plains and the historical relationships between climate, fire, and grazing created considerable annual variation in vegetation composition and structure, thus creating mosaics of habitat at various stages of recovery and succession (Bragg and Steuter, 1996; Fuhlendorf and Engle, 2001). This inherent unpredictability to the grassland ecosystem also contributes to large annual and regional fluctuations in distribution and abundance that grassland birds often exhibit (Cody, 1985; Zimmerman, 1992, 1997; Igl and Johnson, 1999; Winter and others, 2005a, 2005b). Although several researchers have determined relationships between bird abundance and such climate variables as precipitation, temperature, the Palmer Drought Severity Index, and number of wetlands containing water (Ahlering and others, 2009; Grant and others, 2010; Gorzo and others, 2016; Niemuth and others, 2017), the biological meaning of climate variables is unclear, and they are likely correlates of other factors (for example, plant community composition, primary and secondary productivity) that more directly influence species occurrence in concert with other factors such as soils and landform (Niemuth and others, 2008; Niemuth and others, 2017). Climatic conditions and vegetation disturbances may alter not only the vegetation community but also the bird community composition; therefore, consideration by land managers of more than short-term responses to management treatments is warranted in making management decisions.

The context of individual grasslands (that is, the management unit) under management consideration, both within the range of individual bird species and within the landscape in which the unit is embedded, is an important consideration for land managers. Does a focal species breed locally or regionally? Grassland birds frequently are observed outside their breeding ranges as indicated in field guides and planning documents, but it may be ineffective to manage habitat at a site for a species that rarely occurs in a region. Is the management unit part of a larger, contiguous expanse of grassland, or is the management unit isolated or embedded within a largely wooded or agricultural landscape? The landscape context may help predict which species find the management unit suitable. For example, it may not be prudent to manage a small and isolated grassland surrounded by forest for bird species that require large areas of open grassland or that are adversely affected by forested edges.

Other factors that influence the effectiveness of a management approach are regional differences in grassland types (for example, dominance of warm-season or cool-season grasses), grassland health (that is, degree of degradation and level of biotic diversity), microclimate, and soil type and health. Mycorrhizal fungi often are an overlooked component of grassland health and management. Research by Eom and others (1999) has shown that the effects of management practices on aboveground plant communities are likely mediated, in part, through concomitant effects on mycorrhizal fungi and belowground processes. Arbuscular mycorrhizal fungi influence the growth, demography, competitive relationships, relative abundances, and diversity of plants in grassland communities (Eom and others, 1999; Hartnett and Wilson, 1999). Grassland management practices, such as burning, mowing, and fertilization, may influence the abundance and species diversity of mycorrhizal fungi and the development of symbiosis with prairie plants. An understanding of how different environmental factors and management practices influence arbuscular mycorrhizal fungal populations is important because the effect of fungi on prairie plants varies greatly, ranging from mutualistic to neutral to pathogenic (Eom and others, 1999).

The previous and current land uses of a management unit also warrant consideration during development of a management plan. Grassland management for the conservation of grassland birds may include ongoing maintenance of extant or degraded native grasslands, restoration of native grasslands that had been converted to another use (for example, agricultural production), and the creation of human-constructed grasslands from some other land use (for example, reversion of cropland to a grassland enrolled in the CRP). Emulating the historical natural disturbances that formed the grassland unit, which most likely resulted in a mosaic of habitats and vegetation structure, is warranted in management of native grasslands for grassland birds. Ryan (1990) advocated that managers experiment with the combinations of prescribed burning, grazing, mowing, and application of herbicides at different sites with varying soil moisture conditions to maintain the array of habitats required to preserve the biotic diversity of the prairie ecosystem.

A complicating factor with management of native grasslands is that many are highly degraded owing to invasion of non-native plant species, alteration of natural disturbance regimes, and encroachment by woody vegetation. Floristic inventories conducted by Murphy and Grant (2005) and Grant and others (2009) on Federal grasslands in North Dakota and South Dakota revealed that all prairies that they inventoried were moderately to severely degraded, mainly by invasion by smooth brome and Kentucky bluegrass (*Poa pratensis*), but also by woody encroachment. Wetlands, too, are commonly degraded by invasive wetland plants such as Russian olive (*Elaeagnus angustifolia*), purple loosestrife (*Lythrum salicaria*), and narrowleaf cattail (*Typha angustifolia*) (Whitt and others, 1999; Kantrud, 1992; Knopf, 1994; Maddox and Wiedenmann, 2005). The invasion of native habitats by non-native species may simplify ecosystems by reducing forb and grass species richness and arthropod abundance and by outcompeting native vegetation (Wilson and Belcher, 1989; Sutter and Brigham, 1998; Dugger and Dugger, 2002; Flanders and others, 2006; Spyreas and others, 2010). Invasive plants also alter bird communities in detrimental ways, including reductions in bird abundance, species richness, species diversity, nest density, and measures of reproductive success (Sutter and Brigham, 1998; Scheiman and others, 2003; Lloyd and Martin, 2005; Maddox and Wiedenmann, 2005; Flanders and others, 2006; Davis, 2017). Invasive plants also can create habitat conditions that are favorable for less-desirable species, such as the Brown-headed Cowbird (*Molothrus ater*), Red-winged Blackbird (*Agelaius phoeniceus*), and Yellow-headed Blackbird (*Xanthocephalus xanthocephalus*), at the expense of more-desirable species (Naugle and others, 1999; May and others, 2002; Flanders and others, 2006).

The loss of native grazers, the suppression of wildfires, and the planting of trees have led to an increase in the cover of woody vegetation on the landscape. The encroachment or intentional planting of woody vegetation reduces grassland habitat available to grassland birds (Johnson, 1996). The amount of tree cover in the landscape also influences grassland birds by influencing the movements and spatial patterns of predators and brood parasites (Knopf, 1986; McNicholl, 1988; Johnson and Temple, 1990; Wellicome and Haug, 1995; Igl and Johnson, 1997; Naugle and others, 1999; O'Leary and Nyberg, 2000; Winter and others, 2000; Coppedge and others, 2001; Ribic and Sample, 2001). Although some grassland bird species may tolerate woody encroachment, other species may have a threshold at which increased levels of encroaching woody vegetation are no longer tolerated (Herkert and others, 1996; Grant and others, 2004a). Exotic trees, such as Russian olive, may invade prairie stream courses, allowing the influx into grasslands of woodland birds and creating a favorable environment for the Brown-headed Cowbird, an obligate brood parasite (Knopf, 1988, 1994). The loss of historical patterns in grazing and burning has led to increased numbers of wetlands that are partially or completely surrounded by trees (Naugle and others, 1999). Naugle and others (1999) determined that bird species richness declined as the extent of woody vegetation along wetland perimeters increased. Declines in species richness were most marked when woody vegetation encompassed greater than 75 percent of the wetland perimeter. Those bird species that did benefit from increased woody vegetation were species adapted to edge habitats, rather than grassland or wetland specialists. Cunningham and Johnson (2006) reported that tree cover negatively influenced densities of several wetland-dependent bird species.

Restoring degraded native grasslands and wetlands, and then maintaining them after restoration, will require an improved understanding of the factors that have contributed to the ecosystem degradation and the factors necessary for restoring the health of the community (Grant and others, 2009). A process-oriented, adaptive management approach could be used to make these and other management decisions. Using this adaptive management approach requires a long-term evaluation (that is, a commitment beyond a few years) of the prospective strategies aimed at restoring the grassland (for example, reducing non-native plants) (Grant and others, 2009). Such an approach aims to resolve the uncertainties inherent in making management decisions by adopting a transparent and structured decision-making process that reduces *management paralysis* (that is, the inability to move beyond the longstanding or traditional techniques that have not succeeded because of an overwhelming uncertainty of or uneasiness about novel management techniques; Gannon and others, 2013). The approach requires formulating an objective, quantifiable statement of a desired outcome; an experimental design with randomization, treatment and control sites, and replication; a set of decision alternatives; competing, predictive models of decision outcomes; and an inventory and monitoring program, such as that presented in Grant and others (2004b).

Restoration

Restoration can be a confusing term. For example, how does restoring a native prairie that has been converted to another land use (for example, to agricultural production) differ from restoring a degraded prairie or creating a grassland where none existed previously? Munro (2006) suggested that ecological restoration, at a minimum, entails the use of native plant species in an ecological community setting; recontouring of land to original site conditions; emulation of historical reference sites; and use of local, natural materials for hardscaping. For more information on ecological restoration, see Society for Ecological Restoration International (2004) and Clewell and others (2005).

Several studies have determined that grassland birds respond favorably to restored or newly created grasslands (for example, Askins, 1993; Fletcher and Koford, 2002). Degraded grasslands, native and human-created, may benefit from the planting of desired grass and forb species (Sample and Mossman, 1997) or modifying the disturbance regime such that it mimics or resembles historical conditions. Following the principles of ecological restoration (Munro, 2006), using a diverse array of locally derived native plants rather than non-native seeds is preferred (Herkert and others, 2003; Munro, 2006). In preparing a seedbed for grassland restoration,

The restoration of grassland for the purposes of benefitting wildlife species can include the seeding of former cropland to a multi-species array of grasses and forbs or restoring degraded native prairie by removing invasive species so that native grasses and forbs can flourish. *A*, Seeder; photograph by U.S. Fish and Wildlife Service. *B*, Native prairie restoration; photograph by Tony Ifland, U.S. Fish and Wildlife Service.

application of herbicides may be needed to remove exotic or weed species prior to seeding. Other steps also may be necessary and beneficial, such as consulting with land managers within the same region. Land managers should note that ecological restoration may be impractical in some situations (Munro, 2006), such as at large scales (Johnson, 1996).

Soil enhancers (for example, native mycorrhizal fungi and other soil organisms) that were lost during degradation may be used to enhance restoration efforts (K.A. Smith, retired, FWS, Kenmare, North Dakota, written commun. [n.d.]). Many inactive surface mines have been reclaimed or planted to grassland areas (Brothers, 1990). Soil acidity after coal removal makes the development of grassland difficult, but with time, grass coverage may improve and grassland birds may colonize areas (Whitmore and Hall, 1978).

Regardless of whether a land manager is dealing with pristine, degraded, or created grasslands, the following management tools or practices can be used to some degree. That degree may be resolved using an adaptive management approach.

Management Tools for Grasslands

Many management practices and tools are available to resource managers, depending on their desired outcomes and objectives. The primary tools available for grassland management are burning, grazing, mowing, herbicide application, and idling. As mentioned earlier, resource managers may strive to incorporate into management plans the historical natural disturbances (for example, fire, grazing) that once maintained grasslands. Mowing may be used to produce similar outcomes.

Burning, grazing, and mowing are all disturbances that reduce vegetation. Thus, these practices have somewhat similar immediate effects on vegetation structure: reduced vegetation height and biomass. These practices also may be used to suppress or eliminate some non-native plant species or woody vegetation. Burning and mowing are less selective in plant removal than is grazing in that grazing animals may select some plant species over others. Grazing may result in a more heterogeneous vegetation structure than either mowing or burning because of the uneven grazing patterns of livestock (Sample and Mossman, 1997). Burning, grazing, and mowing affect nutrient cycling differently. Burning returns some plant nutrients to the soil in the form of ash and usually increases nutrient cycling; properly timed grazing can stimulate nutrient cycling and returns some nutrients to the soil in the form of animal waste; and mowing returns few plant nutrients to the soil (Anderson, 1982), although properly timed mowing also can stimulate nutrient cycling.

The goal of this report is not to provide specific recommendations regarding management of grassland birds by using specific management practices (such as recommending a specific mowing period [for example, after July 15] within a breeding season to reduce nest destruction); those recommendations are beyond the scope of this publication and often

Management practices that simulate historical natural forces include *A*, prescribed burns (photograph by Jennifer Jewett, U.S. Fish and Wildlife Service); *B*, haying (photograph by Rick Bohn, used with permission); and *C*, grazing by domestic livestock (photograph by Neil Shook, U.S. Fish and Wildlife Service).

are site or species specific. Management recommendations from the literature are summarized in the individual species accounts that constitute this compendium. General management recommendations for grasslands birds, with a more in-depth discussion of management tools covering many broad topics in detail, can be found in Sample and Mossman (1997).

Seasonality, Intensity, and Frequency

Before choosing a particular management practice, a manager will want to consider issues of seasonality, intensity, and frequency. Seasonality refers to *when* a management treatment is applied. For example, disturbances associated with prescribed burns and mowing often are deleterious to grassland birds and their nests during the breeding season, and thus many management plans recommend limiting disturbances to periods before (early spring) or after (late summer or fall) the peak breeding period of nesting birds to avoid harming adults or their nests and young. Because bird species vary in their nesting phenology, management activities that are timed to favor one species may harm another species (Winter and others, 2004). The seasonality of grazing regimes also may influence breeding bird communities, either directly (for example, cattle trampling nests) or indirectly (for example, changes in vegetation relative to the timing of grazing). For example, Wiens (1970) determined that breeding Horned Larks preferred sites that had been heavily grazed during the winter more than sites that had been heavily grazed during the summer, but the reverse was true for Thick-billed Longspurs.

Because most management practices in grasslands inevitably revolve around manipulation of vegetation structure, it is important to understand the phenology of specific plant species and their responses to disturbances (Smith, 2005). It may be important to time a disturbance during a particular life stage of a preferred or undesirable plant species to achieve a desired management effect (Manske, 1995). For example, some undesirable plant species (for example, non-native or invasive species) may be vulnerable during early growth stages or when their root reserves are lowest, making those important periods for disturbances (such as prescribed fires) to reduce, eliminate, or weaken a particular species (Smith, 2005). Burning when root reserves are high may result in increased vigor in that plant species. Similar concerns and considerations can be applied to preferred plant species.

Another consideration in relation to seasonality is the type of management treatment. Different management treatments may have different effects on a plant species within the same management unit, and these effects may vary depending on the plant's life cycle or growth (Risser and others, 1981). Sample and Mossman (1997) provided examples of how the seasonality of burning, grazing, and mowing impact plant species composition. For example, spring burns may affect plant species composition differently than fall burns; spring burns tend to suppress cool-season grasses and promote warm-season grasses, whereas the opposite is true of mid- to late-summer burns. Mid-summer mowing or burning of native warm-season grasses tend to suppress warm-season grasses but maintain native forbs and cool-season grasses. Other native forbs are suppressed by mid-summer mowing but flourish after mowing or burning in early spring or late fall. In Wisconsin, Sample and Mossman (1997) recommended that grazing should be discontinued by early August when managing for warm-season grasses and by mid-September when managing for cool-season grasses. Thus, resource managers would need to time their selected management practice such that the treatment promotes desirable vegetation structure and composition and benefits grassland bird species of interest. Also, it is important to note that terminology used in the literature often varies considerably. For example, terms that refer to the timing of disturbances, such as *spring* and *fall*, are subjective, and their definitions vary among studies and locations. Local or regional phenological events, both for plant and animal species, will dictate the appropriate timing of management practices.

The timing, or seasonality, of when a management practice is applied affects vegetation composition and wildlife differently. For example, prescribed burns applied in spring may harm nesting birds but be most effective at suppressing the spread of invasive plant species by damaging plants during a vulnerable growth stage. Photographs of *A*, spring and *B*, summer prescribed burns by Jennifer Jewett, U.S. Fish and Wildlife Service.

Intensity refers to the degree to which a management tool is applied. For fires, Pyne and others (1996, p. 11) defined intensity as "the amount of heat produced per unit of fuel consumed per unit time." Some fires burn incompletely and leave some vegetation unconsumed, whereas other fires reduce most or all vegetation to ashes. Completeness and intensity of prescribed fires may influence post-burn vegetation and concomitantly how birds respond to post-burn habitats (Ryan, 1986). For example, in southeastern Idaho, partial burns of sagebrush habitats reduced Brewer's Sparrow (*Spizella breweri*) numbers less than complete burns (Petersen and Best, 1987). Grazing intensity can be determined by the number of grazing animals and length of time that they are allowed to graze a management unit, or the percent utilization of available forage (Kantrud and Kologiski, 1982; Bleho and others, 2014). Sometimes these terms are defined in terms of the stocking rate, or number of livestock (for example, number of cow/calf pairs), and the duration of the grazing period on a given area, such as the number of animal unit months per hectare. In other cases, the terms are defined by the density and height (or combination of the two) of the vegetation and the litter that remains after livestock are removed. It is important to be aware that the use of terms related to grazing intensity, such as *lightly*, *moderately*, and *heavily* grazed, are pervasive in the literature but may be highly subjective terms. Objective measures of grazing intensity are necessary to make comparisons among studies and regions. Vague or subjective management recommendations (for example, lightly graze a pasture to benefit a particular species) often are of little practical use to a land manager. Information on vegetation and habitat needs, however, are common in the literature. In each species account that constitute this compendium, the authors provide a capsule statement that summarizes such information from the scientific literature, including measured vegetation variables from published studies throughout a species range. For example, if managing for a wide-ranging grassland bird that requires short and sparse vegetation, a land manager in tallgrass prairie may need to ensure that a grassland patch is more *heavily* grazed to achieve the same vegetation structure as shortgrass prairie that is *lightly* grazed. The necessary level

The intensity with which a management practice is applied affects vegetation composition and wildlife. A grassland grazed by large numbers of cattle or over the entire summer will have less wildlife cover than a grassland grazed by fewer cattle or grazed on a rotational basis. Some bird species prefer heavy grazing, whereas other species prefer light grazing. The photograph shows the same grassland in Kidder County, North Dakota, with heavier grazing on the left side of the fence than on the right side; photograph by Rick Bohn, used with permission.

of grazing intensity to obtain a desired vegetation structure will depend on a region's precipitation in any given year (Sliwinski and Koper, 2015).

Frequency refers to how often management tools have been applied, either within or among seasons. For example, agricultural producers in one region (for example, the Flint Hills) might prefer to burn annually to rejuvenate grassland vegetation for livestock production, whereas a resource manager might prefer to burn every 2–5 years to improve conditions for grassland-nesting birds. Madden (1996) suggested that fire-treatment intervals in grasslands should approximate historical fire-return intervals to benefit nesting birds. Longer burning intervals allow more woody plant regrowth and encroachment and greater litter accumulation than shorter burning intervals, so a determination of the burning interval should depend on the desired structural conditions and plant species composition (Sample and Mossman, 1997). The number of consecutive years that a unit has been burned, grazed, or mowed is important, because the effects of vegetation removal can be cumulative across years (Johnson and others, 2011b; Sliwinski and Koper, 2015). Allowing a management unit to remain idle for too many years, or conversely, repeatedly applying burning, mowing, or grazing to the same management unit, may result in conversion of the vegetation structure and composition to an undesirable state. Smith (2005) contended that land managers must be willing to commit to a management plan; desired changes may not be immediate but may in fact take repeated applications, and the timing between those applications is critical.

Burning, Grazing, and Mowing

In addition to stimulating nutrient cycling, prescribed fire is an effective management tool for reducing or eliminating vegetation biomass and litter, reducing woody plant encroachment, and stimulating production of herbaceous species (Ryan, 1986; Sample and Mossman, 1997). Whether bird species respond to vegetation changes associated with prescribed burning depends on the bird species, degradation of the grassland prior to burning, seasonal timing of the burn, and how often burns are applied (Herkert and others, 1996; Johnson, 1996). For grassland birds, burns conducted outside of the breeding season typically are recommended so that nests are not destroyed and vegetation has time to recover for the nesting season (Higgins, 1986; Herkert and others, 1993; Sample and Mossman, 1997). Burning just prior to the breeding season may delay use by birds of the burned field; for example, in a Wisconsin grassland that had been burned in April, Bobolinks did not occupy the field until early June of that same year; during a year when the field was not burned, Bobolinks took up residency in May (Martin, 1971). Annual burns of grasslands likely will be detrimental to some species; for example, in Kansas, Zimmerman (1997) determined that Henslow's Sparrows were absent on annually burned tallgrass prairies. In contrast, Michaels (1997) determined that the species was more common on areas that were burned two to three growing seasons previously than on areas burned less than two or more than four growing seasons previously. Also of note is that short-term changes may differ from long-term effects. For example, prescribed burning may increase the forb component of Greater Sage-Grouse (*Centrocercus urophasianus*) diets at the expense of long-term habitat suitability (Wrobleski and Kauffman, 2003). Many grasslands are subjected to the combination of burning and grazing. As Richardson and others (2014) noted, the effects of this combination of management practices are greater than the effects of a single disturbance, and thus have merited numerous studies that are discussed later in the section.

Grazing is a valuable management tool that can be used to reduce vegetation biomass, litter, and undesirable woody and herbaceous vegetation; increase plant species diversity; stimulate soil nutrient cycling; and reduce nest-predator abundance and efficiency (Sample, 1989; Hartnett and others, 1997; Sample and Mossman, 1997; Murphy and Grant, 2005; Bleho and others, 2014). Familiarity with the behaviors and foraging preferences of domestic livestock breeds and native species of grazers is beneficial because grazers differ in their grazing pressures (Peden and others, 1974; Schwartz and Ellis, 1981; Plumb and Dodd, 1993; Hartnett and others, 1997). Most studies evaluating the impact of grazing on grassland birds have evaluated domestic livestock, especially cattle, because they are the most common grazer in native prairies (Willms and Jefferson, 1993). Koper and Schmiegelow (2006), Lusk and Koper (2013), and Pipher and others (2016) determined that cattle grazing had little effect on grassland-bird nest survival in Canada, whereas Kerns and others (2010) determined positive and negative effects in North Dakota. Effects of grazing on grassland bird nest survival are likely confounded by environmental conditions such as precipitation, and thus, consistent, year-to-year results may be rare. Pipher and others (2016) suggest that cattle grazing over a range of intensities as applied in Canada is compatible with the conservation of many species of grassland birds. Nest losses owing to trampling by livestock may be a problem in some areas or at high stocking rates, but not in all areas (Sugden, 1933; Jensen and others, 1990). In Canada, Bleho and others (2014) determined that nest predation was the biggest reason for nest failures, not destruction by cattle.

There are several types of grazing systems currently available to resource managers. Although we give a broad overview of the major grazing systems below, it is important to recognize that, even within the same grazing systems, there are subtle to major differences in how the treatments are applied. Season-long or continuous grazing is a grazing system whereby livestock graze one pasture throughout the growing season (or year), without being moved to another area (Messmer, 1990; Sedivec, 1994). Rotational grazing and short-duration grazing occur when livestock are rotated through a series of pastures throughout a year's growing period, allowing vegetation in formerly grazed areas to grow in the absence of grazing pressure for a period of time (Messmer, 1990; Sedivec, 1994; Briske and others, 2008). Twice-over grazing is one common approach to rotational grazing, in which pastures are divided into at least two units and livestock are moved through each unit twice during the grazing season, allowing at least 30 days without grazing before a unit is grazed again (Messmer, 1990; Sedivec, 1994; Schneider, 1998). Including additional pastures in the rotation allows pastures 40–45 days or more of rest (that is, idle conditions) before the second grazing period. Ranellucci and others (2012) provide a more thorough description of grazing systems than can be described here. In finding no consistent or overwhelming benefit of rotational grazing over season-long grazing in their study in Canada, Ranellucci and others (2012) concluded that implementing any of a number of grazing systems may be just as beneficial to grassland birds as advocating for one system over another.

There are numerous complexities in choosing a grazing management system. These complexities were recognized by Briske and others (2008, p. 4) in the following statement: "the absence of consistent management and policy recommendations concerning the adoption of grazing systems after several decades of experimental research and commercial application is a testament to the complexity of this task." Briske and others (2008) compared stocking rates and intervals of rest and grazing for deferred rotation, rest rotation, high-intensity/low-frequency, and short-duration grazing systems. The authors enumerated the variables that make comparison between grazing systems difficult; these variables included ecological variation associated with rainfall regime (that is, amount, seasonality, and intra- and interannual variability), vegetation structure and composition, productivity, soil hydrological characteristics, prior land use, and livestock characteristics (that is, breeds, prior conditioning, care, and handling). Other variables that the authors considered included commitment, ability, goals, opportunities, and land ownership of the managers. The timing (for example, early, continuous, late in the growing season) of grazing also may lead to a variety of changes in vegetation structure and, therefore, to different impacts on grassland birds (Prescott and Wagner, 1996). Despite this overwhelming list of potentially confounding variables, stocking rate emerged as the most consistent management variable that influenced the grazing plan and animal responses to grazing (Briske and others, 2008).

Derner and others (2009) advocate for the utilization of livestock as ecosystem engineers. The manipulation of livestock grazing behavior can be used to create the vegetation structure desired by managers of grassland birds. The concentration of grazing livestock can be manipulated through the careful siting of supplemental feed, water, and the burning of particular patches of pasture. Such use of livestock, however, may require more investments of time than traditional practices of season-long grazing with no rotation among management units. Repeated applications of grazing to a management unit will affect bird species in different ways. Sliwinski and Koper (2015) determined a gradual decline in Baird's Sparrow and Savannah Sparrow abundance with repeated grazing at the highest stocking rates evaluated; noticeable declines in vegetation biomass attributed to livestock grazing also were apparent. Conversely, the abundance of species such as Chestnut-collared Longspur increased at high stocking rates (Sliwinski and Koper, 2015). For future management, the first two species might benefit from low stocking rates or exclusion of grazing, whereas the other species might benefit from higher stocking rates.

In areas like the Flint Hills of Kansas and Oklahoma, a combination of annual, dormant-season burning and a short, intensive grazing period has been used to maximize livestock production at the expense of native plant and animal diversity (Fuhlendorf and others, 2006; Powell, 2006, 2008). With and others (2008) predicted that the continued application of this particular combination of burning and grazing in the Flint Hills would cause the regional populations of Eastern Meadowlark, Grasshopper Sparrow, and Dickcissel to become inviable, a prediction that, 10 years later, could be checked against annual indices of population trends from sources such as the North American Breeding Bird Survey (Pardieck and others, 2018). A combination of management practices makes it difficult for researchers to isolate the effects of grazing from the effects of burning (Rohrbaugh and others, 1999). Brudvig and others (2007) evaluated the effects of combinations of fire and grazing treatments on plant species diversity, life form, and individual plant species and determined that, in general, individual management goals could be met by a specific treatment, but no single treatment satisfied all management goals. Fuhlendorf and others (2006) thus advocate for mimicking the historical fire-grazing interaction under which native prairies evolved by applying fire to discrete patches and allowing grazing animals to select among burned and unburned patches (what they term "patch-burn grazing"). In this way, a more-natural spatial heterogeneity of vegetation structure is created that meets the habitat needs of the grassland bird community in the region (Coppedge and others, 2008; Hovick and others, 2015), while still maintaining livestock production at levels similar to traditional management approaches (Fuhlendorf and Engle, 2004). Churchwell and others (2008) determined that the nest success of Dickcissels was higher, and parasitism and predation were lower, in patch-burned pastures than traditional pastures. Hovick and others (2015) suggested that grassland bird diversity in the southern Great Plains can be maximized with a 3–4-year fire-return interval using the patch-burn grazing approach, a time interval supported by Powell and Busby (2013) for grasslands on the western edge of the tallgrass prairie ecosystem. Application of the patch-burn grazing approach has been of limited success in other regions for fulfilling management goals. Whereas Duchardt and others (2016) reported increased avian diversity in small grasslands in Iowa and Missouri, Hovick and others (2012) reported no clear differences in Grasshopper Sparrow clutch size and nest survival and between the patch-burn approach and a more traditional burn-and-graze approach.

Mowing and haying reduce vegetation height, litter (particularly if hayed vegetation is removed), and woody encroachment (Herkert and others, 1996; Sample and Mossman, 1997). However, mowing and haying conducted during the breeding season may have substantial negative impacts on grassland-nesting birds by reducing availability of invertebrates used to feed nestlings, destroying active nests, and killing recently fledged young (Bollinger and others, 1990; Zalik and Strong, 2008). Hayfields usually are cut one to four times per growing season (Rodenhouse and others, 1995). If conducted multiple times during the breeding season, mowing or haying may prevent birds from successfully nesting for that year (Frawley, 1989; Bollinger and others, 1990; Sample, 1989; Herkert and others, 1996). Although the interval between cuttings may be important for other aspects of land management such as the control of invasive plant species, increasing the number of harvests in hay fields decreases the time available for birds to complete a nesting cycle. Even species that are attracted to the short vegetation created by mowing may have a difficult time successfully nesting because of a short mowing interval (Rodenhouse and others, 1995).

The timing of haying within a season may affect nest survival and success. Currently, earlier-maturing hay varieties often are cut earlier in the growing season than hay fields in the past that were seeded to later-maturing hay varieties, increasing the danger to some grassland birds and their nests but, perhaps in some cases, favoring late-nesting species (Warner and Etter, 1989; Rodenhouse and others, 1995; Herkert and others, 1996; Herkert, 1997a). In general and to the extent possible, mowing should be delayed until after birds finish nesting (that is, after the peak nesting period, generally no earlier than mid-July but preferably closer to late August, especially in the north) (Bollinger and others, 1990; Bryan and Best, 1994; Herkert and others, 1996; Sample and Mossman, 1997; Nocera and others, 2005; Perlut and others, 2006, 2008a, 2008b). Fields hayed later in the breeding season are more beneficial to grassland birds, whereas early hayed fields may be population sinks; for example, in New York and Vermont, Savannah Sparrows using late-hayed fields (hayed after August 1) had a greater than 25 percent higher adult apparent survival than those on the more intensively managed early and middle-hayed fields (Perlut and others, 2008a). Late-hayed fields provided high-quality habitat in which Savannah Sparrows produced more offspring and adults survived longer; high adult survival resulted in stable or near-stable populations in late-hayed fields. Native prairie that is hayed in the Kansas Flint Hills is often mowed late, and so acts more like a

When applied after the peak nesting season for bird species, haying is a valuable management tool for reducing vegetation height and residual cover; photograph by Rick Bohn, used with permission.

"rested" prairie than a hayed prairie; nest success for Dickcissels and Grasshopper Sparrows was 2–4.5 times higher and brood parasitism 3.5–7 times lower in hayfields than in other managed grasslands (Rahmig and others, 2009). In contrast, planted grasslands used for hay in Saskatchewan are likely population sinks (Davis and others, 2016; Davis, 2017). The timing of mowing within a season also may influence plant species composition, with summer cuts favoring cool-season grasses and some native forbs and suppressing warm-season grasses (Sample and Mossman, 1997).

Some bird species may continue to nest in hay fields or may recolonize hayfields after cutting (Shustack and others, 2010). For example, in Michigan, Grasshopper Sparrows continued nesting in an alfalfa field mowed in late June but stopped nesting after a second mowing in early August (Harrison, 1974). Mowing at night may have additional negative effects on breeding birds than mowing during daylight hours because mowing has the potential to injure or kill night-roosting birds as well as nesting birds and their young (Frawley, 1989; Rodenhouse and others, 1995). Additional harvest activity conducted after mowing, such as raking and baling, may destroy additional nests that were not destroyed during mowing (Bollinger and others, 1990). Ground nests are more likely to survive haying than aboveground nests (Frawley, 1989). As with grazing, the frequency of haying (that is, the number of years between haying applications) should depend on local precipitation conditions (Davis and others, 2017). Grassland birds in mesic environments or during years of above-average precipitation may benefit from frequent haying, but frequent haying in arid environments or during drought years may be detrimental to grassland bird species (Madden and others, 2000).

Several haying systems and mowers are available to managers. Haying systems include conventional, seed harvesting, and high mowing; seed-harvesting and high-mowing systems may provide reduced nest destruction and taller post-disturbance vegetation. The type of mower (for example, sickle mower, mower conditioner or windrower, and self-propelled swather) will not only affect management but also the level of nest destruction and wildlife mortality. A pattern of haying, such as mowing from inside a field to the outside of the field, or partially haying a field, may benefit grassland birds because this pattern allows adult birds and their young to escape the patch as it is being cut (Sample and Mossman, 1997; USDA, 1999a, 1999b).

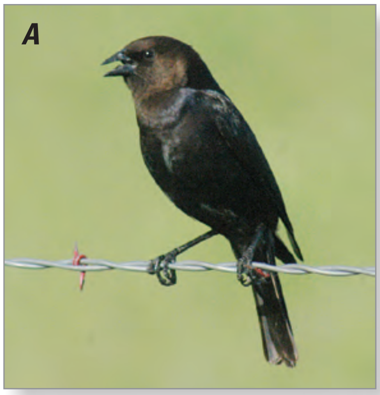
Idling refers to the practice of allowing grasslands a rest from treatments, because complete or even partial removal of vegetation on an annual basis may have an adverse effect on upland-nesting birds (Kirsch and others, 1978). The presence of residual vegetation and litter during the spring and summer are important variables during habitat and nest-site selection for many grassland bird species. Therefore, periods of rest are necessary to allow for adequate vegetative regrowth and accumulation of litter and residual cover. Idling grasslands during the nesting season also benefits species because nests will be less vulnerable to destruction from management applications. Providing a mosaic of idle and managed grasslands will ensure that some residual vegetation is available for those species that require it, especially if adjacent patches had been burned, mowed, or hayed, or received other management treatments (Sample and Mossman, 1997).

In addition to burning, grazing, and mowing, undesirable woody and herbaceous species may be reduced or eliminated using manual removal, herbicides, or mechanical methods (for example, chaining, roller chopping, and disking). Different management practices can create distinct differences in vegetation characteristics; Niemuth and Boyce (1998) determined that although prescribed burning, crown fires, and clearcutting all combatted succession in Wisconsin pine barrens, the vegetation cover, structure, and diversity of woody vegetation differed among practices. Chaining has been suggested as an appropriate tool for reducing woody vegetation, such as juniper invasion in the southern Great Plains (Coppedge and others, 2001). In Florida prairies, woody vegetation was reduced for a longer period of time with roller chopping than with prescribed burning (Fitzgerald and Tanner, 1992). Bird species richness and abundance were lower in roller-chopped plots than in burned plots, regardless of season of treatment, and summer-chopped plots were devoid of birds for up to 5 months (Fitzgerald and Tanner, 1992). Disking may reduce vegetation height and density without removing biomass from the plot (USDA, 1999a, 1999b), but it has the potential for destroying bird nests if done during the breeding season.

Water-level manipulation may be used to enhance wet meadows for grassland and sedge-meadow birds (Sample and Mossman, 1997). Raising the water table or flooding an area can allow for the restoration of sedge meadows or emergent marshes (Mossman and Sample, 1990).

Other Management Concerns

The Brown-headed Cowbird is an obligate brood parasite that commonly parasitizes nests of many North American grassland birds (Shaffer and others, 2019a). The species evolved in the Great Plains, where it associated with herds of grazing bison. Its breeding range and abundance increased during the 20th century owing to increases in habitat fragmentation, livestock production, and agriculture (Johnsgard, 2001). Rates of cowbird parasitism in grasslands vary (Shaffer and others, 2019a), but are strongly tied to the abundance of cowbirds (Herkert and others, 2003; Igl and Johnson, 2007); cowbird abundance, in turn, is positively correlated with the abundance and diversity of the breeding bird community (Igl and Johnson, 2007). Brown-headed Cowbirds are associated with livestock, which likely flush arthropods that cowbirds then consume (Goguen and Mathews, 2001). The species' association with livestock also may reflect higher insect abundance or lower vegetation height associated with grazing (Goguen and Mathews, 1999, 2001). In addition to areas with livestock, cowbirds are attracted to waste grains in crop fields, possibly leading to increased brood parasitism in agricultural



The Brown-headed Cowbird (*Molothrus ater*) is an obligate brood parasite that commonly parasitizes the nests of many North American grassland birds. *A*, Male and *B*, female cowbird photographs by David Lambeth, used with permission. *C*, A parasitized Clay-colored Sparrow (*Spizella pallida*) nest with two blue sparrow eggs and three cowbird eggs; photograph by Lawrence D. Igl, U.S. Geological Survey.

areas (Rodenhouse and others, 1995). Cowbird parasitism often is higher at nests located near woodland areas than at nests located away from woodland areas (Berger, 1951; Best, 1978; Johnson and Temple, 1990). The keys to discouraging cowbird parasitism or limiting populations of Brown-headed Cowbirds in grassland habitats in the Great Plains are maintaining large expanses of grassland, eliminating foraging areas (for example, feedlots) and perch sites, and reducing the extent of overgrazed pastures (Shaffer and others, 2003). However, cowbirds may travel several kilometers from foraging areas to breeding areas (Goguen and Mathews, 2001), and cowbird parasitism of grassland birds in some areas may be lower in landscapes with more trees (Pietz and others, 2009).

Resource managers are increasingly dealing with the effects of anthropogenic activity in grassland landscapes. Those effects are likely to increase as the North American human population grows; the Pew Research Center estimates that the United States will have around 438 million people by 2050 (Passel and Cohn, 2008). Total urban area has more than doubled in the United States during the last 40 years, from 10 million ha to 23 million ha (Trauger and others, 2003). Increasing encroachment of urban areas will negatively impact grassland birds through direct loss of habitat and such indirect impacts as noise and changes to the plant and predator communities (Haire and others, 2000; Lenth and others, 2006; Marra and Santella, 2016). Urbanization can reduce densities of grassland birds (Lenth and others, 2006; McLaughlin and others, 2014) as well as lower nest density (Lenth and others, 2006). Species such as the Greater Sage-Grouse are very intolerant of human activities such that the species seldom locates leks within 5 kilometers (km) of developed lands (that is, urban and suburban areas and interstate and State highways) (Johnson and others, 2011a), and most cases of nest abandonment by this species are related to human disturbance (Schroeder and others, 1999).

Roads and, to a lesser extent, recreational trails are a common feature in grassland landscapes. Humans can travel no further than 35 km from a road in the conterminous United States (Watts and others, 2007). In examining causes of endangerment for North American species that are classified as threatened or endangered by the FWS, Czech and others (2000) concluded that roads were associated with more causes of species endangerment than any other source. Roads may affect wildlife and their habitats in various ways. The negative effects of roads may include increasing human use and access to an area, facilitating the loss of biodiversity, providing avenues for the spread of invasive plants and creating optimal growing sites for those plants, serving as barriers for animal dispersal (and perhaps genetically isolating populations), enhancing movements of predators and brood parasites, altering the physical and chemical environments, and causing mortality during road construction and through collisions with vehicles (Trombulak and Frissell, 2000; Kuvlesky and others, 2007). Increased and easier access for vehicles and machinery may accelerate the conversion of grassland to cropland or other uses (for example, energy development) as well as increase avenues for the spread of invasive plants. Roads also allow vehicular access to remote grasslands, thus increasing habitat fragmentation (Saunders and others, 2002). The response of grassland birds to trails and roads can take the form of reduced density, territoriality, nesting, and nest success (Miller and others, 1998; Sutter and others, 2000; Pitman and others, 2005; Koper and Schmiegelow, 2006; Linnen, 2008; Dale and others, 2009; Sliwinski and Koper, 2012; Wellicome and others, 2014; Ludlow and others, 2015; Yoo and Koper, 2017; Nenninger and Koper, 2018).

Encroaching urbanization creates the proliferation of structures such as cellular communications towers, transmission lines, and energy-conversion facilities, all of which have been determined to cause mortality to birds (Erickson and others, 2001; Government Accountability Office, 2005; Arnett

The increasing encroachment of non-agricultural anthropogenic activities, such as wind-energy generation facilities, has a modern-day impact on bird populations. Researchers have documented the behavioral avoidance of some species of grassland birds and waterfowl to wind-energy infrastructure, such as to this wind facility in Dickey County, North Dakota; photograph by Chuck Loesch, U.S. Fish and Wildlife Service.

and others, 2007; Kuvlesky and others, 2007; Mabey and Paul, 2007; Winder and others, 2014a). Grassland birds and grassland-nesting waterbirds may avoid otherwise-suitable breeding habitat near wind infrastructure (Loesch and others, 2013; Niemuth and others, 2013; Winder and others, 2014b; Shaffer and Buhl, 2016). Shaffer and Buhl (2016) reported that seven of nine grassland bird species exhibited avoidance within 300 m of turbines, and in some cases beyond 300 m, and that avoidance effects were generally larger from 2–5 years post-construction than the year immediately following construction. Shaffer and others (2019b) calculated average avoidance rates ranging from 18 percent for the first-year post-construc-tion to 53 percent by the fifth-year post-construction for eight species of grassland bird species in the northern Great Plains. Mahoney and Chalfoun (2016) attributed reduced nest survival and nestling mass of Horned Larks to turbine density. Winder and others (2014b) reported behavioral avoidance of wind turbines by female Greater Prairie-Chickens (*Tympanuchus cupido*); average home range size ranged from 54 km^2 during the pre-construction phase to 97 km^2 during the post-construction phase. Winder and others (2015) determined that distance to wind turbine had a negative effect on lek persistence for leks that were less than 8 km (5 mi) from turbines during a 2–3 year post-construction period; abandonment rate was about 3 times higher for leks less than 8 km (5 mi) from a turbine compared to leks that were 8 km (5 mi) or more from a turbine. Whalen and others (2018) reported that male Greater Prairie-Chickens adjusted the acoustic properties of their vocalizations in response to the noise generated by wind turbines. For female Greater Sage-Grouse, LeBeau and others (2014) determined that for every 1-km (0.6 mi) increase in distance from the nearest turbine, the risk of nest or brood failure declined 7.1 percent and 38.1 percent, respectively.

As with wind development, oil and gas development can lower the quality of grassland habitat near energy infrastructure. Impacts include behavioral avoidance; reduced abundance, parental care, and nest success; and changes in acoustic song properties (Hamilton and others, 2011; Thompson and others, 2015; Bernath-Plaisted and Koper, 2016; Sutter and others, 2016; Ng, 2017; Nenninger and Koper, 2018; Warrington and others, 2018). Van Wilgenburg and others (2013) estimated that the number of nests of boreal forest and grassland songbirds disturbed annually within the Western Canadian Sedimentary Basin by all terrestrial oil and

Researchers have documented the behavioral avoidance of some species of grassland birds to oil infrastructure, such as to this well pump jack in Fallon County, Montana; photograph by Lawrence D. Igl, U.S. Geological Survey.

gas sectors combined (including seismic exploration, pipeline right-of-way clearing, well-pad clearing, and oil sands mining) ranged between 11,840 and 60,380. For grouse species, energy development can cause avoidance; lek abandonment; and declines in recruitment, annual survival, and abundance (Pitman and others, 2005; Rowland, 2019).

Cumulative impacts of anthropogenic disturbances on birds and other wildlife include increased road construction and vehicular traffic, increased human presence, alteration of biological communities, spread of non-native plants, the presence of very large structures on the landscape (for example, wind turbines), and other anthropogenic disturbances. The cumulative impacts of anthropogenic pressures on wildlife are unknown and are very difficult to study.

The potential effects of global climate change on grassland birds are largely unknown and beyond the management scope of this document. Price (1995) predicted that the summer distributions of 23 grassland bird species would shift under a global climate change scenario. Several species were predicted to become locally or regionally extirpated, and the species composition of grassland bird communities also was predicted to change. Niemuth and others (2014) cautioned that direct effects of climate change in the northern Great Plains may be overshadowed by indirect effects such as intensified land use and increased pressure to convert grasslands and drain wetlands.

Considerations in Grassland Reserve Design

The insights gleaned from habitat fragmentation studies can inform land management decisions on how best to manage grasslands for grassland birds. Research and management initially focused on characteristics of the proximate habitat, but more recent approaches consider characteristics of grasslands based on their location within a larger landscape matrix. Sample and Mossman (1997) suggested managing grassland bird habitats at three scales: large landscapes (greater than or equal to 4,050 ha), medium landscapes (405–4,050 ha), and small blocks (16–405 ha). With this approach, a resource manager can maintain a diversity of habitats and a more diverse grassland bird community at larger scales and manage for the needs of individual species at smaller scales. Larger grasslands also can be partitioned into a mosaic of management treatments, thus providing a variety of vegetation heights and densities for several grassland bird species with disparate habitat needs (Renken and Dinsmore, 1987; Hands and others, 1989; Askins, 1993; Collister, 1994; Herkert and others, 1996; Sample and Mossman, 1997; Vickery and others, 2000; Fuhlendorf and Engle, 2001; Winter and others, 2005a).

Larger grasslands are advantageous over smaller patches when managing for grassland birds because larger areas support a diversity of habitats, a more diverse grassland bird community, and a larger number of individuals of a given species, especially area-sensitive species (Herkert, 1994; Sample and Mossman, 1997; Herkert and others, 2003; Winter and others, 2006). Some species of birds, such as raptors and prairie grouse, have large home ranges and thus need larger areas of grassland to support their habitat needs (Hamerstrom and others, 1957; Knopf, 1988). Providing patches with a higher proportion of interior habitat relative to edge habitat will be important for many grassland bird species, especially those that are area sensitive (Davis, 2004). Ribic and others (2009), however, cautioned against blindly extrapolating patterns of area sensitivity found in one region to another, because multiple factors are likely operating. Understanding the factors that influence certain patterns of area sensitivity will improve regional conservation efforts.

Despite the undeniable importance of large grasslands for grassland birds, small grassland fragments may have value to grassland birds. Small patches typically are less expensive to acquire and easier to manage (Skagen and others, 2005;

Winter and others, 2006). Individual grassland tracts may be best suited for the management of a specific set of unique conditions or for a few species rather than for maximizing avian diversity (Vickery and others, 1999, 2000). For example, small patches may have conservation value if they provide important breeding habitat to young-age cohorts, to subordinate first-year breeders, or if they harbor important vegetation types or rare and endemic plant species (Ryan, 1990; Skagen and others, 2005; Winter and others, 2006). As demonstrated by Niemuth (2000) for Greater Prairie-Chickens, it may be important to distinguish among different types of grasslands. Some species thought to require large grassland patches may use smaller patches if the small patches are part of a larger grassland complex (Ribic and others, 2009). Small patches also may act as “stepping stones” or corridors to nearby, larger patches (Ryan, 1990). Small native prairie patches with minimal edge habitat are important for those species that are not sensitive to patch size or shape (Davis, 2004). Care is warranted, however, to avoid managing grassland tracts that may be too small or too isolated to provide conservation benefits, because the area required to attract a species of grassland bird may be smaller than the area necessary to maintain a viable population of that grassland bird (Sample and Mossman, 1997). Isolated grasslands may hinder a grassland bird’s abilities to disperse, immigrate, and reproduce (Herkert and others, 1996; Winter and Faaborg, 1999; Davis, 2003).

Ryan (1990) provides some guidance on the tradeoffs between large and small patches. For example, decisions concerning the acquisition of small or large patches of wildlife habitat may depend more on the species present within the patches, the condition of the habitat and its potential for management, options for other acquisitions, the presence or absence of adjacent parcels, and on economic and political considerations rather than on ecological theory.

Managers may increase the size of grassland patches and reduce the amount of grassland edge by increasing the number of contiguous patches of grassland within reserves. In agricultural or fragmented regions, restoring and enhancing small and large grassland patches within landscapes that have a high proportion of grassland habitats and little or no woodland habitats would likely provide the greatest benefit for grassland birds (Fletcher and Koford, 2002; Ribic and others, 2009). Native prairies dissected by cropland likely provide more suitable grassland bird habitat than equivalently sized prairies fragmented by woodland (Jensen and Finck, 2004). If small patches of grassland are the only grasslands available for the creation of reserves, locating protected grasslands within proximity to one another and to other grassland habitats reduces the effects of isolation and improves connectivity by providing corridors of suitable habitat (Herkert and others, 1993). Square or circular patches have less edge habitat relative to interior habitat than patches that are longer or more irregular in shape (Herkert and others, 1993; Sample and Mossman, 1997; Johnson and Winter, 1999). Grant and others (2004a) recommended that the first priority of managers should be to reduce woodland encroachment to less than 20 percent in grasslands because even small increases in woody vegetation compromised the use of grasslands by several grassland bird species. As a general guide, tall woody plants should be reduced to levels within the range of natural variation of major ecological processes within the region of interest (Grant and others, 2006). Renfrew (2002) also encouraged the removal of wooded areas, treelines, and shrubby hedgerows near grasslands. Likewise, Naugle and others (1999) called for managers to limit the extent of woody vegetation encroachment in restored and natural wetlands.

Conservation planning and acquisition efforts should consider the landscape context in which grassland fragments under consideration are embedded (Niemuth and others, 2008). Because patch size might be less relevant to grassland passerines when fragments are located in treeless landscapes, the size requirements of a grassland reserve may vary with the quality of the core grassland, the proportion of grassland and forest in the surrounding landscape, diversity of land-cover types, edge density, and the composition of the local predator community (Davis, 2004; Winter and others, 2006). Ribic and others (2009) cautioned that easement and acquisition programs that protect individual patches of grassland habitat without regard for the surrounding landscape may meet with limited success. The findings of Bakker and others (2002) that occupancy rates for several grassland bird species were higher in small patches within landscapes with high grassland abundance than in large patches within landscapes with low grassland abundance further emphasize that the composition of the surrounding landscape may be more important than patch size. Lockhart and Koper (2018) stressed the importance of considering grassland configuration, expressed as a Landscape Shape Index, when evaluating the influence of grassland fragmentation on avian abundance and richness. Stephens and others (2004) stressed the necessity of concentrating anthropogenic disturbances in one locale rather than dispersing them across a management unit, as well as the need to develop reserves of large blocks of contiguous grassland. Cumulative effects of disturbance warrant examination from a landscape context. Local characteristics (for example, vegetation composition and structure) are more easily modified through an array of management treatments (for example, burning and grazing) than are the characteristics of the landscape (and its associated land uses) in which the grassland fragments are embedded (Niemuth and others, 2005). Spatially explicit habitat models, such as the Grassland Bird Conservation Area conceptual model (Johnson and others, 2010), can be used to help guide landscape-level conservation planning by predicting the occurrence of a particular species and the general suitability of a landscape (Niemuth and others, 2005; Niemuth and others, 2017). Models can provide an objective, quantitative method of evaluating landscapes for conservation and provide a basis for making conservation decisions. Conservation of highly suitable landscapes for grassland birds could then be promoted through aggressive easement programs (Higgins and others, 2002; May and others, 2002).

Predators and Brood Parasites

An additional consideration in the design and implementation of grassland reserves is the distribution and density of predators and brood parasites. For example, in mixed-grass prairies in Saskatchewan, vegetation structure was important in the selection of habitat by grassland birds, but nest success was not strongly related to vegetation structure, suggesting that extrinsic concerns such as predator density may be important for managing grassland birds (Davis, 2003). Smaller patches may place grassland birds in proximity to the brood-parasitic Brown-headed Cowbird, but it appears that the prevalence of cowbird brood parasitism is related less to patch size and more to the density or abundance of cowbirds in the grassland (Davis, 2003; Herkert and others, 2003). A species' avoidance of risks associated with predation and parasitism at grassland edges may be one of the mechanisms creating patch-size and patch-shape effects (Johnson, 2001).

Lahti (2001) suggested that knowledge of the predators in an area, including their responses to edges and fragmentation, is critical to understanding the effects of edges on predation. The nest-predator community for grassland birds can differ from one region to another (Thompson and others, 1999; Pietz and Granfors, 2000; Renfrew and Ribic, 2003), but account for a large proportion of nest failures. In an analysis of 18 grazing studies from nine ecoregions in Canada, Bleho and others (2014) concluded that 87 percent of 9,132 grassland bird nest failures were caused by predation, with cattle accounting for less than 3 percent of nest failures. Control of one predator species or subset of predators as a means to improve avian reproductive success may be offset by numerical increases or changes in foraging habitats of other predators (Renfrew and Ribic, 2003; Skagen and others, 2005). For example, removing woody edges may help to connect large, open areas that lack woody edges, but it also may redistribute mammalian nest predators and influence their movement patterns. Therefore, management efforts may benefit from monitoring programs that include the identification of specific nest predators and their distributions, with respect to important habitat features and their response to management, to predict patterns of nest predation (Grant and others, 2006). Management efforts then can be customized to the predators primarily responsible for local nest mortality (Chalfoun and others, 2002).

The eggs, young, and adults of birds are preyed upon by a number of species of mammals, snakes, and other birds, including the *A*, coyote (*Canis latrans*), *B*, raccoon (*Procyon lotor*), *C*, striped skunk (*Mephitis mephitis*), *D*, American badger (*Taxidea taxus*), *E*, red fox (*Vulpes vulpes*), *F*, plains garter snake (*Thamnophis radix*), and *G*, Great Horned Owl (*Bubo virginianus*). Photograph credits: coyote, John Carr, U.S. Fish and Wildlife Service; raccoon, Gary Miller, U.S. Fish and Wildlife Service; skunk, K. Theule, U.S. Fish and Wildlife Service; badger, Cindy Souders, U.S. Fish and Wildlife Service; fox, Pete Ramirez, Jr., U.S. Fish and Wildlife Service; snake, Krista Lundgren, U.S. Fish and Wildlife Service; owl, Tom Koerner, U.S. Fish and Wildlife Service.

Final Thoughts

Many questions remain for further research into the effects of vegetation, patch size and shape, edge, landscape, predators, and management on grassland birds, and how those factors influence management decisions. However, regardless of the particular question, it may be useful to replicate studies temporally and spatially to partition variance into process and sampling components (Stephens and others, 2004). Johnson (2002, p. 919) argued that "Similar conclusions obtained from studies of the same phenomenon conducted under widely differing conditions will give us greater confidence in the generality of those findings than would any single study."

In terms of management prescriptions, Ryan (1990, p. 103) aptly stated: "The current literature is valuable in describing approaches to prairie management but it cannot be used as prescriptions for on-site management actions. In listening to prairie managers I am continually impressed by the specificity of response of different grassland tracts to disturbance treatments. Combinations of soils, topography, existing plant community, management history, climatic conditions, timing of treatments, etc. produce unique results spatially and even temporally at the site. There is no substitute for experienced managers and their creative experimentation with available tools. What is an effective fire prescription to eliminate or control woody invasion at a North Dakota site is likely to be ineffective in Illinois. In some cases, adjoining tracts require different management regimes to effect similar results. Often only long-term trial and error by dedicated managers will provide desired results." To this we would add that careful, detailed documentation and publication of the results of management effects on grassland biota by experienced managers would provide valuable information for present and future resource managers.

Summary

The Great Plains of North America is defined as the land mass that encompasses the entire central portion of the North American continent that, at the time of European settlement, was an unbroken expanse of primarily herbaceous vegetation. The Great Plains extend from central Saskatchewan and Alberta to central Mexico and from Indiana to the Rocky Mountains. The expanses of herbaceous vegetation are often referred to as native prairie or native grasslands. Native grasslands share the characteristics of a general uniformity in vegetation structure, dominance by grasses and forbs, a near absence of trees and shrubs, annual precipitation ranging from 25 to 100 centimeters, extreme intra-annual fluctuations in temperature and precipitation, and a flat to rolling topography over which fires can spread. To the west of the Great Plains lie the sagebrush communities of the Great Basin, which extend from British Columbia and Saskatchewan to northern Arizona and New Mexico and from the eastern slopes of the Sierra Nevada and Cascade mountain ranges to western South Dakota. Sagebrush communities share similar characteristics to native grasslands, but their location east of the Rocky Mountains creates a more moderating influence from prevailing westerly winds that affect timing of peak precipitation and growth form of dominant vegetation. Native grasslands and sagebrush communities harbor a diverse array of grassland, wetland, and woodland plant and animal communities that are uniquely adapted to the natural forces of the Great Plains and Great Basin, namely the interactive forces of climate, fire, and grazing. The arrival of European settlers to North America brought profound change to native grassland and sagebrush communities, including the establishment of permanent towns and cities, the proliferation of cropland-based agricultural systems, and the suppression of wildfires. The near extirpation of bison by the 1860s paved the way for dramatic changes in the dominant grazers and a shift in the disturbance patterns that historically influenced vegetation structure. The greatest threat to native grasslands and sagebrush communities in modern times is their loss due to conversion to rowcrop agriculture and to urbanization. Concomitant with habitat loss is a precipitous decline in populations of bird species that evolved with, and are uniquely adapted to, the native grassland and sagebrush habitats. Avian population trends are linked strongly to agricultural land use. Besides outright loss of suitable breeding habitat, agricultural practices affect birds through factors such as pesticide exposure, habitat fragmentation, shifts in predator community composition, and occurrence of brood parasites. Bird populations face other stressors, such as loss of habitat to and behavioral avoidance of urbanized areas, roads, and infrastructure associated with energy production.

Despite the many anthropogenic changes to North American grassland and sagebrush communities, some bird species are adaptable and opportunistic in their habitat selection and now utilize one or more human-created habitats. Human-created habitats include pastures, hayfields, agricultural terraces, crop buffer strips, field borders, grassed waterways, fencerows, road rights-of-way, airports, reclaimed coal mines, and planted wildlife cover. Fields of seeded grasslands enrolled in Federal long-term set-aside programs, such as the Conservation Reserve Program in the United States and the Permanent Cover Program in Canada, provide important nesting habitat for grassland bird species. The array of habitats used by birds makes habitat and avian management a complex undertaking, and the scale (for example, local, regional, international) at which management actions can be implemented are such that a universal approach to managing grasslands for the conservation of the entire suite of bird species does not exist. Experienced land managers recognize that it is impossible to manage for all bird species simultaneously, and thus, prioritization is necessary towards those habitats or bird species that the manager or management agency ranks highest for a specific region or management unit. The primary tools available for management are burning, grazing, mowing, herbicide application, and idling, but before choosing a particular practice, a manager will want to consider issues of seasonality, intensity, and frequency.

Despite the thousands of studies that are cited in this compendium, much remains unknown about the effects of management practices on bird species. The series of species accounts in this compendium review the current state of knowledge regarding management of grassland and sagebrush bird species and summarize information on the effects of management practices on individual species. The accounts do not give definitive statements on the effects of management practices for any particular species, primarily because there are very few replicated studies in which identical management practices have been applied in the same geographical area with consistent results, which are elements necessary to provide concrete recommendations for the management of a particular species in a particular area. Documentation of the effects of management treatments on individual species through statistically sound methods that incorporate multiple years and locations will further scientists' and land managers' knowledge far more than 1–2-year studies that are limited in scope as well as time, but studies of that scope and breadth are rare.

References

Ahlering, M.A., Johnson, D.H., and Faaborg, J., 2009, Factors associated with arrival densities of Grasshopper Sparrow (*Ammodramus savannarum*) and Baird's Sparrow (*A. Bairdii*) in the upper Great Plains: The Auk, v. 126, no. 4, p. 799–808. [Also available at https://doi.org/10.1525/auk.2009.08154.]

Allen, A.W., and Vandever, M.W., 2012, Conservation Reserve Program (CRP) contributions to wildlife habitat, management issues, challenges and policy choices—An annotated bibliography: U.S. Geological Survey Scientific Investigations Report 2012–5066, 185 p. [Also available at https://doi.org/10.3133/sir20125066.]

Anderson, J.E., and Inouye, R.S., 2001, Landscape-scale changes in plant species abundance and biodiversity of a sagebrush steppe over 45 years: Ecological Monographs, v. 71, no. 4, p. 531–556. [Also available at https://dx.doi.org/10.1890/0012-9615(2001)071%5B0531:LSCIPS%5D2.0.CO;2.]

Anderson, R.C., 1982, An evolutionary model summarizing the roles of fire, climate, and grazing animals in the origin and maintenance of grasslands—An end paper, *in* Estes, J.R., Tyrl, R.J., and Brunken, J.N., eds., Grasses and grasslands—Systematics and ecology: Norman, Okla., University of Oklahoma Press, p. 297–308.

Andrén, H., 1994, Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat—A review: Oikos, v. 71, no. 3, p. 355–366. [Also available at https://doi.org/10.2307/3545823.]

Arnett, E.B., Inkley, D.B., Johnson, D.H., Larkin, R.P., Manes, S., Manville, A.M., Mason, J.R., Morrison, M.L., Strickland, M.D., and Thresher, R., 2007, Impacts of wind energy facilities on wildlife and wildlife habitat: Bethesda, Md., The Wildlife Society, Technical Review 07-2, 49 p., accessed July 2019 at https://wildlife.org/wp-content/uploads/2014/05/Wind07-2.pdf.

Askins, R.A., 1993, Population trends in grassland, shrubland, and forest birds in eastern North America: Current Ornithology, v. 11, p. 1–34. [Also available at https://doi.org/10.1007/978-1-4757-9912-5_1.]

Axelrod, D.I., 1985, Rise of the grassland biome, central North America: Botanical Review, v. 51, no. 2, p. 163–201. [Also available at https://doi.org/10.1007/BF02861083.]

Babbitt, B., 1998, Statement by Secretary of the Interior Bruce Babbitt on invasive alien species, *in* Proceedings of the Science in Wildland Weed Management Symposium: Denver, Colo., accessed July 2019 at https://archive.org/details/proceedingsscien40scie.

Baer, N.W., 1989, Shelterbelts and windbreaks in the Great Plains: Journal of Forestry, v. 87, no. 4, p. 32–36.

Bailey, R.G., 1980, Description of the ecoregions of the United States: U.S. Department of Agriculture, Miscellaneous Publication, no. 1391, 77 p.

Bajema, R.A., DeVault, T.L., Scott, P.E., and Lima, S.L., 2001, Reclaimed coal mine grasslands and their significance for Henslow's Sparrows in the American Midwest: The Auk, v. 118, no. 2, p. 422–431. [Also available at https://dx.doi.org/10.2307/4089803.]

Baker, W.L., 2011, Pre-Euro-American and recent fire in sagebrush ecosystems, *in* Knick, S.T, and Connelly, J.W., eds., Greater Sage-Grouse—Ecology and conservation of a landscape species and its habitats: Studies in Avian Biology, v. 38, p. 185–201. [Also available at https://doi.org/10.1525/california/9780520267114.003.0012.]

Bakker, K.K., Naugle, D.E., and Higgins, K.F., 2002, Incorporating landscape attributes into models for migratory grassland bird conservation: Conservation Biology, v. 16, no. 6, p. 1638–1646. [Also available at https://dx.doi.org/10.1046/j.1523-1739.2002.01328.x.]

Basore, N.S., Best, L.B., and Wooley, J.B., 1986, Bird nesting in Iowa no-tillage and tilled cropland: The Journal of Wildlife Management, v. 50, no. 1, p. 19–28. [Also available at https://dx.doi.org/10.2307/3801482.]

Batt, B.D.J., 1996, Prairie ecology-prairie wetlands, *in* Samson, F.B., and Knopf, F.L., eds., Prairie conservation—Preserving North America's most endangered ecosystem: Washington, D.C., Island Press, p. 78–88.

Beecher, N.A., Johnson, R.J., Brandle, J.R., Case, R.M., and Young, L.J., 2002, Agroecology of birds in organic and nonorganic farmland: Conservation Biology, v. 16, no. 6, p. 1620–1631. [Also available at https://doi.org/10.1046/j.1523-1739.2002.01228.x.]

Belsky, A.J., and Gelbard, J.L., 2000, Livestock grazing and weed invasions in the arid West: Bend, Oreg., Oregon Natural Desert Association, 31 p.

Berger, A.J., 1951, The cowbird and certain host species in Michigan: The Wilson Bulletin, v. 63, no. 1, p. 26–34.

Bernath-Plaisted, J., and Koper, N., 2016, Physical footprint of oil and gas infrastructure, not anthropogenic noise, reduces nesting success of some grassland songbirds: Biological Conservation, v. 204, p. 434–441. [Also available at https://doi.org/10.1016/j.biocon.2016.11.002.]

Best, L.B., 1978, Field Sparrow reproductive success and nesting ecology: The Auk, v. 95, no. 1, p. 9–22. [Also available at https://dx.doi.org/10.2307/4085491.]

Best, L.B., 1985, Conservation vs. conventional tillage—Wildlife management consideration, *in* D'ltri, F.M., ed., A systems approach to conservation tillage: Chelsea, Mich., Lewis Publications, Inc., p. 315–326.

Best, L.B., 1986, Conservation tillage—Ecological traps for nesting birds?: Wildlife Society Bulletin, v. 14, no. 3, p. 308–317.

Best, L.B., Campa, H., III, Kemp, K.E., Robel, R.J., Ryan, M.R., Savidge, J.A., Weeks, H.P., Jr., and Winterstein, S.R., 1997, Bird abundance and nesting in CRP fields and cropland in the Midwest—A regional approach: Wildlife Society Bulletin, v. 25, no. 4, p. 864–877.

Beyersbergen, G.W., Niemuth, N.D., and Norton, M.R., coordinators, 2004, Northern prairie and parkland waterbird conservation plan—A plan associated with the Waterbird Conservation for the Americas initiative: Denver, Colo., Prairie Pothole Joint Venture, 183 p., accessed July 2019 at https://www.fws.gov/mountain-prairie/refuges/hapetResources/updatedFiles/publications/Beyersbergen.et.al.2004.waterbird.plan.PPJV.pdf.

Billings, W.D., 1994, Ecological impacts of cheatgrass and resultant fire on ecosystems in the western Great Basin, *in* Monsen, S.B., and Kitchen, S.G., compilers, Proceedings of the Ecology and Management of Annual Rangelands Symposium: Ogden, Utah, U.S. Department of Agriculture, Forest Service, Intermountain Research Station, General Technical Report INT-GTR-313, p. 22–30.

Blackburn, T.E., and Anderson, K., eds., 1993, Before the wilderness—Environmental management by native Californians: Menlo Park, Calif., Ballena Press, 476 p.

Blair, R.B., 1996, Land use and avian species diversity along an urban gradient: Ecological Applications, v. 6, no. 2, p. 506–519. [Also available at https://doi.org/10.2307/2269387.]

Blann, K.L., Anderson, J.L., Sands, G.R., and Vondracek, B., 2009, Effects of agricultural drainage on aquatic ecosystems—A review: Critical Reviews in Environmental Science and Technology, v. 39, no. 11, p. 909–1001. [Also available at https://doi.org/10.1080/10643380801977966.]

Bleho, B.I., Koper, N., and Machtans, C.S., 2014, Direct effects of cattle on grassland birds in Canada: Conservation Biology, v. 28, no. 3, p. 724–734. [Also available at https://dx.doi.org/10.1111/cobi.12259.]

Bock, C.E., and Bock, J.H., 1993, Cover of perennial grasses in southeastern Arizona in relation to livestock grazing: Conservation Biology, v. 7, no. 2, p. 371–377. [Also available at https://doi.org/10.1046/j.1523-1739.1993.07020371.x.]

Bock, C.E., Saab, V.A., Rich, T.D., and Dobkin, D.S., 1993, Effects of livestock grazing on Neotropical migratory landbirds in western North America, *in* Finch, D.M., and Stangel, P.W., eds., Status and management of Neotropical migratory birds: Fort Collins, Colo., U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-229, p. 296–309.

Bollinger, E.K., Bollinger, P.B., and Gavin, T.A., 1990, Effects of hay-cropping on eastern populations of the Bobolink: Wildlife Society Bulletin, v. 18, no. 2, p. 142–150.

Boyd, C.S., Beck, J.L., and Tanaka, J.A., 2014, Livestock grazing and Sage-Grouse habitat—Impacts and opportunities: Journal of Rangeland Applications, v. 1, p. 58–77.

Brady, N.C., 1990, The nature and properties of soils (10th ed.): New York, N.Y., Macmillan Publishing Company, 621 p.

Bragg, T.B., 1995, The physical environment of Great Plains grasslands, *in* Joern, A., and Keeler, K.H., eds., The changing prairie—North American grasslands: New York, N.Y., Oxford Press, p. 49–81.

Bragg, T.B., and Steuter, A.A., 1996, Prairie ecology—The mixed-grass prairie, in Samson, F.B., and Knopf, F.L., eds., Prairie conservation—Preserving North America's most endangered ecosystem: Washington, D.C., Island Press, p. 53–65.

Briske, D.D., Derner, J.D., Brown, J.R., Fuhlendorf, S.D., Teague, W.R., Havstad, K.M., Gillen, R.L., Ash, A.J., and Willms, W.D., 2008, Rotational grazing on rangelands—Reconciliation of perception and experimental evidence: Rangeland Ecology and Management, v. 61, no. 1, p. 3–17. [Also available at https://doi.org/10.2111/06-159R.1.]

Brothers, T.S., 1990, Surface-mine grasslands: Geographical Review, v. 80, no. 3, p. 209–225. [Also available at https://doi.org/10.2307/215300.]

Brown, J.R., and Archer, S., 1989, Woody plant invasion of grasslands—Establishment of honey mesquite (*Prosopis glandulosa* var. *glandulosa*) on sites differing in herbaceous biomass and grazing history: Oecologia, v. 80, no. 1, p. 19–26. [Also available at https://doi.org/10.1007/BF00789926.]

Brudvig, L.A., Mabry, C.M., Miller, J.R., and Walker, T.A., 2007, Evaluation of central North American prairie management based on species diversity, life form, and individual species metrics: Conservation Biology, v. 21, no. 3, p. 864–874. [Also available at https://doi.org/10.1111/j.1523-1739.2006.00619.x.]

Bryan, G.G., and Best, L.B., 1991, Bird abundance and species richness in grassed waterways in Iowa rowcrop fields: American Midland Naturalist, v. 126, no. 1, p. 90–102. [Also available at https://dx.doi.org/10.2307/2426153.]

Bryan, G.G., and Best, L.B., 1994, Avian nest density and success in grassed waterways in Iowa rowcrop fields: Wildlife Society Bulletin, v. 22, no. 4, p. 583–592.

Burkhardt, J.W., 1996, Herbivory in the Intermountain West—An overview of evolutionary history, historic cultural impacts and lessons from the past: Moscow, Idaho, University of Idaho, College of Natural Resources, Idaho Forest, Wildlife and Range Experiment Station, Station Bulletin 58, 18 p.

Caccamise, D.F., Reed, L.M., Delay, L.S., Bennett, K.A., and Dosch, J.J., 1996, The avian communities of a suburban grassland refugium—Population studies at an airport in Northeastern United States: Acta Ornithologica, v. 31, no. 1, p. 3–13.

Camp, M., and Best, L.B., 1994, Nest density and nesting success of birds in roadsides adjacent to rowcrop fields: American Midland Naturalist, v. 131, no. 2, p. 347–358. [Also available at https://doi.org/10.2307/2426260.]

Castrale, J.S., 1985, Responses of wildlife to various tillage conditions: Transactions of the North American Wildlife and Natural Resources Conference, v. 50, p. 142–149.

Chalfoun, A.D., Thompson, F.R., III, and Ratnaswamy, M.J., 2002, Nest predators and fragmentation—A review and meta-analysis: Conservation Biology, v. 16, no. 2, p. 306–318. [Also available at https://doi.org/10.1046/j.1523-1739.2002.00308.x.]

Chapman, K.A., Fischer, A., and Ziegenhagen, M.K., 1998, Valley of grass—Tallgrass prairie and parkland of the Red River Region: St. Cloud, Minn., North Star Press of St. Cloud, Inc., 122 p.

Churchwell, R.T., Davis, C.A., Fuhlendorf, S.D., and Engle, D.M., 2008, Effects of patch-burn management on Dickcissel nest success in a tallgrass prairie: The Journal of Wildlife Management, v. 72, no. 7, p. 1596–1604.

Clements, R.E., 1920, Plant indicators—The relation of plant communities to process and practice: Washington, D.C., Carnegie Institute of Washington, 388 p. [Also available at https://doi.org/10.5962/bhl.title.28569.]

Clewell, A.F., Rieger, J., and Munro, J.W., 2005, Guidelines for constructing and managing ecological restoration projects (2nd ed.): Tucson, Ariz., Society for Ecological Restoration International, 16 p.

Cody, M.L., 1985, Habitat selection in grassland and open-country birds, *in* Cody, M.L., ed., Habitat selection in birds: Orlando, Fla., Academic Press, p. 191–226.

Collins, S.L., 1992, Fire frequency and community heterogeneity in tallgrass prairie vegetation: Ecology, v. 73, no. 6, p. 2001–2006. [Also available at https://doi.org/10.2307/1941450.]

Collister, D.M., 1994, Breeding ecology and habitat preservation of the Loggerhead Shrike in southeastern Alberta: Calgary, Alberta, University of Calgary, Master's Thesis, 161 p.

Coppedge, B.R., Engle, D.M., Masters, R.E., and Gregory, M.S., 2001, Avian response to landscape change in fragmented southern Great Plains grasslands: Ecological Applications, v. 11, no. 1, p. 47–59. [Also available at https://doi.org/10.1890/1051-0761(2001)011[0047:ARTLCI]2.0.CO;2.]

Coppedge, B.R., Fuhlendorf, S.D., Harrell, W.C., and Engle, D.M., 2008, Avian community response to vegetation and structural features in grasslands managed with fire and grazing: Biological Conservation, v. 141, no. 5, p. 1196–1203. [Also available at https://dx.doi.org/10.1016/j.biocon.2008.02.015.]

Coppock, D.L., Detling, J.K., Ellis, J.E., and Dyer, M.I., 1983, Plant-herbivore interactions in a North American mixed-grass prairie—I. Effects of Black-tailed Prairie Dogs on intraseasonal aboveground plant biomass and nutrient dynamics and plants species diversity: Oecologia, v. 56, no. 1, p. 1–9. [Also available at https://doi.org/10.1007/BF00378210.]

Crawford, J.A., Olson, R.A., West, N.E., Mosley, J.C., Schroeder, M.A., Whitson, T.D., Miller, R.F., Gregg, M.A., and Boyd, C.S., 2004, Ecology and management of Sage-Grouse and Sage-Grouse habitat: Journal of Range Management, v. 57, no. 1, p. 2–19. [Also available at https://dx.doi.org/10.2307/4003949.]

Cunningham, M.A., and Johnson, D.H., 2006, Proximate and landscape factors influence grassland bird distributions: Ecological Applications, v. 16, no. 3, p. 1062–1075. [Also available at https://dx.doi.org/10.1890/1051-0761(2006)016%5B1062:PALFIG%5D2.0.CO;2.]

Czech, B., Krausman, P.R., and Devers, P.K., 2000, Economic associations among causes of species endangerment in the United States: associations among causes of species endangerment in the United States reflect the integration of economic sectors, supporting the theory and evidence that economic growth proceeds at the competitive exclusion of nonhuman species in the aggregate: BioScience, v. 50, no. 7, p. 593–601. [Also available at https://doi.org/10.1641/0006-3568(2000)050[0593:EAACOS]2.0.CO;2.]

Dahl, T.E., 1990, Wetlands losses in the United States 1780‘s to 1980’s: Washington, D.C., U.S. Department of the Interior, Fish and Wildlife Service, 13 p., accessed July 2019 at https://www.fws.gov/wetlands/Documents/Wetlands-Losses-in-the-United-States-1780s-to-1980s.pdf.

Dahl, T.E., 2006, Status and trends of wetlands in the conterminous United States, 1998 to 2004: Washington, D.C., U.S. Department of the Interior, Fish and Wildlife Service, 112 p., accessed July 2019 at https://www.fws.gov/wetlands/Documents/Status-and-Trends-of-Wetlands-in-the-Conterminous-United-States-1998-to-2004.pdf.

Dahl, T.E., 2011, Status and trends of wetlands in the conterminous United States, 2004 to 2009: Washington, D.C., U.S. Department of the Interior, Fish and Wildlife Service, 107 p., accessed April July at https://www.fws.gov/wetlands/Documents/Status-and-Trends-of-Wetlands-in-the-Conterminous-United-States-2004-to-2009.pdf.

Dahl, T.E., 2014, Status and trends of prairie wetlands in the United States 1997 to 2009: Washington, D.C., U.S. Department of the Interior, Fish and Wildlife Service, Ecological Services, 67 p., accessed at July 2019 https://www.fws.gov/wetlands/Documents/Status-and-Trends-of-Prairie-Wetlands-in-the-United-States-1997-to-2009.pdf.

Dale, B.C., Wiens, T.S., and Hamilton, L.E., 2009, Abundance of three grassland songbirds in an area of natural gas infill drilling in Alberta, Canada, *in* Rich, T.D., Arizmendi, C., Demarest, D.W., and Thompson, C., eds., Proceedings of the fourth International Partners in Flight Conference: Tundra to Tropics: McAllen, Tex., Partners in Flight, p. 194–204.

Daubenmire, R., 1978, Plant geography with special reference to North America: New York, N.Y., Academic Press, 346 p.

Davis, S.K., 2003, Habitat selection and demography of mixed-grass prairie songbirds in a fragmented landscape: Regina, Saskatchewan, University of Regina, Ph.D. Dissertation, 131 p.

Davis, S.K., 2004, Area sensitivity in grassland passerines—Effects of patch size, patch shape, and vegetation structure on bird abundance and occurrence in southern Saskatchewan: The Auk, v. 121, no. 4, p. 1130–1145. [Also available at https://dx.doi.org/10.2307/4090481.]

Davis, S.K., 2017, Sprague’s Pipit breeding biology and reproductive success in planted and native grasslands: Journal of Avian Biology, v. 49, no. 4. [Also available at https://doi.org/10.1111/jav.01547.]

Davis, S.K., Devries, J.H., and Armstrong, L.M., 2017, Variation in passerine use of burned and hayed planted grasslands: The Journal of Wildlife Management, v. 81, no. 8, p. 1494–1504. [Also available at https://doi.org/10.1002/jwmg.21316.]

Davis, S.K., Ludlow, S.M., and McMaster, D.G., 2016, Reproductive success of songbirds and waterfowl in native mixed-grass pasture and planted grasslands used for pasture and hay: The Condor, v. 118, no. 4, p. 815–834. [Also available at https://dx.doi.org/10.1650/CONDOR-16-16.1.]

Derner, J.D., Lauenroth, W.K., Stapp, P., and Augustine, D.J., 2009, Livestock as ecosystem engineers for grassland bird habitat in the western Great Plains of North America: Rangeland Ecology and Management, v. 62, no. 2, p. 111–118. [Also available at https://doi.org/10.2111/08-008.1.]

DeVault, T.L., Scott, P.E., Bajema, R.A., and Lima, S.L., 2002, Breeding bird communities of reclaimed coal-mine grasslands in the American Midwest: Journal of Field Ornithology, v. 73, no. 3, p. 268–275. [Also available at https://doi.org/10.1648/0273-8570-73.3.268.]

Dix, R.L., 1964, A history of the biotic and climatic changes within the North American grassland, *in* Crips, D.J., ed., Grazing in terrestrial and marine environments: Oxford, England, Blackwell Scientific Publications, British Ecological Society, p. 71–89.

Dosskey, M.G., Eisenhauer, D.E., and Helmers, M.J., 2005, Establishing conservation buffers using precision information: Journal of Soil and Water Conservation, v. 60, no. 6, p. 349–354.

Droege, S., and Sauer, J.R., 1994, Are more North American species decreasing than increasing? *in* Hagemeijer, E.J.M., and Verstrael, T.J., eds., Bird numbers 1992—Distribution, monitoring, and ecological aspects: Proceedings of the twelfth International Conference of IBCC and EOAC, Noordwijkerhout, The Netherlands, p. 297–306.

Duchardt, C.J., Miller, J.R., Debinski, D.M., and Engle, D.M., 2016, Adapting the fire-grazing interaction to smallpastures in a fragmented landscape for grassland bird conservation: Rangeland Ecology and Management, v. 69, no. 4, p. 300–309. [Also available at https://dx.doi.org/10.1016/j.rama.2016.03.005.]

Ducks Unlimited, 2019, Ducks Unlimited website: accessed July 2019 at https://www.ducks.org.

Duebbert, H.F., 1981, Breeding birds on waterfowl production areas in northeastern North Dakota: Prairie Naturalist, v. 13, no. 1, p. 19–22.

Duebbert, H.F., Erling, T.J., Kenneth, F.H., and Erling, B.P., 1981, Establishment of seeded grasslands for wildlife habitat in the Prairie Pothole Region: Jamestown, N. Dak., U.S. Fish and Wildlife Service, Northern Prairie Wildlife Research Center, Special Scientific Report-Wildlife, no. 234, 21 p.

Dugger, B.D., and Dugger, K.M., 2002, Long-billed Curlew (*Numenius americanus*) (ver. 2.0), *in* Poole, A.F., and Gill, F.B., eds., The birds of North America database: Ithaca, N.Y., Cornell Lab of Ornithology, accessed July 2019 at https://doi.org/10.2173/bna.628.

Eom, A.-H., Hartnett, D.C., Wilson, G.W.T., and Figge, D.A.H., 1999, The effect of fire, mowing and fertilizer amendment on arbuscular mycorrhizas in tallgrass prairie: American Midland Naturalist, v. 142, no. 1, p. 55–70. [Also available at https://doi.org/10.1674/0003-0031(1999)142[0055:TEOFMA]2.0.CO;2.]

Erickson, W.P., Johnson, G.D., Strickland, M.D., Young, D.P., Jr., Sernka, K.J., and Good, R.E., 2001, Avian collisions with wind turbines—A summary of existing studies and comparisons to other sources of avian collision mortality in the United States: Washington, D.C., National Wind Coordinating Committee (NWCC), Resource Document, 62 p., accessed July 2019 at https://www.west-inc.com/reports/avian_collisions.pdf.

Euliss, N.H., Jr., LaBaugh, J.W., Fredrickson, L.H., Mushet, D.M., Laubhan, M.K., Swanson, G.A., Winter, T.C., Rosenberry, D.O., and Nelson, R.D., 2004, The wetland continuum—A conceptual framework for interpreting biological studies: Wetlands, v. 24, no. 2, p. 448–458. [Also available at https://doi.org/10.1672/0277-5212(2004)024[0448:TWCACF]2.0.CO;2.]

Fahnestock, J.T., Larson, D.L., Plumb, G.E., and Detling, J.K., 2003, Effects of ungulates and prairie dogs on seed banks and vegetation in a North American mixed-grass prairie: Plant Ecology, v. 167, no. 2, p. 255–268. [Also available at https://doi.org/10.1023/A:1023929627168.]

Farris, A.L., and Cole, J.H., 1981, Strategies and goals for wildlife habitat restoration on agricultural lands: Transactions of the North American Wildlife and Natural Resources Conference, v. 46, p. 130–136.

Fedkiw, J., 1989, The evolving use and management of the nation's forests, grasslands, croplands, and related resources: Fort Collins, Colo., U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-175.

Fenneman, N.M., 1931, Physiography of the western United States: New York, N.Y., McGraw-Hill Book Company Inc., 534 p.

Fitzgerald, J.A., Pashley, D.N., Lewis, S.J., and Pardo, B., 1998, Partners in Flight Bird Conservation Plan for The Northern Tallgrass Prairie (Physiographic Area 40) (ver. 1.0): American Bird Conservancy, 49 p., accessed July 2019 at https://www.partnersinflight.org/resources/pif-bird-conservation-plan-the-northern-tall-grass-prairie-physiographic-area-40/.

Fitzgerald, S.M., and Tanner, G.W., 1992, Avian community response to fire and mechanical shrub control in south Florida: Journal of Range Management, v. 45, no. 4, p. 396–400. [Also available at https://doi.org/10.2307/4003090.]

Flanders, A.A., Kuvlesky, W.P., Jr., Ruthven, D.C., III, Zaiglin, R.E., Bingham, R.L., Fulbright, T.E., Hernandez, F., and Brennan, L.A., 2006, Effects of invasive exotic grasses on south Texas rangeland breeding birds: The Auk, v. 123, no. 1, p. 171–182. [Also available at https://dx.doi.org/10.1642/0004-8038(2006)123%5B0171:EOIEGO%5D2.0.CO;2.]

Fletcher, R.J., Jr., and Koford, R.R., 2002, Habitat and landscape associations of breeding birds in native and restored grasslands: The Journal of Wildlife Management, v. 66, no. 4, p. 1011–1022. [Also available at https://dx.doi.org/10.2307/3802933.]

Freemark, K.E., and Kirk, D.A., 2001, Birds on organic and conventional farms in Ontario—Partitioning effects of habitat and practices on species composition and abundance: Biological Conservation, v. 101, no. 3, p. 337–350. [Also available at https://doi.org/10.1016/S0006-3207(01)00079-9.]

Fox, G.A., Mineau, P., Collins, B., and James, P.C., 1989, The impact of the insecticide carbofuran (Furadan 480F) on the Burrowing Owl in Canada: Ottawa, Ontario, Canadian Wildlife Service, Technical Report Series, no. 72, 21 p.

Frawley, B.J., 1989, The dynamics of nongame bird breeding ecology in Iowa alfalfa fields: Ames, Iowa, Iowa State University, Master's Thesis, 94 p.

Fuhlendorf, S.D., and Engle, D.M., 2001, Restoring heterogeneity on rangelands—Ecosystem management based on evolutionary grazing patterns—We propose a paradigm that enhances heterogeneity instead of homogeneity to promote biological diversity and wildlife habitat on rangelands grazed by livestock: BioScience, v. 51, no. 8, p. 625–632. [Also available at https://doi.org/10.1641/0006-3568(2001)051[0625:RHOREM]2.0.CO;2.]

Fuhlendorf, S.D., and Engle, D.M., 2004, Application of the fire-grazing interaction to restore a shifting mosaic on tallgrass prairie: Journal of Applied Ecology, v. 41, no. 4, p. 604–614. [Also available at https://doi.org/10.1111/j.0021-8901.2004.00937.x.]

Fuhlendorf, S.D., Harrell, W.C., Engle, D.M., Hamilton, R.G., Davis, C.A., and Leslie, D.M., Jr., 2006, Should heterogeneity be the basis for conservation? Grassland bird response to fire and grazing: Ecological Applications, v. 16, no. 5, p. 1706–1716. [Also available at https://dx.doi.org/10.1890/1051-0761(2006)016%5B1706:SHBTBF%5D2.0.CO;2.]

Gannon, J.J., Shaffer, T.L., and Moore, C.T., 2013, Native prairie adaptive management—A multi region adaptive approach to invasive plant management on Fish and Wildlife Service owned native prairies: U.S. Geological Survey Open-File Report 2013–1279, 184 p. with appendixes. [Also available at https://dx.doi.org/10.3133/ofr20131279.]

Gauthier, D.A., and Wiken, E.B., 2003, Monitoring the conservation of grassland habitats, prairie ecozone, Canada: Environmental Monitoring and Assessment, v. 88, no. 1–3, p. 343–364. [Also available at https://doi.org/10.1023/A:1025585527169.]

Gibson, D.J., and Hulbert, L.C., 1987, Effects of fire, topography and year-to-year climate variation on species composition in tallgrass prairie: Vegetatio, v. 72, no. 3, p. 175–185.

Gillihan, S.W., Hanni, D.J., Hutchings, S.W., Toombs, T., and VerCauteren, T., 2001, Sharing your land with shortgrass prairie birds: Brighton, Colo., Rocky Mountain Bird Observatory, 35 p.

Gleason, R.A., Laubhan, M.K., and Euliss, N.H., Jr., eds., 2008, v. 1745. Ecosystem services derived from wetland conservation practices in the United States Prairie Pothole Region with an emphasis on the U.S. Department of Agriculture Conservation Reserve and Wetlands Reserve Programs, U.S. Geological Survey Professional Paper, 58 p. [Also available at https://doi.org/10.3133/pp1745.]

Goguen, C.B., and Mathews, N.E., 1999, Review of the causes and implications of the association between cowbirds and livestock, *in* Hahn, D.C., Hall, L.S., Morrison, M.L., Robinson, S.K., Rothstein, S.I., and Rich, T.D., eds., Research and management of the Brown-headed Cowbird in western landscapes: Studies in Avian Biology, v. 18, p. 10–17.

Goguen, C.B., and Mathews, N.E., 2001, Brown-headed Cowbird behavior and movements in relation to livestock grazing: Ecological Applications, v. 11, no. 5, p. 1533–1544. [Also available at https://doi.org/10.2307/3060937.]

Gorzo, J.M., Pidgeon, A.M., Thogmartin, W.E., Allstadt, A.J., Radeloff, V.C., Heglund, P.J., and Vavrus, S.J., 2016, Using the North American Breeding Bird Survey to assess broad-scale response of the continent's most imperiled avian community, grassland birds, to weather variability: The Condor, v. 118, no. 3, p. 502–512. [Also available at https://doi.org/10.1650/CONDOR-15-180.1.]

Government Accountability Office (GAO), 2005, Wind power—Impacts on wildlife and government responsibilities for regulating development and protecting wildlife: United States Government Accountability Office, Report to Congressional Requesters, GAO-05-906, 60 p., accessed July 2019 at https://www.gao.gov/new.items/d05906.pdf.

Grant, T.A., Flanders-Wanner, B., Shaffer, T.L., Murphy, R.K., and Knutsen, G.A., 2009, An emerging crisis across northern prairie refuges—Prevalence of invasive plants and a plan for adaptive management: Ecological Restoration, v. 27, no. 1, p. 58–65. [Also available at https://doi.org/10.3368/er.27.1.58.]

Grant, T.A., Madden, E., and Berkey, G.B., 2004a, Tree and shrub invasion in northern mixed-grass prairie—Implications for breeding grassland birds: Wildlife Society Bulletin, v. 32, no. 3, p. 807–818.

Grant, T.A., Madden, E.M., Murphy, R.K., Smith, K.A., and Nenneman, M.P., 2004b, Monitoring native prairie vegetation—The belt transect method: Ecological Restoration, v. 22, no. 2, p. 106–112. [Also available at https://dx.doi.org/10.3368/er.22.2.106.]

Grant, T.A., Madden, E.M., Shaffer, T.L., and Dockens, J.S., 2010, Effects of prescribed fire on vegetation and passerine birds in northern mixed-grass prairie: The Journal of Wildlife Management, v. 74, no. 8, p. 1841–1851. [Also available at https://dx.doi.org/10.2193/2010-006.]

Grant, T.A., Madden, E.M., Shaffer, T.L., Pietz, P.J., Berkey, G.B., and Kadrmas, N.J., 2006, Nest survival of Clay-colored and Vesper sparrows in relation to woodland edge in mixed-grass prairies: The Journal of Wildlife Management, v. 70, no. 3, p. 691–701. [Also available at https://dx.doi.org/10.2193/0022-541X(2006)70%5B691:NSOCAV%5D2.0.CO;2.]

Grant, T.A., and Murphy, R.K., 2005, Changes in woodland cover on prairie refuges in North Dakota, USA: Natural Areas Journal, v. 25, no. 4, p. 359–368.

Greenwood, R.J., Sargeant, A.B., Johnson, D.H., Cowardin, L.M., and Shaffer, T.L., 1995, Factors associated with duck nest success in the Prairie Pothole Region of Canada: Wildlife Monographs, v. 128, p. 1–57.

Griffith, P.W., 1976, Welcome to the shelterbelt symposium, *in* Tinus, R.W., ed., Shelterbelts on the Great Plains: Proceedings of the Symposium, Denver, Colo., Great Plains Agricultural Council, Publication no. 78, p. 3–7.

Haila, Y., 2002, A conceptual geneaology of fragmentation research—From island biogeography to landscape ecology: Ecological Applications, v. 12, no. 2, p. 321–334. [Also available at https://doi.org/10.1890/1051-0761(2002)012[0321:ACGOFR]2.0.CO;2.]

Haire, S.L., Bock, C.E., Cade, B.S., and Bennett, B.C., 2000, The role of landscape and habitat characteristics in limiting abundance of grassland nesting songbirds in an urban open space: Landscape and Urban Planning, v. 48, no. 1–2, p. 65–82. [Also available at https://dx.doi.org/10.1016/S0169-2046(00)00044-X.]

Hamerstrom, F.N., Jr., Mattson, O.E., and Hamerstrom, F., 1957, A guide to prairie chicken management: Madison, Wisc., Wisconsin Conservation Department, Technical Wildlife Bulletin, no. 15, 128 p.

Hamilton, L.E., Dale, B.C., and Paszkowski, C.A., 2011, Effects of disturbance associated with natural gas extraction on the occurrence of three grassland songbirds: Avian Conservation and Ecology, v. 6, no. 1, article 7. [Also available at https://dx.doi.org/10.5751/ACE-00458-060107.]

Hands, H.M., Drobney, R.D., and Ryan, M.R., 1989, Status of the Loggerhead Shrike in the northcentral United States: Columbia, Mo., University of Missouri, Missouri Cooperative Fish and Wildlife Research Unit, Report prepared for U.S. Fish and Wildlife Service, Fort Snelling, Minn., 15 p.

Hanks, S.H., 1976, Welcome to the shelterbelt symposium, *in* Tinus, R.W., ed., Shelterbelts on the Great Plains: Proceedings of the Symposium, Denver, Colo., Great Plains Agricultural Council, Publication no. 78, p. 2.

Harrison, K.G., 1974, Aspects of habitat selection in grassland birds: Kalamazoo, Mich., Western Michigan University, Master's Thesis, 82 p.

Hartley, M.J., 1994, Passerine abundance and productivity indices in grasslands managed for waterfowl nesting cover: Transactions of the North American Wildlife and Natural Resources Conference, v. 59, p. 322–327.

Hartnett, D.C., Steuter, A.A., and Hickman, K.R., 1997, Comparative ecology of native and introduced ungulates, *in* Knopf, F.L., and Samson, F.B., eds., Ecology and Conservation of Great Plains Vertebrates: New York, N.Y., Springer, Ecological Studies, v. 125, p. 72–101. [Also available at https://doi.org/10.1007/978-1-4757-2703-6_4.]

Hartnett, D.C., and Wilson, G.W.T., 1999, Mycorrhizae influence plant community structure and diversity in tallgrass prairie: Ecology, v. 80, no. 4, p. 1187–1195. [Also available at https://doi.org/10.1890/0012-9658(1999)080[1187:MIPCSA]2.0.CO;2.]

Heard, L.P., Allen, A.W., Best, L.B., Brady, S.J., Burger, W., Esser, A.J., Hackett, E., Johnson, D.H., Pederson, R.L., Reynolds, R.E., Rewa, C., Ryan, M.R., Molleur, R.T., and Buck, P., 2000, A comprehensive review of Farm Bill contributions to wildlife conservation, 1985–2000: Washington, D.C., U.S. Department of Agriculture, Natural Resources Conservation Service, Wildlife Habitat Management Institute, Technical Report USDA/NRCS/WHMI-2000, 208 p.

Henderson, R.A., and Epstein, E.J., 1995, Oak savannas in Wisconsin, *in* LaRoe, E.T., Farris, G.S., Puckett, C.E., Doran, P.D., and Mac, M.J., eds., Our living resources—A report to the nation on the distribution, abundance, and health of U.S. plants, animals, and ecosystems: Washington, D.C., U.S. Department of the Interior, National Biological Service, p. 230–232.

Herkert, J.R., 1994, The effects of habitat fragmentation on Midwestern grassland bird communities: Ecological Applications, v. 4, no. 3, p. 461–471. [Also available at https://dx.doi.org/10.2307/1941950.]

Herkert, J.R., 1997a, Bobolink (*Dolychonix oryzivorous*) population decline in agricultural landscapes in the midwestern USA: Biological Conservation, v. 80, no. 1, p. 107–112. [Also available at https://doi.org/10.1016/S0006-3207(96)00066-3.]

Herkert, J.R., 1997b, Population trends of the Henslow's Sparrow in relation to the Conservation Reserve Program in Illinois, 1975–1995: Journal of Field Ornithology, v. 68, no. 2, p. 235–244.

Herkert, J.R., 1998, The influence of CRP on Grasshopper Sparrow population trends in the mid-continental United States: Wildlife Society Bulletin, v. 26, no. 2, p. 227–231.

Herkert, J.R., Reinking, D.L., Wiedenfeld, D.A., Winter, M., Zimmerman, J.L., Jensen, W.E., Finck, E.J., Koford, R.R., Wolfe, D.H., Sherrod, S.K., Jenkins, M.A., Faaborg, J., and Robinson, S.K., 2003, Effects of prairie fragmentation on the nest success of breeding birds in the midcontinental United States: Conservation Biology, v. 17, no. 2, p. 587–594. [Also available at https://dx.doi.org/10.1046/j.1523-1739.2003.01418.x.]

Herkert, J.R., Sample, D.W., and Warner, R.E., 1996, Management of midwestern grassland landscapes for the conservation of migratory birds, *in* Thompson, F.R., III, ed., Management of midwestern landscapes for the conservation of Neotropical migratory birds: St. Paul, Minn., U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station, General Technical Report GTR-NC-187, p. 89–116.

Herkert, J.R., Szafoni, R.E., Kleen, V.M., and Schwegman, J.E., 1993, Habitat establishment, enhancement and management for forest and grassland birds in Illinois: Springfield, Ill., Illinois Department of Conservation, Division of Natural Heritage, Natural Heritage Technical Publication 1, 20 p.

Higgins, K.F., 1986, A comparison of burn season effects on nesting birds in North Dakota mixed-grass prairie: Prairie Naturalist, v. 18, no. 4, p. 219–228.

Higgins, K.F., Naugle, D.E., and Forman, K.J., 2002, A case study of changing land use practices in the Northern Great Plains, U.S.A.—An uncertain future for waterbird conservation: Waterbirds, v. 25, Special Publication 2, p. 42–50.

Hill, J.M., Egan, J.F., Stauffer, G.E., and Diefenbach, D.R., 2014, Habitat availability is a more plausible explanation than insecticide acute toxicity for U.S. grassland bird species declines: PLoS One, v. 9, no. 5, p. e98064. [Also available at https://doi.org/10.1371/journal.pone.0098064.]

Hobbs, R.J., and Huenneke, L.F., 1992, Disturbance, diversity, and invasion—Implications for conservation: Conservation Biology, v. 6, no. 3, p. 324–337. [Also available at https://doi.org/10.1046/j.1523-1739.1992.06030324.x.]

Hoekstra, J.M., Boucher, T.M., Ricketts, T.H., and Roberts, C., 2005, Confronting a biome crisis—Global disparities of habitat loss and protection: Ecology Letters, v. 8, no. 1, p. 23–29. [Also available at https://doi.org/10.1111/j.1461-0248.2004.00686.x.]

Hornaday, W.T., 1889, Extermination of the North American Bison with a sketch of its discovery and life history, *in* a Report of the National Museum under the direction of the Smithsonian Institution 1887: Washington, D.C., Government Printing Office, p. 367–548.

Houston, C.S., and Bechard, M.J., 1984, Decline of the Ferruginous Hawk in Saskatchewan: American Birds, v. 38, no. 2, p. 166–170.

Hovick, T.J., Elmore, R.D., Fuhlendorf, S.D., Engle, D.M., and Hamilton, R.G., 2015, Spatial heterogeneity increases diversity and stability in grassland bird communities: Ecological Applications, v. 25, no. 3, p. 662–672. [Also available at https://dx.doi.org/10.1890/14-1067.1.]

Hovick, T.J., Miller, J.R., Dinsmore, S.J., Engle, D.M., Debinski, D.M., and Fuhlendorf, S.D., 2012, Effects of fire and grazing on Grasshopper Sparrow nest survival: The Journal of Wildlife Management, v. 76, no. 1, p. 19–27. [Also available at https://dx.doi.org/10.1002/jwmg.243.]

Hubbard, D.E., 1982, Breeding birds in two dry wetlands in eastern South Dakota: Prairie Naturalist, v. 14, no. 1, p. 6–8.

Hultquist, J.M., and Best, L.B., 2001, Bird use of terraces in Iowa rowcrop fields: American Midland Naturalist, v. 145, no. 2, p. 275–287. [Also available at https://doi.org/10.1674/0003-0031(2001)145[0275:BUOTII]2.0.CO;2.]

Igl, L.D., 2009, Breeding bird use of grasslands enrolled in the Conservation Reserve Program in the northern Great Plains: Fargo, N. Dak., North Dakota State University, Ph.D. Dissertation, 199 p.

Igl, L.D., and Johnson, D.H., 1997, Changes in breeding bird populations in North Dakota—1967 to 1992–93: The Auk, v. 114, no. 1, p. 74–92. [Also available at https://doi.org/10.2307/4089067.]

Igl, L.D., and Johnson, D.H., 1999, Le Conte's Sparrows breeding in Conservation Reserve Program fields—Precipitation and patterns of population change, *in* Vickery, P.D., and Herkert, J.R., eds., Ecology and conservation of grassland birds of the Western Hemisphere: Studies in Avian Biology, v. 19, p. 178–186.

Igl, L.D., and Johnson, D.H., 2007, Brown-headed Cowbird, *Molothrus ater*, parasitism and abundance in the northern Great Plains: Canadian Field-Naturalist, v. 121, no. 3, p. 239–255. [Also available at https://dx.doi.org/10.22621/cfn.v121i3.471.]

Ingold, D.J., 2002, Use of a reclaimed stripmine by grassland nesting birds in east-central Ohio: The Ohio Journal of Science, v. 102, no. 3, p. 56–62.

Innes, R.J., 2016, *Centrocercus minimus, C. urophasianus, in* Fire Effects Information System (FEIS): U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Missoula Fire Sciences Laboratory, accessed July 2019 at https://www.fs.fed.us/database/feis/animals/bird/cent/all.html.

James, P.C., and Fox, G.A., 1987, Effects of some insecticides on productivity of Burrowing Owls: Blue Jay, v. 45, no. 2, p. 65–71.

Jensen, H.P., Rollins, D., and Gillen, R.L., 1990, Effects of cattle stock density on trampling loss of simulated ground nests: Wildlife Society Bulletin, v. 18, no. 1, p. 71–74.

Jensen, W.E., and Finck, E.J., 2004, Edge effects on nesting Dickcissels (*Spiza americana*) in relation to edge type of remnant tallgrass prairie in Kansas: American Midland Naturalist, v. 151, no. 1, p. 192–199. [Also available at https://dx.doi.org/10.1674/0003-0031(2004)151%5B0192:EEONDS%5D2.0.CO;2.]

Joern, A., and Keeler, K.H., 1995, Getting the lay of the land—Introducing North American native grasslands, *in* Joern, A., and Keeler, K.H., eds., The changing prairie—The North American grassland: New York, N.Y., Oxford University Press, p. 11–24.

Johnsgard, P.A., 2001, Prairie birds—Fragile splendor in the Great Plains: Lawrence, Kans., University Press of Kansas, 331 p.

Johnsgard, P.A., 2005, Prairie dog empire—A saga of the shortgrass prairie: Lincoln, Nebr., University of Nebraska Press, 243 p.

Johnston, C.A., 2013, Wetland losses due to row crop expansion in the Dakota Prairie Pothole Region: Wetlands, v. 33, no. 1, p. 175–182. [Also available at https://doi.org/10.1007/s13157-012-0365-x.]

Johnson, D.H., 1996, Management of northern prairies and wetlands for the conservation of Neotropical migratory birds, *in* Thompson, F.R., III, ed., Management of midwestern landscapes for the conservation of Neotropical migratory birds: St. Paul, Minn., U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station, General Technical Report NC-187, p. 53–67.

Johnson, D.H., 2001, Habitat fragmentation effects on birds in grasslands and wetlands—A critique of our knowledge: Great Plains Research, v. 11, no. 2, p. 211–231.

Johnson, D.H., 2002, The importance of replication in wildlife research: The Journal of Wildlife Management, v. 66, no. 4, p. 919–932. [Also available at https://doi.org/10.2307/3802926.]

Johnson, D.H., Holloran, M.J., Connelly, J.W., Hanser, S.E., Amundson, C.L., and Knick, S.T., 2011a, Influences of environmental and anthropogenic features on Greater Sage-Grouse populations, 1997–2007, *in* Knick, S.T, and Connelly, J.W., eds., Greater Sage-Grouse—Ecology and conservation of a landscape species and its habitats: Berkeley, Calif., University of California Press, Studies in Avian Biology, v. 38, p. 407–450. [Also available at https://doi.org/10.1525/california/9780520267114.003.0018.]

Johnson, D.H., and Igl, L.D., 1995, Contributions of the Conservation Reserve Program to populations of breeding birds in North Dakota: The Wilson Bulletin, v. 107, no. 4, p. 709–718.

Johnson, D.H., and Igl, L.D., 2001, Area requirements of grassland birds—A regional perspective: The Auk, v. 118, no. 1, p. 24–34. [Also available at https://dx.doi.org/10.2307/4089756.]

Johnson, D.H., and Schwartz, M.D., 1993a, The Conservation Reserve Program and grassland birds: Conservation Biology, v. 7, no. 4, p. 934–937. [Also available at https://dx.doi.org/10.1046/j.1523-1739.1993.740934.x.]

Johnson, D.H., and Schwartz, M.D., 1993b, The Conservation Reserve Program—Habitat for grassland birds: Great Plains Research, v. 3, no. 2, p. 273–295.

Johnson, D.H., and Winter, M., 1999, Reserve design for grasslands: considerations for bird populations, *in* Harmon, D., ed., On the frontiers of conservation: Proceedings of the tenth Conference on Research and Resource Management in Parks and on Public Lands, Hancock, Mich., George Wright Society, p. 391–396.

Johnson, R.G., and Temple, S.A., 1990, Nest predation and brood parasitism of tallgrass prairie birds: The Journal of Wildlife Management, v. 54, no. 1, p. 106–111. [Also available at https://dx.doi.org/10.2307/3808909.]

Johnson, R.R., Granfors, D.A., Niemuth, N.D., Estey, M.E., and Reynolds, R.E., 2010, Delineating Grassland Bird Conservation Areas in the U.S. Prairie Pothole Region: Journal of Fish and Wildlife Management, v. 1, no. 1, p. 38–42. [Also available at https://doi.org/10.3996/JFWM-022.]

Johnson, R.R., Higgins, K.F., Kjellsen, M.L., and Elliott, C.R., 1997, Eastern South Dakota wetlands: Brookings, S. Dak., South Dakota State University, 28 p.

Johnson, T., Kennedy, P.L., DelCurto, T., and Taylor, R.V., 2011b, Bird community responses to cattle stocking rates in a Pacific Northwest bunchgrass prairie: Agriculture, Ecosystems and Environment, v. 144, no. 1, p. 338–346. [Also available at https://doi.org/10.1016/j.agee.2011.10.003.]

Jones, A., 2000, Effects of cattle grazing on North American arid ecosystems—A quantitative review: Western North American Naturalist, v. 60, no. 2, p. 155–164.

Jones, S.R., and Cushman, R.C., 2004, The North American prairie: New York, N.Y., Houghton Mifflin Company, 510 p.

Kantrud, H.A., 1981, Grazing intensity effects on the breeding avifauna of North Dakota native grasslands: Canadian Field-Naturalist, v. 95, no. 4, p. 404–417.

Kantrud, H.A., 1992, History of cattails on the prairies—Wildlife impacts, *in* Linz, G.M., ed., Proceedings of the Cattail Management Symposium: Fargo, N. Dak., North Dakota State University, p. 9–12.

Kantrud, H.A., and Kologiski, R.L., 1982, Effects of soils and grazing on breeding birds of uncultivated upland grasslands of the northern Great Plains: Washington, D.C., U.S. Fish and Wildlife Service, Wildlife Research Report 15, 33 p.

Kantrud, H.A., and Stewart, R.E., 1984, Ecological distribution and crude density of breeding birds on prairie wetlands: The Journal of Wildlife Management, v. 48, no. 2, p. 426–437. [Also available at https://dx.doi.org/10.2307/3801174.]

Kay, C.E., 1998, Are ecosystems structured from the top-down or bottom-up—A new look at an old debate: Wildlife Society Bulletin, v. 26, no. 3, p. 484–498.

Kerns, C.K., Ryan, M.R., Murphy, R.K., Thompson, F.R., III, and Rubin, C.S., 2010, Factors affecting songbird nest survival in northern mixed-grass prairie: The Journal of Wildlife Management, v. 74, no. 2, p. 257–264. [Also available at https://doi.org/10.2193/2008-249.]

Kershner, E.L., and Bollinger, E.K., 1996, Reproductive success of grassland birds at east-central Illinois airports: American Midland Naturalist, v. 136, no. 2, p. 358–366. [Also available at https://dx.doi.org/10.2307/2426740.]

Kirby, R.E., Ringelman, J.K., Anderson, D.R., and Sojda, R.S., 1992, Grazing on national wildlife refuges—Do the needs outweigh the problems?: Transactions of the North American Wildlife and Natural Resources Conference, v. 57, p. 611–626.

Kirsch, L.M., Duebbert, H.F., and Kruse, A.D., 1978, Grazing and haying effects of upland nesting birds: Transactions of the North American Wildlife and Natural Resource Conference, v. 43, p. 486–497.

Knapp, A.K., Blair, J.M., Briggs, J.M., Collins, S.L., Hartnett, D.C., Johnson, L.C., and Towne, E.G., 1999, The keystone role of bison in North American tallgrass prairie: Bioscience, v. 49, no. 1, p. 39–50. [Also available at https://doi.org/10.2307/1313492.]

Knick, S.T., 2011, Historical development, principal federal legislation, and current management of sagebrush habitats—Implications for conservation, *in* Knick, S.T, and Connelly, J.W., eds., Greater Sage-Grouse—Ecology and conservation of a landscape species and its habitats: Berkeley, Calif., University of California Press, Studies in Avian Biology, v. 38, p. 13–31. [Also available at https://doi.org/10.1525/california/9780520267114.003.0002.]

Knick, S.T., Hanser, S.E., Miller, R.F., Pyke, D.A., Wisdom, M.J., Finn, S.P., Rinkes, E.T., and Henny, C.J., 2011, Ecological influence and pathways of land use in sagebrush, *in* Knick, S.T., and Connelly, J.W., eds., Greater Sage-Grouse—Ecology and conservation of a landscape species and its habitats: Berkeley, Calif., University of California Press, Studies in Avian Biology, v. 38, p. 203–251. [Also available at https://doi.org/10.1525/california/9780520267114.003.0013.]

Knopf, F.L., 1986, Changing landscapes and the cosmopolitism of the eastern Colorado avifauna: Wildlife Society Bulletin, v. 14, no. 2, p. 132–142.

Knopf, F.L., 1988, Conservation of steppe birds in North America, *in* Goriup, P.D., ed., Ecology and conservation of grassland birds: Cambridge, United Kingdom, International Council for Bird Preservation, Technical Publication 7, p. 27–41.

Knopf, F.L., 1993, Perspectives on grazing non-game bird habitats, *in* Krausman, P.R., ed., Rangeland wildlife: Society for Range Management, p. 51–58.

Knopf, F.L., 1994, Avian assemblages on altered grasslands, *in* Jehl, J.R., Jr., and Johnson, N.K., eds., A century of avifaunal change in western North America: Studies in Avian Biology, v. 15, p. 247–257.

Knutsen, G.A., and Euliss, N.H., Jr., 2001, Wetland restoration in the Prairie Pothole Region of North America—A literature review: Reston, Va., U.S. Geological Survey, Biological Science Report 2001–0006, 54 p.

Koper, N., and Schmiegelow, F.K.A., 2006, Effects of habitat management for ducks on target and nontarget species: Journal of Wildlife Management, v. 70, no. 3, p. 823–834. [Also available at https://doi.org/10.2193/0022-541X(2006)70[823:EOHMFD]2.0.CO;2.]

Kriz, M., 2007, Corn power: National Journal, v. 39, no. 15, p. 20–26.

Küchler, A.W., 1964, Potential natural vegetation of the conterminous United States: New York, N.Y., American Geographical Society, Special Publication 36, 116 p.

Kushlan, J.A., Steinkamp, M.J., Parsons, K.C., Capp, J., Acosta Cruz, M., Coulter, M., Davidson, I., Dickson, L., Edelson, N., Elliot, R., Erwin, R.M., Hatch, S., Kress, S., Milko, R., Miller, S., Mills, K., Paul, R., Phillips, R., Saliva, J.E., Sydeman, B., Trapp, J., Wheeler, J., and Wohl, K., 2002, Waterbird conservation for the Americas—The North American Waterbird Conservation Plan (ver. 1.0): Washington, D.C., Waterbird Conservation for the Americas, 78 p., accessed July 2019 at https://iwjv.org/resource/north-american-waterbird-conservation-plan.

Kuvlesky, W.P., Jr., Brennan, L.A., Morrison, M.L., Boydston, K.K., Ballard, B.M., and Bryant, F.C., 2007, Wind energy development and wildlife conservation—Challenges and opportunities: The Journal of Wildlife Management, v. 71, no. 8, p. 2487–2498. [Also available at https://doi.org/10.2193/2007-248.]

Lahti, D.C., 2001, The “edge effect on nest predation” hypothesis after twenty years: Biological Conservation, v. 99, no. 3, p. 365–374. [Also available at https://doi.org/10.1016/S0006-3207(00)00222-6.]

Lark, T.J., Salmon, J.M., and Gibbs, H.K., 2015, Cropland expansion outpaces agricultural biofuel policies in the United States: Environmental Research Letters, v. 10, no. 4, p. 044003. [Also available at https://doi.org/10.1088/1748-9326/10/4/044003.]

Lauenroth, W.K., Milchunas, D.G., Dodd, J.L., Hart, R.H., Heitschmidt, R.K., and Rittenhouse, L.R., 1994, *in* Vavra, M., Laycock, W.A., and Pieper, R.D., eds., Ecological implications of livestock herbivory in the West: Denver, Colo., Society for Range Management, p. 69–100.

Larson, D.L., Euliss, N., Lannoo, M.J., and Mushet, D.M., 1998, Amphibians of the northern grasslands, *in* Mac, M.J., Opler, P.A., Puckett Haecker, C.E., and Doran, P.D., eds., Status and trends of the nation’s biological resources v. 2: Reston, Va., U.S. Department of the Interior, U.S. Geological Survey, p. 450–451.

LeBeau, C.W., Beck, J.L., Johnson, G.D., and Holloran, M.J., 2014, Short-term impacts of wind energy development on Greater Sage-Grouse fitness: The Journal of Wildlife Management, v. 78, no. 3, p. 522–530. [Also available at https://doi.org/10.1002/jwmg.679.]

Lenth, B.A., Knight, R.L., and Gilgert, W.C., 2006, Conservation value of clustered housing developments: Conservation Biology, v. 20, no. 5, p. 1445–1456. [Also available at https://doi.org/10.1111/j.1523-1739.2006.00491.x.]

Lesica, P., and DeLuca, T.H., 1996, Long-term harmful effects of crested wheatgrass on Great Plains grassland ecosystems: Journal of Soil and Water Conservation, v. 51, no. 5, p. 408–409.

Leston, L.F.V., and Koper, N., 2017, Managing urban and rural rights-of-way as potential habitats for grassland birds: Avian Conservation and Ecology, v. 12, no. 2, article 4. [Also avail-able at https://doi.org/10.5751/ACE-01049-120204.]

Lewis, M.G., 1966, William Gilpin and the concept of the Great Plains region: Annals of the Association of American Geographers, v. 56, no. 1, p. 33–51. [Also available at https://doi.org/10.1111/j.1467-8306.1966.tb00542.x.]

Lewis, H.T., 1985, Why Indians burned—Specific versus general reasons, *in* Lotan, J.E., Kilgore, B.M., Fischer, W.C., and Mutch, R.W., eds., Proceedings of the Symposium and Workshop on Wilderness Fire: Ogden, Utah, U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, General Technical Report INT-182, p. 75–80.

Linnen, C.G., 2008, Effects of oil and gas development on grassland birds: Saskatoon, Saskatchewan, Northern EnviroSearch Ltd., Report prepared for Petroleum Technology Alliance Canada, 26 p.

Lloyd, J.D., and Martin, T.E., 2005, Reproductive success of Chestnut-collared Longspurs in native and exotic grassland: The Condor, v. 107, no. 2, p. 363–374. [Also available at https://doi.org/10.1650/7701.]

Lockhart, J., and Koper, N., 2018, Northern prairie songbirds are more strongly influenced by grassland configuration than grassland amount: Landscape Ecology, v. 33, no. 9, p. 1543–1558. [Also available at https://doi.org/10.1007/s10980-018-0681-5.]

Loesch, C.R., Walker, J.A., Reynolds, R.E., Gleason, J.S., Niemuth, N.D., Stephens, S.E., and Erickson, M.A., 2013, Effect of wind energy development on breeding duck densities in the Prairie Pothole Region: The Journal of Wildlife Management, v. 77, no. 3, p. 587–598. [Also available at https://doi.org/10.1002/jwmg.481.]

Lockwood, J.A., 2004, Locust—The devastating rise and mysterious disappearamce of the insect that shaped the American frontier: New York, N.Y., Basic Books, 294 p.

Lokemoen, J.T., and Beiser, J.A., 1997, Bird use and nesting in conventional, minimum-tillage, and organic cropland: The Journal of Wildlife Management, v. 61, no. 3, p. 644–655. [Also available at https://dx.doi.org/10.2307/3802172.]

Ludlow, S.M., Brigham, R.M., and Davis, S.K., 2015, Oil and natural gas development has mixed effects on the density and reproductive success of grassland songbirds: The Condor, v. 117, no. 1, p. 64–75. [Also available at https://dx.doi.org/10.1650/CONDOR-14-79.1.]

Lusk, J., and Koper, N., 2013, Grazing and songbird nest survival in southwestern Saskatchewan: Rangeland Ecology and Management, v. 66, no. 4, p. 401–409. [Also available at https://doi.org/10.2111/REM-D-11-00203.1.]

Mabey, S., and Paul, E., 2007, Critical literature review—Impact of wind energy and related human activities on grassland and shrub-steppe birds: The Ornithological Council, Prepared for the National Wind Coordinating Collaborative, 115 p., accessed July 2019 at https://nationalwind.org/wp-content/uploads/2013/05/Critical-Lit-Review-Oct-2007.pdf.

Mack, R.N., 1981, Invasion of *Bromus tectorum* L. into western North America—An ecological chronicle: Agro-ecosystems, v. 7, no. 2, p. 145–165. [Also available at https://doi.org/10.1016/0304-3746(81)90027-5.]

Mack, R.N., and Thompson, J.N., 1982, Evolution in steppe with few large, hooved mammals: American Naturalist, v. 119, no. 6, p. 757–773. [Also available at https://dx.doi.org/10.1086/283953.]

Madden, E.M., 1996, Passerine communities and bird-habitat relationships on prescribe-burned, mixed-grass prairie in North Dakota: Bozeman, Mont., Montana State University, Master's Thesis, 153 p.

Madden, E.M., Murphy, R.K., Hansen, A.J., and Murray, L., 2000, Models for guiding management of prairie bird habitat in northwestern North Dakota: American Midland Naturalist, v. 144, no. 2, p. 377–392. [Also available at https://dx.doi.org/10.1674/0003-0031(2000)144%5B0377:MFGMOP%5D2.0.CO;2.]

Maddox, J.D., and Wiedenmann, R.N., 2005, Nesting of birds in wetlands containing purple loosestrife (*Lythrum salicaria*) and cattail (*Typha* spp.): Natural Areas Journal, v. 25, no. 4, p. 369–373.

Mahoney, A., and Chalfoun, A.D., 2016, Reproductive success of Horned Lark and McCown's Longspur in relation to wind energy infrastructure: The Condor, v. 118, no. 2, p. 360–375. [Also available at https://doi.org/10.1650/CONDOR-15-25.1.]

Manske, L.L., 1995, Adaptive tolerance mechanisms in grass plants: Dickinson, N. Dak., North Dakota State University, Dickinson Research Extension Center, Range management report, 3 p.

Marra, P.P., and Santella, C., 2016, Cat wars—The devastating consequences of a cuddly killer: Princeton, N.J., Princeton University Press, 216 p.

Marsh, R.E., 1984, Ground squirrels, prairie dogs, and marmots as pests on rangeland, *in* Proceedings of the Conference for Organization and Practice of Vertebrate Pest Control: Fernherst, England, ICI Plant Protection Division, v. 30, p. 195–208.

Martin, S.G., 1971, Polygyny in the Bobolink—Habitat quality and the adaptive complex: Corvallis, Oreg., Oregon State University, Ph.D. Dissertation, 181 p.

Martin, P.A., and Forsyth, D.J., 2003, Occurrence and productivity of songbirds in prairie farmland under conventional versus minimum tillage regimes: Agriculture, Ecosystems and Environment, v. 96, no. 1–3, p. 107–117. [Also available at https://dx.doi.org/10.1016/S0167-8809(02)00234-7.]

Martin, P.A., Johnson, D.L., Forsyth, D.J., and Hill, B.D., 1998, Indirect effects of the pyrethroid insecticide deltamethrin on reproductive success of chestnut-collared longspurs: Ecotoxicology (London, England), v. 7, no. 2, p. 89–97. [Also available at https://doi.org/10.1023/A:1008815903340.]

Martin, P.A., Johnson, D.L., Forsyth, D.J., and Hill, B.D., 2000, Effects of two grasshopper control insecticides on food resources and reproductive success of two species of grassland songbirds: Environmental Toxicology and Chemistry, v. 19, no. 12, p. 2987–2996. [Also available at https://doi.org/10.1002/etc.5620191220.]

Marzluff, J.M., and Ewing, K., 2001, Restoration of fragmented landscapes for the conservation of birds—A general framework and specific recommendations for urbanizing landscapes: Restoration Ecology, v. 9, no. 3, p. 280–292. [Also available at https://doi.org/10.1046/j.1526-100x.2001.009003280.x.]

May, S.M., Naugle, D.E., and Higgins, K.F., 2002, Effects of land use on nongame wetland birds in western South Dakota stock ponds, U.S.A: Waterbirds, v. 25, Special Publication 2, p. 51–55.

McConnell, M., and Burger, L.W., 2011, Precision conservation—A geospatial decision support tool for optimizing conservation and profitability in agricultural landscapes: Journal of Soil and Water Conservation, v. 66, no. 6, p. 347–354. [Also available at https://doi.org/10.2489/jswc.66.6.347.]

McCoy, T.D., Ryan, M.R., Kurzejeski, E.W., and Burger, L.W., Jr., 1999, Conservation reserve program—Source or sink habitat for grassland birds in Missouri?: The Journal of Wildlife Management, v. 63, no. 2, p. 530–538. [Also available at https://dx.doi.org/10.2307/3802639.]

McEwen, L.C., Knittle, C.E., and Richmond, M.L., 1972, Wildlife effects from grasshopper insecticides sprayed on short-grass range: Journal of Range Management, v. 25, no. 3, p. 188–194. [Also available at https://dx.doi.org/10.2307/3897053.]

McGarigal, K., and Cushman, S.A., 2002, Comparative evaluation of experimental approaches to the study of habitat fragmentation effects: Ecological Applications, v. 12, no. 2, p. 335–345. [Also available at https://doi.org/10.1890/1051-0761(2002)012[0335:CEOEAT]2.0.CO;2.]

McLaughlin, M.E., Janousek, W.M., McCarty, J.P., and Wolfenbarger, L.L., 2014, Effects of urbanization on site occupancy and density of grassland birds in tallgrass prairie fragments: Journal of Field Ornithology, v. 85, no. 3, p. 258–273. [Also available at https://dx.doi.org/10.1111/jofo.12066.]

McMaster, D.G., and Davis, S.K., 2001, An evaluation of Canada's Permanent Cover Program—Habitat for grassland birds?: Journal of Field Ornithology, v. 72, no. 2, p. 195–210. [Also available at https://dx.doi.org/10.1648/0273-8570-72.2.195.]

McNicholl, M.K., 1988, Ecological and human influences on Canadian populations of grassland birds, *in* Goriup, P.D., ed., Ecology and conservation of grassland birds: Cambridge, United Kingdom, International Council for Bird Preservation, Technical Publication 7, p. 1–25.

McPherson, G.R., 1997, Ecology and management of North American savannas: Tucson, Ariz., The University of Arizona Press, 208 p.

Meinke, C.W., Knick, S.T., and Pyke, D.A., 2009, A spatial model to prioritize sagebrush landscapes in the Intermountain West (U.S.A.) for restoration: Restoration Ecology, v. 17, no. 5, p. 652–659. [Also available at https://doi.org/10.1111/j.1526-100X.2008.00400.x.]

Mengel, R.M., 1970, The North American Central Plains as an isolating agent in bird speciation, *in* Dort, W., Jr., and Jones, J.K., Jr., eds., Pleistocene and recent environments of the Central Great Plains: Lawrence, Kans., University Press of Kansas, p. 279–340.

Messmer, T.A., 1990, Influence of grazing treatments on nongame birds and vegetation structure in south central North Dakota: Fargo, N. Dak., North Dakota State University, Ph.D. Dissertation, 164 p.

Michaels, H.L., 1997, Landscape and fine scale habitat associations of the Loggerhead Shrike and Henslow's Sparrow on Fort Riley Military Installation, Kansas: Manhattan, Kans., Kansas State University, Master's Thesis, 109 p.

Milchunas, D.G., Sala, O.E., and Lauenroth, W.K., 1988, A generalized model of the effects of grazing by large herbivores on grassland community structure: American Naturalist, v. 132, no. 1, p. 87–106. [Also available at https://dx.doi.org/10.1086/284839.]

Miller, B., Ceballos, G., and Reading, R., 1994, The prairie dog and biotic diversity: Conservation Biology, v. 8, no. 3, p. 677–681. [Also available at https://doi.org/10.1046/j.1523-1739.1994.08030677.x.]

Miller, D., 2008, Busting the CRP: Progressive Farmer, v. 123, no. 5, p. 44–48.

Miller, R.F., Knick, S.T., Pyke, D.A., Meinke, C.W., Hanser, S.E., Wisdom, M.J., and Hild, A.L., 2011, Characteristics of sagebrush habitats and limitations to long-term conservation, *in* Knick, S.T, and Connelly, J.W., eds., Greater Sage-Grouse: Berkeley, Calif., University of California Press, Studies in Avian Biology, v. 38, p. 145–184. [Also available at https://doi.org/10.1525/california/9780520267114.003.0011.]

Miller, S.G., Knight, R.L., and Miller, C.K., 1998, Influence of recreational trails on breeding bird communities: Ecological Applications, v. 8, no. 1, p. 162–169. [Also available at https://doi.org/10.2307/2641318.]

Mineau, P., and Whiteside, M., 2013, Pesticide acute toxicity is a better correlate of U.S. grassland bird declines than agricultural intensification: PLoS One, v. 8, no. 2, p. e57457. [Also available at https://doi.org/10.1371/journal.pone.0057457.]

Monsen, S.B., 1994, The competitive influences of cheatgrass (*Bromus tectorum*) on site restoration, *in* Monsen, S.B., and Kitchen, S.G., compilers, Proceedings of the Ecology and Management of Annual Rangelands Symposium: Ogden, Utah, U.S. Department of Agriculture, Forest Service, Intermountain Research Station, General Technical Report INT-GTR-313, p. 43–50.

Morefield, P.E., LeDuc, S.D., Clark, C.M., and Iovanna, R., 2016, Grasslands, wetlands, and agriculture: the fate of land expiring from the Conservation Reserve Program in the Midwestern United States: Environmental Research Letters, v. 11, no. 9, p. 004005. [Also available at https://doi.org/10.1088/1748-9326/11/9/094005.]

Mossman, M.J., and Sample, D.W., 1990, Birds of Wisconsin sedge meadows: Passenger Pigeon, v. 52, no. 1, p. 39–55.

Munro, J.W., 2006, Ecological restoration and other conservation practices—The difference: Ecological Restoration, v. 24, no. 3, p. 182–189. [Also available at https://dx.doi.org/10.3368/er.24.3.182.]

Murphy, M.T., 2003, Avian population trends within the evolving agricultural landscape of eastern and central United States: The Auk, v. 120, no. 1, p. 20–34. [Also available at https://doi.org/10.1642/0004-8038(2003)120[0020:APTWTE]2.0.CO;2.]

Murphy, R.K., and Grant, T.A., 2005, Land management history and floristics in mixed-grass prairie, North Dakota, USA: Natural Areas Journal, v. 25, no. 4, p. 351–358.

Mushet, D.M., Neau, J.L., and Euliss, N.H., Jr., 2014, Modeling effects of conservation grassland losses on amphibian habitat: Biological Conservation, v. 174, p. 93–100. [Also available at https://dx.doi.org/10.1016/j.biocon.2014.04.001.]

National Land Trust Alliance, 2015, 2015 National Land Trust Census report: Washington, D.C., Land Trust Alliance, 24 p., accessed July 2019 at https://s3.amazonaws.com/landtrustalliance.org/2015NationalLandTrustCensusReport.pdf.

Naugle, D.E., Higgins, K.F., and Nusser, S.M., 1999, Effects of woody vegetation on prairie wetland birds: Canadian Field-Naturalist, v. 113, no. 3, p. 487–492.

Nenninger, H.R., and Koper, N., 2018, Effects of conventional oil wells on grassland songbird abundance are caused by presence of infrastructure, not noise: Biological Conservation, v. 218, p. 124–133. [Also available at https://doi.org/10.1016/j.biocon.2017.11.014.]

Ng, C.S., 2017, Proximity to conventional oil and gas development is associated with reduced parental care in Chestnut-collared Longspurs (*Calcarius ornatus*): Winnipeg, Manitoba, University of Manitoba, Master's Thesis, 96 p.

Niemuth, N.D., 2000, Land use and vegetation associated with Greater Prairie-Chicken leks in an agricultural landscape: The Journal of Wildlife Management, v. 64, no. 1, p. 278–286. [Also available at https://doi.org/10.2307/3803000.]

Niemuth, N.D., and Boyce, M.S., 1998, Disturbance in Wisconsin pine barrens—Implications for management: Transactions of the Wisconsin Academy of Sciences, Arts, and Letters, v. 86, p. 167–176.

Niemuth, N.D., Estey, M.E., Fields, S.P., Wangler, B., Bishop, A.A., Moore, P.J., Grosse, R.C., and Ryba, A.J., 2017, Developing spatial models to guide conservation of grassland birds in the U.S. northern Great Plains: The Condor, v. 119, no. 3, p. 506–525. [Also available at https://dx.doi.org/10.1650/CONDOR-17-14.1.]

Niemuth, N.D., Estey, M.E., and Loesch, C.R., 2005, Developing spatially explicit habitat models for grassland bird conservation planning in the Prairie Pothole Region of North Dakota, *in* Ralph, C.J., and Rich, T.D., eds., Bird conservation implementation and integration in the Americas: Proceedings of the third International Partners in Flight Conference, Albany, Calif., U.S. Department of Agriculture, Forest Service, General Technical Report PSW-GTR-191, p. 469–477.

Niemuth, N.D., Fleming, K.K., and Reynolds, R.E., 2014, Waterfowl conservation in the US Prairie Pothole Region—Confronting the complexities of climate change: PLoS One, v. 9, no. 6, p. e100034. [Also available at https://doi.org/10.1371/journal.pone.0100034.]

Niemuth, N.D., Solberg, J.W., and Shaffer, T.L., 2008, Influence of moisture on density and distribution of grassland birds in North Dakota: The Condor, v. 110, no. 2, p. 211–222. [Also available at https://dx.doi.org/10.1525/cond.2008.8514.]

Niemuth, N.D., Walker, J.A., Gleason, J.S., Loesch, C.R., Reynolds, R.E., Stephens, S.E., and Erickson, M.A., 2013, Influence of wind turbines on presence of Willet, Marbled Godwit, Wilson's Phalarope and Black Tern on wetlands in the Prairie Pothole Region of North Dakota and South Dakota: Waterbirds, v. 36, no. 3, p. 263–276. [Also available at https://doi.org/10.1675/063.036.0304.]

Nocera, J.J., Parsons, G.J., Milton, G.R., and Fredeen, A.H., 2005, Compatibility of delayed cutting regime with bird breeding and hay nutritional quality: Agriculture, Ecosystems and Environment, v. 107, no. 2–3, p. 245–253. [Also available at https://dx.doi.org/10.1016/j.agee.2004.11.001.]

North American Waterfowl Management Plan, 2012, People conserving waterfowl and wetlands: Canadian Wildlife Service, U.S. Fish and Wildlife Service, Secretaria de Medio Ambiente y Recursos Naturales, 48 p., accessed July 2019 at https://www.fws.gov/migratorybirds/pdf/management/NAWMP/2012NAWMP.pdf.

North Dakota Game and Fish Department [NDGF], 2016, Private lands initiative program: accessed July 2019 at https://gf.nd.gov/private-lands.

Noss, R.F., LaRoe, E.T., III, and Scott, J.M., 1995, Endangered ecosystems of the United States: a preliminary assessment of loss and degradation: Washington, D.C., U.S. Department of the Interior, National Biological Service, Biological Report, no. 28, 58 p.

Nuzzo, V.A., 1986, Extent and status of midwest oak savanna—Presettlement and 1985: Natural Areas Journal, v. 6, no. 2, p. 6–36.

O'Leary, C.H., and Nyberg, D.W., 2000, Treelines between fields reduce the density of grassland birds: Natural Areas Journal, v. 20, no. 3, p. 243–249.

Olson, E., 1997, National grasslands managemement—A primer: Washington, D.C., U.S. Department of Agriculture, Office of the General Counsel, 36 p., accessed July 2019 at https://www.fs.fed.us/grasslands/documents/primer/NG_Primer.pdf.

Owens, M.K., and Norton, B.E., 1992, Interactions of grazing and plant protection on basin big sagebrush (*Artemisia tridentata* spp. *tridentata*) seedling survival: Journal of Range Management, v. 45, no. 3, p. 257–262. [Also available at https://doi.org/10.2307/4002974.]

Paige, C., and Ritter, S.A., 1999, Birds in a sagebrush sea—Managing sagebrush for bird communities: Boise, Idaho, Partners in Flight Western Working Group, 47 p. [Also available at https://doi.org/10.5962/bhl.title.141890.]

Pardieck, K.L., Ziolkowski, D.J., Jr., Lutmerding, M., and Hudson, M.-A.R., 2018, North American Breeding Bird Survey dataset 1966–2017, version 2017.0: U.S. Geological Survey, Patuxent Wildlife Research Center, accessed July 2019 at https://doi.org/10.5066/F76972V8.

Parker, I.M., Mertens, S.K., and Schemske, D.W., 1993, Distribution of seven native and two exotic plants in a tallgrass prairie in southeastern Wisconsin—The importance of human disturbance: American Midland Naturalist, v. 130, no. 1, p. 43–55. [Also available at https://doi.org/10.2307/2426273.]

Passel, J.S., and Cohn, D., 2008, U.S. population projections—2005–2050: Washington, D.C., Pew Research Center, accessed July 2019 at https://www.pewsocialtrends.org/2008/02/11/us-population-projections-2005-2050/.

Patterson, M.P., 1994, Bird species abundance, composition, and vegetation characteristics, and bird productivity in Conservation Reserve Program land in central Iowa: Ames, Iowa, Iowa State University, Master's Thesis, 54 p.

Patterson, M.P., and Best, L.B., 1996, Bird abundance and nesting success in Iowa CRP fields—The importance of vegetation structure and composition: American Midland Naturalist, v. 135, no. 1, p. 153–167. [Also available at https://dx.doi.org/10.2307/2426881.]

Peden, D.G., Van Dyne, G.M., Rice, R.W., and Hansen, R.M., 1974, The trophic ecology of *Bison bison* L. on shortgrass plains: Journal of Applied Ecology, v. 11, no. 2, p. 489–497. [Also available at https://doi.org/10.2307/2402203.]

Perlut, N.G., Strong, A.M., Donovan, T.M., and Buckley, N.J., 2006, Grassland songbirds in a dynamic management landscape—Behavioral responses and management strategies: Ecological Applications, v. 16, no. 6, p. 2235–2247. [Also available at https://dx.doi.org/10.1890/1051-0761(2006)016%5B2235:GSIADM%5D2.0.CO;2.]

Perlut, N.G., Strong, A.M., Donovan, T.M., and Buckley, N.J., 2008a, Grassland songbird survival and recruitment in agricultural landscapes—Implications for source-sink demography: Ecology, v. 89, no. 7, p. 1941–1952. [Also available at https://dx.doi.org/10.1890/07-0900.1.]

Perlut, N.G., Strong, A.M., Donovan, T.M., and Buckley, N.J., 2008b, Regional population viability of grassland songbirds—Effects of agricultural management: Biological Conservation, v. 141, no. 12, p. 3139–3151. [Also available at https://dx.doi.org/10.1016/j.biocon.2008.09.011.]

Petersen, K.L., and Best, L.B., 1987, Effects of prescribed burning on nongame birds in a sagebrush community: Wildlife Society Bulletin, v. 15, no. 3, p. 317–329.

Pietz, P.J., and Granfors, D.A., 2000, Identifying predators and fates of grassland passerine nests using miniature video cameras: The Journal of Wildlife Management, v. 64, no. 1, p. 71–87. [Also available at https://doi.org/10.2307/3802976.]

Pietz, P.J., Buhl, D.A., Shaffer, J.A., Winter, M., and Johnson, D.H., 2009, Influence of trees in the landscape of parasitism rates of grassland passerine nests in southeastern North Dakota: The Condor, v. 111, no. 1, p. 36–42. [Also available at https://dx.doi.org/10.1525/cond.2009.080012.]

Pipher, E.N., Curry, C.M., and Koper, N., 2016, Cattle grazing intensity and duration have varied effects on songbird nest survival in mixed-grass prairies: Rangeland Ecology and Management, v. 69, no. 6, p. 437–443. [Also available at https://doi.org/10.1016/j.rama.2016.07.001.]

Pimental, D., 1993, Habitat factors in new pest invasions, *in* Kim, K.C., and McPheron, B.A., eds., Evolution of insect pests—Patterns of variation: New York, N.Y., John Wiley and Sons, p. 165–181.

Pimentel, D., Zuniga, R., and Morrison, D., 2005, Update on the environmental and economic costs associated with alien-invasive species in the United States: Ecological Economics, v. 52, no. 3, p. 273–288. [Also available at https://doi.org/10.1016/j.ecolecon.2004.10.002.]

Pitman, J.C., Hagen, C.A., Robel, R.J., Loughin, T.M., and Applegate, R.D., 2005, Location and success of Lesser Prairie-Chicken nests in relation to vegetation and human disturbance: Journal of Wildlife Management, v. 69, no. 3, p. 1259–1269. [Also available at https://doi.org/10.2193/0022-541X(2005)069[1259:LASOLP]2.0.CO;2.]

Plumb, G.E., and Dodd, J.L., 1993, Foraging ecology of bison and cattle on a northern mixed prairie—Implications for natural area management: Ecological Applications, v. 3, no. 4, p. 631–643. [Also available at https://doi.org/10.2307/1942096.]

Powell, A.F.L.A., 2006, Effects of prescribed burns and bison (*Bos bison*) grazing on breeding bird abundance in tallgrass prairie: The Auk, v. 123, no. 1, p. 183–197. [Also available at https://dx.doi.org/10.1642/0004-8038(2006)123%5B0183:EOPBAB%5D2.0.CO;2.]

Powell, A.F.L.A., 2008, Responses of breeding birds in tallgrass prairie to fire and cattle grazing: Journal of Field Ornithology, v. 79, no. 1, p. 41–52. [Also available at https://doi.org/10.1111/j.1557-9263.2008.00144.x.]

Powell, A.F.L.A., and Busby, W.H., 2013, Effects of grassland management on breeding birds at the western edge of the tallgrass prairie ecosystem in Kansas: Natural Areas Journal, v. 33, no. 2, p. 130–138. [Also available at https://doi.org/10.3375/043.033.0202.]

Prairie Conservation Action Plan, 2014, Saskatchewan Prairie Conservation Action Plan Framework 2014–2018: 29 p., accessed July 2019 at http://www.pcap-sk.org/rsu_docs/documents/PCAP_framework_2014.pdf.

Prescott, D.R.C., and Wagner, G.M., 1996, Avian responses to implementation of a complementary/rotational grazing system by the North American Waterfowl Management Plan in southern Alberta—The Medicine Wheel Project: Edmonton, Alberta, Alberta NAWMP Centre, NAWMP-018, 24 p.

Price, J.T., 1995, Potential impacts of global climate change on the summer distributions of some North American grassland birds: Detroit, Mich.,Wayne State University, Ph.D. Dissertation, 356 p.

Pyne, S.J., Andrews, P.L., and Laven, R.D., 1996, Introduction to wildland fire (2nd ed.): New York, N.Y., John Wiley & Sons, Inc., 808 p.

Rahmig, C.J., Jensen, W.E., and With, K.A., 2009, Grassland bird responses to land management in the largest remaining tallgrass prairie: Conservation Biology, v. 23, no. 2, p. 420–432. [Also available at https://dx.doi.org/10.1111/j.1523-1739.2008.01118.x.]

Ranellucci, C.L., Koper, N., and Henderson, D.C., 2012, Twice-over rotational grazing and its impacts on grassland songbird abundance and habitat structure: Rangeland Ecology and Management, v. 65, no. 2, p. 109–118. [Also available at https://dx.doi.org/10.2111/REM-D-11-00053.1.]

Rashford, B.S., Bastian, C.T., and Cole, J.G., 2011a, Agricultural land-use change in Prairie Canada—Implications for wetland and waterfowl habitat conservation: Canadian Journal of Agricultural Economics, v. 59, no. 2, p. 185–205. [Also available at https://doi.org/10.1111/j.1744-7976.2010.01212.x.]

Rashford, B.S., Walker, J.A., and Bastian, C.T., 2011b, Economics of grassland conversion to cropland in the Prairie Pothole Region: Conservation Biology, v. 25, no. 2, p. 276–284. [Also available at https://doi.org/10.1111/j.1523-1739.2010.01618.x.]

Reisner, M.D., Grace, J.D., Pyke, D.A., and Doescher, P.S., 2013, Conditions favoring *Bromus tectorum* dominance of endangered sagebrush steppe ecosystems: Journal of Applied Ecology, v. 50, no. 4, p. 1039–1049. [Also available at https://dx.doi.org/10.1111/1365-2664.12097.]

Renfrew, R.B., 2002, The influence of patch and landscape characteristics on grassland passerine density, nest success, and predators in southwestern Wisconsin pastures: Madison, Wisc., University of Wisconsin–Madison, Ph.D. Dissertation, 180 p.

Renfrew, R.B., and Ribic, C.A., 2001, Grassland birds associated with agricultural riparian practices in southwestern Wisconsin: Journal of Range Management, v. 54, no. 5, p. 546–552. [Also available at https://dx.doi.org/10.2307/4003583.]

Renfrew, R.B., and Ribic, C.A., 2002, Influence of topography on density of grassland passerines in pastures: American Midland Naturalist, v. 147, no. 2, p. 315–325. [Also available at https://dx.doi.org/10.1674/0003-0031(2002)147%5B0315:IOTODO%5D2.0.CO;2.]

Renfrew, R.B., and Ribic, C.A., 2003, Grassland passerine nest predators near pasture edges identified on videotape: The Auk, v. 120, no. 2, p. 371–383. [Also available at https://doi.org/10.1642/0004-8038(2003)120[0371:GPNPNP]2.0.CO;2.]

Renken, R.B., and Dinsmore, J.J., 1987, Nongame bird communities on managed grasslands in North Dakota: Canadian Field-Naturalist, v. 101, no. 4, p. 551–557.

Reynolds, R.E., Shaffer, T.L., Loesch, C.R., and Cox, R.R., Jr., 2006, The Farm Bill and duck production in the Prairie Pothole Region—Increasing the benefits: Wildlife Society Bulletin, v. 34, no. 4, Special Section: Farm Bill Contributions to Wildlife Conservation, p. 963–974.

Ribic, C.A., and Sample, D.W., 2001, Associations of grassland birds with landscape factors in southern Wisconsin: American Midland Naturalist, v. 146, no. 1, p. 105–121. [Also available at https://dx.doi.org/10.1674/0003-0031(2001)146%5B0105:AOGBWL%5D2.0.CO;2.]

Ribic, C.A., Koford, R.R., Herkert, J.R., Johnson, D.H., Niemuth, N.D., Naugle, D.E., Bakker, K.K., Sample, D.W., and Renfrew, R.B., 2009, Area sensitivity in North American grassland birds—Patterns and processes: The Auk, v. 126, no. 2, p. 233–244. [Also available at https://dx.doi.org/10.1525/auk.2009.1409.]

Richardson, A.N., Koper, N., and White, K.A., 2014, Interactions between ecological disturbances—Burning and grazing and their effects on songbird communities in northern mixed-grass prairies: Avian Conservation and Ecology, v. 9, no. 2, article 5. [Also available at https://dx.doi.org/10.5751/ACE-00692-090205.]

Ricketts, T.H., Dinerstein, E., Olson, D.M., Loucks, C.J., Eichbaum, W., DellaSala, D., Kavanagh, K., Hedao, P., Hurley, P.T., Carney, K.M., Abell, R., and Walters, S., 1999, Terrestrial Ecoregions of North America—A conservation assessment: Washington, D.C., Island Press, 508 p.

Ringelman, J., 2007, Biofuels and ducks: Ducks Unlimited, v. May/June, p. 50–55.

Risser, P.G., 1990, Landscape processes and the vegetation of the North American grassland, *in* Coolins, S.L., and Wallace, L.L., eds., Fire in North American tallgrass prairies: Norman, Okla., University of Oklahoma Press, p. 133–146.

Risser, P.G., Birney, E.C., Blocker, H.D., May, S.W., Parton, W.J., and Weins, J.A., 1981, The true prairie ecosystem: Stroudsburg, Pa., Hutchinson Ross Publishing Company, U.S. International Biological Program Synthesis Series, no. 16, 557 p.

Rodenhouse, N.L., Best, L.B., O'Connor, R.J., and Bollinger, E.K., 1993, Effects of temperate agriculture on Neotropical migrant landbirds, *in* Finch, D.M., and Stangel, P.W., eds., Status and management of Neotropical migratory birds: Fort Collins, Colo., U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-229, p. 280–295.

Rodenhouse, N.L., Best, L.B., O'Connor, R.J., and Bollinger, E.K., 1995, Effects of agricultural practices and farmland structures on Neotropical migratory birds, *in* Martin, T.E., and Finch, D.M., eds., Ecology and management of Neotropical migratory birds—A synthesis and review of critical issues: New York, N.Y., Oxford University Press, p. 269–293.

Roe, F.G., 1951, The North American buffalo—A critical study of the species in its wild state: Toronto, Canada, University of Toronto Press, 957 p.

Rohrbaugh, R.W., Jr., Reinking, D.L., Wolfe, D.H., Sherrod, S.K., and Jenkins, M.A., 1999, Effects of prescribed burning and grazing on nesting and reproductive success of three grassland passerine species in tallgrass prairie, *in* Vickery, P.D., and Herkert, J.R., eds., Ecology and conservation of grassland birds of the Western Hemisphere: Studies in Avian Biology, v. 19, p. 165–170.

Rosenberg, K.V., Kennedy, J.A., Dettmers, R., Ford, R.P., Reynolds, D., Alexander, J.D., Beardmore, C.J., Blancher, P.J., Bogart, R.E., Butcher, G.S., Camfield, A.F., Couturier, A., Demarest, D.W., Easton, W.E., Giocomo, J.J., Keller, R.H., Mini, A.E., Panjabi, A.O., Pashley, D.N., Rich, T.D., Ruth, J.M., Stabins, H., Stanton, J., and Will, T., 2016, Partners in Flight Landbird Conservation Plan—2016 Revision for Canada and Continental United States: Partners in Flight Science Committee, 119 p., accessed July 2019 at https://www.partnersinflight.org/resources/the-plan/.

Rotenberry, J.T., and Wiens, J.A., 1980, Habitat structure, patchiness, and avian communities in North American steppe vegetation—A multivariate analysis: Ecology, v. 61, no. 5, p. 1228–1250. [Also available at https://dx.doi.org/10.2307/1936840.]

Rowland, M.M., in press, The effects of management practices on grassland birds—Greater Sage-Grouse (*Centrocercus urophasianus*), chap. B *of* Johnson, D.H., Igl, L.D., Shaffer, J.A., and DeLong, J.P., eds., The effects of management practices on grassland birds: U.S. Geological Survey Professional Paper 1842.

Ryan, M.R., 1986, Nongame management in grassland and agricultural ecosystems, *in* Hale, J.B., Best, L.B., and Clawson, R.L., eds., Management of nongame wildlife in the Midwest—A developing art: Chelsea, Mich., BookCrafters, North Central Section of The Wildlife Society, p. 117–136.

Ryan, M.R., 1990, A dynamic approach to the conservation of the prairie ecosystem in the Midwest, *in* Sweeney, J.M., ed., Management of dynamic ecosystems: West Lafayette, Ind., The Wildlife Society, North Central Section, p. 91–106.

Ryan, M.R., Burger, L.W., and Kurzejeski, E.W., 1998, The impact of CRP on avian wildlife—A review: Journal of Production Agriculture, v. 11, no. 1, p. 61–66. [Also available at https://doi.org/10.2134/jpa1998.0061.]

Saab, V.A., Bock, C.E., Rich, T.D., and Dobkin, D.S., 1995, Livestock grazing effects in western North America, *in* Martin, T.E., and Finch, D.M., eds., Ecology and management of Neotropical migratory birds: New York, N.Y., Oxford University Press, p. 311–353.

Sample, D.W., 1989, Grassland birds in southern Wisconsin—Habitat preference, population trends, and response to land use changes: Madison, Wis., University of Wisconsin, Master's Thesis, 588 p.

Sample, D.W., and Mossman, M.J., 1997, Managing habitat for grassland birds—A guide for Wisconsin: Madison, Wisc., Wisconsin Department of Natural Resources, 154 p.

Samson, F., and Knopf, F., 1994, Prairie conservation in North America: Bioscience, v. 44, no. 6, p. 418–421. [Also available at https://doi.org/10.2307/1312365.]

Samson, F.B., Knopf, F.L., and Ostlie, W.R., 1998, Regional trends of biological resources—Grasslands, *in* Mac, M.J., Opler, P.A., Puckett Haecker, C.E., and Doran, P.D., eds., Status and trends of the nation's biological resources—Volume 2: Reston, Va., U.S. Geological Survey, p. 437–472.

Samson, F.B., Knopf, F.L., and Ostlie, W.R., 2004, Great Plains ecosystems—Past, present, and future: Wildlife Society Bulletin, v. 32, no. 1, p. 6–15. [Also available at https://doi.org/10.2193/0091-7648(2004)32[6:GPEPPA]2.0.CO;2.]

Sandoz, M., 1954, The buffalo hunters—The story of the hide men: New York, N.Y., Hastings House, 378 p.

Sauer, C.O., 1950, Grassland, climax, fire, and man: Journal of Range Management, v. 3, no. 1, p. 16–22. [Also available at https://doi.org/10.2307/3894702.]

Sauer, J.R., Link, W.A., Fallon, J.E., Pardieck, K.L., and Ziolkowski D.J., Jr., 2013, The North American breeding bird survey 1966–2011—Summary analysis and species accounts: North American Fauna, no. 79, p. 1–32. [Also available at https://doi.org/10.3996/nafa.79.0001.]

Saunders, S.C., Mislivets, M.R., Chen, J., and Cleland, D.T., 2002, Effects of roads on landscape structure within nested ecological units of the Northern Great Lakes Region, USA: Biological Conservation, v. 103, no. 2, p. 209–225. [Also available at https://doi.org/10.1016/S0006-3207(01)00130-6.]

Scheiman, D.M., Bollinger, E.K., and Johnson, D.H., 2003, Effects of leafy spurge infestation on grassland birds: The Journal of Wildlife Management, v. 67, no. 1, p. 115–121. [Also available at https://dx.doi.org/10.2307/3803067.]

Schmiegelow, F.K.A., and Monkkonen, M., 2002, Habitat loss and fragmentation in dynamic landscapes—Avian perspectives from the boreal forest: Ecological Applications, v. 12, no. 2, p. 375–389. [Also available at https://doi.org/10.2307/3060949.]

Schmutz, J.K., 1984, Ferruginous and Swainson's hawk abundance and distribution in relation to land use in southeastern Alberta: The Journal of Wildlife Management, v. 48, no. 4, p. 1180–1187. [Also available at https://doi.org/10.2307/3801779.]

Schneider, N.A., 1998, Passerine use of grasslands managed with two grazing regimes on the Missouri Coteau in North Dakota: Brookings, South Dakota, South Dakota State University, M.S. thesis, 94 p.

Schroeder, M.A., Young, J.R., and Braun, C.E., 1999, Greater Sage-Grouse (*Centrocercus urophasianus*) (ver. 2.0), *in* Poole, A.F., and Gill, F.B., eds., The birds of North America database: Ithaca, N.Y., Cornell Lab of Ornithology, accessed July 2019 at https://birdsna.org/Species-Account/bna/species/saggro. [Also available at https://doi.org/10.2173/bna.425.]

Schulte, L.A., MacDonald, A.L., Niemi, J.B., and Helmers, M.J., 2016, Prairie strips as a mechanism to promote land sharing by birds in industrial agricultural landscapes: Agriculture, Ecosystems and Environment, v. 220, p. 55–63. [Also available at https://doi.org/10.1016/j.agee.2016.01.007.]

Schulte, L.A., Niemi, J., Helmers, M.J., Liebman, M., Arbuckle, J.G., James, D.E., Kolka, R.K., O'Neal, M.E., Tomer, M.D., Tyndall, J.C., Asbjornsen, H., Drobney, P., Neal, J., Van Ryswyk, G., and Witte, C., 2017, Prairie strips improve biodiversity and the delivery of multiple ecosystem services from corn–soybean croplands: Proceedings of the National Academy of Sciences of the United States of America, v. 114, no. 42, p. 11247–11252. [Also available at https://doi.org/10.1073/PNAS.1620229114.]

Schwartz, C.C., and Ellis, J.E., 1981, Feeding ecology and niche separation in some native and domestic ungulates on the shortgrass prairie: Journal of Applied Ecology, v. 18, no. 2, p. 343–353. Also available at https://doi.org/10.2307/2402399.]

Scott, P.E., DeVault, T.L., Bajema, R.A., and Lima, S.L., 2002, Grassland vegetation and bird abundances on reclaimed midwestern coal mines: Wildlife Society Bulletin, v. 30, p. 1006–1014.

Sedivec, K.K., 1994, Grazing treatment effects on and habitat use of upland nesting birds on native rangeland: Fargo, North Dakota, North Dakota State University, Ph.D. dissertation, 124 p.

Service, R.F., 2007, A growing threat down on the farm: Science, v. 316, no. 5828, p. 1114–1117. [Also available at https://doi.org/10.1126/science.316.5828.1114.]

Shaffer, J.A., and Buhl, D.A., 2016, Effects of wind-energy facilities on breeding grassland bird distributions: Conservation Biology, v. 30, no. 1, p. 59–71. [Also available at https://dx.doi.org/10.1111/cobi.12569.]

Shaffer, J.A., Goldade, C.M., Dinkins, M.F., Johnson, D.H., Igl, L.D., and Euliss, B.R., 2003, Brown-headed Cowbirds in grasslands—Their habitats, hosts, and response to management: Prairie Naturalist, v. 35, no. 3, p. 146–186.

Shaffer, J.A., Igl, L.D., and Johnson, D.H., 2019a, The effects of management practices on grassland birds—Rates of Brown-headed Cowbird (*Molothrus ater*) parasitism in nests of North American grassland birds, chap. PP *of* Johnson, D.H., Igl, L.D., Shaffer, J.A., and DeLong, J.P., eds., The effects of management practices on grassland birds: U.S. Geological Survey Professional Paper 1842, 24 p., accessed July 2019 at https://doi.org/10.3133/pp1842PP.

Shaffer, J.A., Loesch, C.R., and Buhl, D.A., 2019b, Estimating offsets for avian displacement effects of anthropogenic impacts: Ecological Applications, v. 29, no. 8, e01983. [Also available at https://doi.org/10.1002/eap.1983.]

Shustack, D.P., Strong, A.M., and Donovan, T.M., 2010, Habitat use patterns of Bobolinks and Savannah Sparrows in the northeastern United States: Avian Conservation and Ecology, v. 5, no. 2, article 11. [Also available at https://dx.doi.org/10.5751/ACE-00423-050211.]

Sims, P.L., and Risser, P.G., 2000, Grasslands, *in* Barbour, M.G., and Billings, W.D., eds., North American terrestrial vegetation (2nd ed.): Cambridge, United Kingdom, Cambridge University Press, p. 323–356.

Skagen, S.K., Yackel Adams, A.A., and Adams, R.D., 2005, Nest survival relative to patch size in a highly fragmented shortgrass prairie landscape: The Wilson Bulletin, v. 117, no. 1, p. 23–34. [Also available at https://doi.org/10.1676/04-038.]

Sliwinski, M.S., and Koper, N., 2012, Grassland bird response to three edge types in a fragmented mixed-grass prairie: Avian Conservation and Ecology, v. 7, no. 2, article 6. [Also available at https://dx.doi.org/10.5751/ACE-00534-070206.]

Sliwinski, M.S., and Koper, N., 2015, Managing mixed-grass prairies for songbirds using variable cattle stocking rates: Rangeland Ecology and Management, v. 68, no. 6, p. 470–475. [Also available at https://dx.doi.org/10.1016/j.rama.2015.07.010.]

Smith, K.A., 2005, Prescribed burning suggestions, appendix A *of* Heitschmidt, R., Johnson, D., Kirby, D., Luttschwager, K., Peterson, H., Printz, J., Sedivec, K., and Smith, K., Report of the Scientific Review Team: Bismarck, N. Dak., U.S. Forest Service, Dakota Prairie Grasslands, p. 42–46.

Snyder, D.L., Riemenschneider, V., and Inman, V.W., 1987, The Upland Sandpiper, *Bartramia longicauda*, breeding area in South Bend, Indiana: Proceedings of the Indiana Academy of Sciences, v. 96, p. 537–542.

Society for Ecological Restoration International [SER], 2004, SER International primer on ecological restoration: Tucson, Ariz., Society for Ecological Restoration International, 13 p.

Spyreas, G., Wilm, B.W., Plocher, A.E., Ketzner, D.M., Matthews, J.W., Ellis, J.L., and Heske, E.J., 2010, Biological consequences of invasion by reed canary grass (*Phalaris arundinacea*): Biological Invasions, v. 12, no. 5, p. 1253–1267. [Also available at https://dx.doi.org/10.1007/s10530-009-9544-y.]

Steinauer, E.M., and Collins, S.L., 1996, Prairie ecology—The tallgrass prairie, *in* Samson, F.B., and Knopf, F.L., eds., Prairie conservation—Preserving North America’s most endangered ecosystem: Washington, D.C., Island Press, p. 39–52.

Stephens, S.E., Koons, D.N., Rotella, J.J., and Willey, D.W., 2004, Effects of habitat fragmentation on avian nesting success—A review of the evidence at multiple spatial scales: Biological Conservation, v. 115, no. 1, p. 101–110. [Also available at https://doi.org/10.1016/S0006-3207(03)00098-3.]

Stephens, S.E., Walker, J.A., Blunck, D.R., Jayaraman, A., Naugle, D.E., Ringelman, J.K., and Smith, A.J., 2008, Predicting risk of habitat conversion in native temperate grasslands: Conservation Biology, v. 22, no. 5, p. 1320–1330. [Also available at https://doi.org/10.1111/j.1523-1739.2008.01022.x.]

Stewart, R.E., 1975, Breeding birds of North Dakota: Fargo, N. Dak., Tri-College Center for Environmental Studies, 295 p.

Sugden, J.W., 1933, Range restriction of the Long-billed Curlew: The Condor, v. 35, no. 1, p. 3–9. [Also available at https://dx.doi.org/10.2307/1363458.]

Summers, C.A., and Linder, R.L., 1978, Food habits of the black-tailed prairie dog in western South Dakota: Journal of Range Management, v. 31, no. 2, p. 134–136. [Also available at https://doi.org/10.2307/3897663.]

Sutter, G.C., and Brigham, R.M., 1998, Avifaunal and habitat changes resulting from conversion of native prairie to crested wheatgrass—Patterns at songbird community and species levels: Canadian Journal of Zoology, v. 76, no. 5, p. 869–875. [Also available at https://doi.org/10.1139/z98-018.]

Sutter, G.C., Davis, S.K., and Duncan, D.C., 2000, Grassland songbird abundance along roads and trails in southern Saskatchewan: Journal of Field Ornithology, v. 71, no. 1, p. 110–116. [Also available at https://dx.doi.org/10.1648/0273-8570-71.1.110.]

Sutter, G.C., Davis, S.K., Skiffington, J.C., Keating, L.M., and Pittaway, L.A., 2016, Nesting behaviour and reproductive success of Sprague’s Pipit (*Anthus spragueii*) and Vesper Sparrow (*Pooecetes gramineus*) during pipeline construction: Canadian Field-Naturalist, v. 130, no. 2, p. 99–109. [Also available at https://dx.doi.org/10.22621/cfn.v130i2.1833.]

Tangen, B.A., and Finocchiaro, R.G., 2017, A case study examining the efficacy of drainage setbacks for limiting effects to wetlands in the Prairie Pothole Region, USA: Journal of Fish and Wildlife Management, v. 8, no. 2, p. 513–529. [Also available at https://doi.org/10.3996/022017-JFWM-012.]

The Nature Conservancy, 2019, The Nature Conservancy website: accessed July 2019 at https://www.nature.org/.

Thompson, F.R., III, Dijak, W., and Burhans, D.E., 1999, Video identification of predators at songbird nests in old fields: The Auk, v. 116, no. 1, p. 259–264. [Also available at https://doi.org/10.2307/4089477.]

Thompson, S.J., Johnson, D.H., Niemuth, N.D., and Ribic, C.A., 2015, Avoidance of unconventional oil wells and roads exacerbates habitat loss for grassland birds in the North American Great Plains: Biological Conservation, v. 192, p. 82–90. [Also available at https://dx.doi.org/10.1016/j.biocon.2015.08.040.]

Tinus, R.W., 1976, Shelterbelts on the Great Plains, Proceedings of the Symposium: Denver, Colo., Great Plains Agricultural Council Publication, no. 78, 218 p.

Trauger, D.L., Czech, B., Erickson, J.D., Garrettson, P.R., Kernohan, B.J., and Miller, C.A., 2003, The relationship of economic growth to wildlife conservation: Bethesda, Md., The Wildlife Society, Wildlife Society Technical Review 03-1, 22 p., accessed July 2019 at https://steadystate.org/wp-content/uploads/Trauger_et_al_Tech_Review_Economic_Growth.pdf.

Trimble, D.E., 1990, The geologic story of the Great Plains—Reprint with minor revisions of Geological Survey Bulletin 1493: Bismarck, N. Dak., Theodore Roosevelt Nature and History Association, 54 p.

Trombulak, S.C., and Frissell, C.A., 2000, Review of ecological effects of roads on terrestrial and aquatic communities: Conservation Biology, v. 14, no. 1, p. 18–30. [Also available at https://doi.org/10.1046/j.1523-1739.2000.99084.x.]

U.S. Department of Agriculture [USDA], 1999a, Grassland birds: Washington, D.C., U.S. Department of Agriculture, Natural Resources Conservation Service, Wildlife Habitat Management Institute, Fish and Wildlife Habitat Management Leaflet, no. 8, 12 p.

U.S. Department of Agriculture [USDA], 1999b, Ring-necked Pheasant (*Phasianus colchicus*): Washington, D.C., U.S. Department of Agriculture, Natural Resources Conservation Service, Wildlife Habitat Management Institute, Fish and Wildlife Habitat Management Leaflet, no. 10, 12 p.

U.S. Department of Agriculture [USDA], 2018, Natural Resources Conservation Service: accessed July 2019 at https://www.nrcs.usda.gov/programs.

U.S. Department of the Interior, 2019, National Park Service: accessed July 2019 at https://www.nps.gov/index.htm.

U.S. Fish and Wildlife Service [FWS], 2017, The small wetlands program—A half century of conserving prairie habitat: accessed July 2019 at https://www.fws.gov/refuges/whm/wetlands.html.

Vale, T.R., 1974, Sagebrush conversion projects—An element of contemporary environmental change in the western United States: Biological Conservation, v. 6, no. 4, p. 274–284. [Also available at https://dx.doi.org/10.1016/0006-3207(74)90006-8.]

Van Wilgenburg, S.L., Hobson, K.A., Bayne, E.M., and Koper, N., 2013, Estimated avian nest loss associated with oil and gas exploration and extraction in the western Canadian sedimentary basin: Avian Conservation and Biology, v. 8, no. 2, article 9. [Also available at https://dx.doi.org/10.5751/ACE-00585-080209.]

Vander Haegen, W.M., Dobler, F.C., and Pierce, D.J., 2000, Shrubsteppe bird response to habitat and landscape variables in eastern Washington, U.S.A: Conservation Biology, v. 14, no. 4, p. 1145–1160. [Also available at https://dx.doi.org/10.1046/j.1523-1739.2000.99293.x.]

Vander Haegen, W.M., Schroeder, M.A., and DeGraaf, R.M., 2002, Predation on real and artificial nests in shrubsteppe landscapes fragmented by agriculture: The Condor, v. 104, no. 3, p. 496–506. [Also available at https://dx.doi.org/10.1650/0010-5422(2002)104%5B0496:PORAAN%5D2.0.CO;2.]

Vickery, P.D., and Herkert, J.R., 2001, Recent advances in grassland bird research—Where do we go from here?: The Auk, v. 118, no. 1, p. 11–15. [Also available at https://doi.org/10.2307/4089754.]

Vickery, P.D., Herkert, J.R., Knopf, F.L., Ruth, J., and Keller, C.E., 2000, Grassland birds—An overview of threats and recommended management strategies, *in* Bonney, R.E., Jr., Pashley, D.N., and Cooper, R., eds., Strategies for bird conservation—Creating the Partners in Flight planning process: Fort Collins, Colo., U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, p. 74–77.

Vickery, P.D., Tubaro, P.L., Silva, J.M.C., Peterjohn, B.G., Herkert, J.R., and Cavalcanti, R.B., 1999, Conservation of grassland birds in the western hemisphere, *in* Vickery, P.D., and Herkert, J.R., eds., Ecology and conservation of grassland birds of the Western Hemisphere: Studies in Avian Biology, v. 19, p. 2–26.

Villard, M.-A., 2002, Habitat fragmentation—Major conservation issue or intellectual attractor?: Ecological Applications, v. 12, no. 2, p. 319–320. [Also available at https://doi.org/10.1890/1051-0761(2002)012[0319:HFMCIO]2.0.CO;2.]

Vitousek, P.M., Mooney, H.A., Lubchenco, J., and Melillo, J.M., 1997, Human domination of Earth's ecosystems: Science, v. 277, no. 5325, p. 494–499. [Also available at https://doi.org/10.1126/science.277.5325.494.]

Warner, R.E., and Etter, S.L., 1989, Hay cutting and the survival of pheasants—A long-term perspective: The Journal of Wildlife Management, v. 53, no. 2, p. 455–461. [Also available at https://doi.org/10.2307/3801150.]

Warrington, M.H., Curry, C.M., Antze, B., and Koper, N., 2018, Noise from four types of extractive energy infrastructure affects song features of Savannah Sparrows: The Condor, v. 120, no. 1, p. 1–15. [Also available at https://doi.org/10.1650/CONDOR-17-69.1.]

Watts, R.D., Compton, R.W., McCammon, J.H., Rich, C.L., Wright, S.M., Owens, T., and Ouren, D.S., 2007, Roadless space of the conterminous United States: Science, v. 316, no. 5825, p. 736–738. [Also available at https://doi.org/10.1126/science.1138141.]

Weaver, J.E., 1954, North American prairie: Lincoln, Nebr., Johnsen Publishing Company, 348 p.

Weaver, J.E., 1965, Native vegetation of Nebraska: Lincoln, Nebr., University of Nebraska Press, 185 p.

Weaver, J.E., 1968, Prairie plants and their environment—A fifty-year study in the Midwest: Lincoln, Nebr., University of Nebraska Press, 276 p.

Weaver, T., Payson, E.M., and Gustafson, D.L., 1996, Prairie ecology—The shortgrass prairie, *in* Samson, F.B., and Knopf, F.L., eds., Prairie conservation—Preserving North America's most endangered ecosystem: Washington, D.C., Island Press, p. 67–75.

Wellicome, T.I., and Haug, E.A., 1995, Second update of status report on the Burrowing Owl, *Speotyto cunicularia*, *in* Canada: Ottawa, Ontario, Committee on the Status of Endangered Wildlife in Canada.

Wellicome, T.I., Kardynal, K.J., Franken, R.J., and Gillies, C.S., 2014, Off-road sampling reveals a different grassland bird community than roadside sampling—Implications for survey design and estimates to guide conservation: Avian Conservation and Ecology, v. 9, no. 1, article 4. [Also available at https://dx.doi.org/10.5751/ACE-00624-090104.]

Werner, B., Tracy, J., Johnson, W.C., Voldseth, R.A., Guntenspergen, G.R., and Millett, B., 2016, Modeling the effects of tile drain placement on the hydrologic function of farmed prairie wetlands: Journal of the American Water Resources Association, v. 52, no. 6, p. 1482–1492. [Also available at https://dx.doi.org/10.1111/1752-1688.12471.]

West, N.E., 1999, Managing for biodiversity of rangelands, *in* Collins, W.W., and Qualset, C.O., eds., Biodiversity in agroecosystems: Boca Raton, Fla., CRC Press, p. 101–126.

Whalen, C.E., Bomberger Brown, M., McGee, J., Powell, L.A., and Walsh, E.J., 2018, Male Greater Prairie-Chickens adjust their vocalizations in the presence of wind turbine noise: The Condor, v. 120, no. 1, p. 137–148. [Also available at https://doi.org/10.1650/CONDOR-17-56.1.]

Whitmore, R.C., and Hall, G.A., 1978, The response of passerine species to a new resource—Reclaimed surface mines in West Virginia: American Birds, v. 32, no. 1, p. 6–9.

Whitt, M.B., Prince, H.H., and Cox, R.R., Jr., 1999, Avian use of purple loosestrife dominated habitat relative to other vegetation types in a Lake Huron wetland complex: The Wilson Bulletin, v. 111, no. 1, p. 105–114.

Wiens, J.A., 1969, An approach to the study of ecological relationships among grassland birds: Ornithological Monographs 8, p. 1–93. [Also available at https://dx.doi.org/10.2307/40166677.]

Wiens, J.A., 1970, Avian populations and patterns of habitat occupancy at the Pawnee site, 1968–1969: Fort Collins, Colo., Colorado State University, U.S. International Biological Program, Grassland Biome Technical Report 63, 57 p.

Wiens, J.A., 1974a, Climatic instability and the "ecological saturation" of bird communities in North American grasslands: The Condor, v. 76, no. 4, p. 385–400. [Also available at https://dx.doi.org/10.2307/1365813.]

Wiens, J.A., 1974b, Habitat heterogeneity and avian community structure in North American grasslands: American Midland Naturalist, v. 91, no. 1, p. 195–213. [Also available at https://doi.org/10.2307/2424522.]

Wilcove, D.S., Rothstein, D., Dubow, J., Phillips, A., and Losos, E., 1998, Quantifying threats to imperiled species in the United States: Bioscience, v. 48, no. 8, p. 607–615. [Also available at https://doi.org/10.2307/1313420.]

Wilcoxen, C.A., Walk, J.W., and Ward, M.P., 2018, Use of cover crop fields by migratory and resident birds: Agriculture, Ecosystems and Environment, v. 252, p. 42–50. [Also available at https://doi.org/10.1016/j.agee.2017.09.039.]

Willms, W.D., and Jefferson, P.G., 1993, Production characteristics of the mixed prairie—Constraints and potential: Canadian Journal of Animal Science, v. 73, no. 4, p. 765–778. [Also available at https://doi.org/10.4141/cjas93-080.]

Wilson, S.D., and Belcher, J.W., 1989, Plant and bird communities of native prairie and introduced Eurasian vegetation in Manitoba, Canada: Conservation Biology, v. 3, no. 1, p. 39–44. [Also available at https://dx.doi.org/10.1111/j.1523-1739.1989.tb00222.x.]

Winder, V.L., Gregory, A.J., McNew, L.B., and Sandercock, B.K., 2015, Responses of male Greater Prairie-Chickens to wind energy development: The Condor, v. 117, no. 2, p. 284–296. [Also available at https://doi.org/10.1650/CONDOR-14-98.1.]

Winder, V.L., McNew, L.B., Gregory, A.J., Hunt, L.M., Wisely, S.M., and Sandercock, B.K., 2014a, Effects of wind energy development on survival of female Greater Prairie Chickens: Journal of Applied Ecology, v. 51, no. 2, p. 395–405. [Also available at https://doi.org/10.1111/1365-2664.12184.]

Winder, V.L., McNew, L.B., Gregory, A.J., Hunt, L.M., Wisely, S.M., and Sandercock, B.K., 2014b, Space use by female Greater Prairie-Chickens in response to wind energy development: Ecosphere, v. 5, no. 1, article 3. [Also available at https://dx.doi.org/10.1890/ES13-00206.1.]

Winter, M., and Faaborg, J., 1999, Patterns of area sensitivity in grassland-nesting birds: Conservation Biology, v. 13, no. 6, p. 1424–1436. [Also available at https://dx.doi.org/10.1046/j.1523-1739.1999.98430.x.]

Winter, M., Johnson, D.H., and Faaborg, J., 2000, Evidence for edge effects on multiple levels in tallgrass prairie: Condor, v. 102, no. 2, p. 256–266. [Also available at https://doi.org/10.1650/0010-5422(2000)102[0256:EFEEOM]2.0.CO;2.]

Winter, M., Johnson, D.H., and Shaffer, J.A., 2005a, Variability in vegetation effects on density and nesting success of grassland birds: The Journal of Wildlife Management, v. 69, no. 1, p. 185–197. [Also available at https://dx.doi.org/10.2193/0022-541X(2005)069%3C0185:VIVEOD%3E2.0.CO;2.]

Winter, M., Johnson, D.H., Shaffer, J.A., and Svedarsky, W.D., 2004, Nesting biology of three grassland passerines in northern tallgrass prairie: The Wilson Bulletin, v. 116, no. 3, p. 211–223. [Also available at https://dx.doi.org/10.1676/03-082.]

Winter, M., Johnson, D.H., Shaffer, J.A., Donovan, T.M., and Svedarsky, W.D., 2006, Patch size and landscape effects on density and nesting success of grassland birds: The Journal of Wildlife Management, v. 70, no. 1, p. 158–172. [Also available at https://dx.doi.org/10.2193/0022-541X(2006)70%5B158:PSALEO%5D2.0.CO;2.]

Winter, M., Shaffer, J.A., Johnson, D.H., Donovan, T.M., Svedarsky, W.D., Jones, P.W., and Euliss, B.R., 2005b, Habitat and nesting of Le Conte's Sparrows in the northern tallgrass prairie: Journal of Field Ornithology, v. 76, no. 1, p. 61–71. [Also available at https://dx.doi.org/10.1648/0273-8570-76.1.61.]

With, K.A., King, A.W., and Jensen, W.E., 2008, Remaining large grasslands may not be sufficient to prevent grassland bird declines: Biological Conservation, v. 141, no. 12, p. 3152–3167. [Also available at https://dx.doi.org/10.1016/j.biocon.2008.09.025.]

Wright, C.K., and Wimberly, M.C., 2013, Recent land use change in the Western Corn Belt threatens grasslands and wetlands: Proceedings of the National Academy of Sciences of the United States of America, v. 110, no. 10, p. 4134–4139. [Also available at https://doi.org/10.1073/pnas.1215404110.]

Wrobleski, D.W., and Kauffman, J.B., 2003, Initial effects of prescribed fire on morphology, abundance, and phenology of forbs in big sagebrush communities of southeastern Oregon: Restoration Ecology, v. 11, no. 1, p. 82–90. [Also available at https://dx.doi.org/10.1046/j.1526-100X.2003.00084.x.]

Yoo, J., and Koper, N., 2017, Effects of shallow natural gas well structures and associated roads on grassland songbird reproductive success in Alberta, Canada: PLoS One, v. 12, no. 3, p. e0174243. [Also available at https://doi.org/10.1371/journal.pone.0174243.]

Young, C.E., and Osborn, C.T., 1990, Costs and benefits of the Conservation Reserve Program: Journal of Soil and Water Conservation, v. 45, no. 3, p. 370–373.

Zalik, N.J., and Strong, A.M., 2008, Effects of hay cropping on invertebrate biomass and the breeding ecology of Savannah Sparrows (*Passerculus sandwichensis*): The Auk, v. 125, no. 3, p. 700–710. [Also available at https://dx.doi.org/10.1525/auk.2008.07106.]

Zimmerman, J.L., 1992, Density-independent factors affecting the avian diversity of the tallgrass prairie community: The Wilson Bulletin, v. 104, no. 1, p. 85–94.

Zimmerman, J.L., 1997, Avian community responses to fire, grazing, and drought in the tallgrass prairie, *in* Knopf, F.L., and Samson, F.B., eds., Ecology and conservation of Great Plains vertebrates: New York, N.Y., Springer-Verlag, p. 167–180. [Also available at https://doi.org/10.1007/978-1-4757-2703-6_7.]

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