

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. Shaffer¹ and John P. DeLong^{1,2}

Chapter A of

The Effects of Management Practices on Grassland Birds

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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

$$^{\circ}\text{F} = (1.8 \times ^{\circ}\text{C}) + 32.$$

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

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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 (km²) (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;

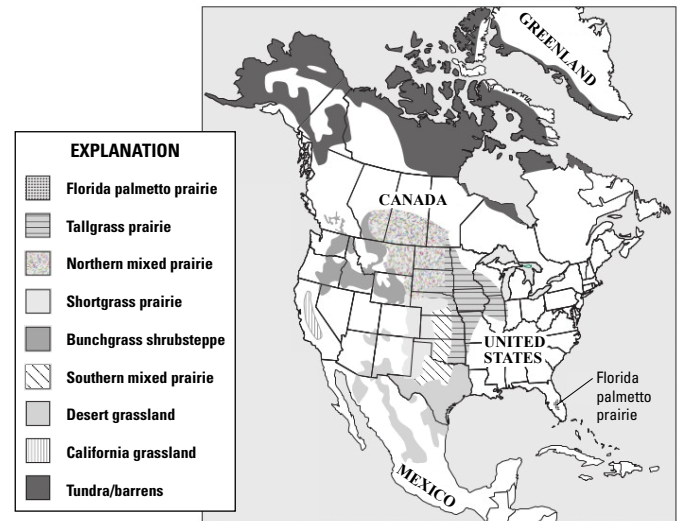


Figure A1. Distribution of major grassland ecosystems in North America prior to European settlement. Modified from Vickery and others (1999) and used with permission.

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.

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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 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.

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).

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 Grasslands, 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



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.

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

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 (Fuhlen-dorf 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 desiccating 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 McCown's 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 herbaceous and grassland 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 km² 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 platensis*), 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 (Duebber 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 (Duebber 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. Wilcoxon 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 cucularia 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 McCown's 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; Fuhlerdorf 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 long-standing 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 hard-scaping. 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 Iffland, 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 McCown's 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 tall-grass 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 (Wroblewski 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.

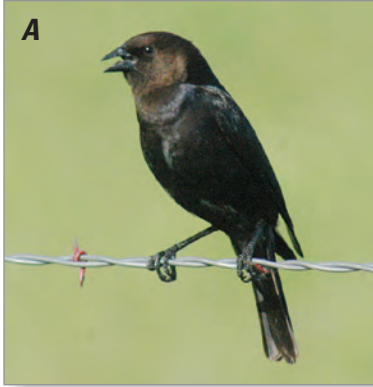
Providing a mosaic of idle and managed grasslands will ensure that some residual vegetation is available for those species that require it, especially if adjacent patches had been burned, mowed, or hayed, or received other management treatments (Sample and Mossman, 1997).

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, 2019). 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, 2019), 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 (in press) calculated average avoidance rates ranging from 18 percent for the first-year post-construction 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 km² during the pre-construction phase to 97 km² 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.

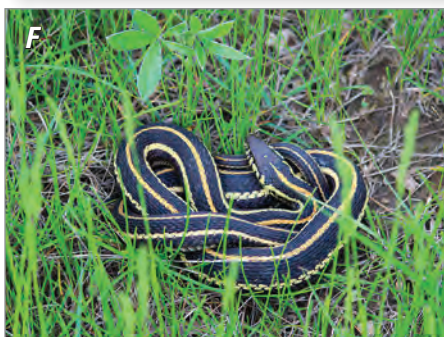
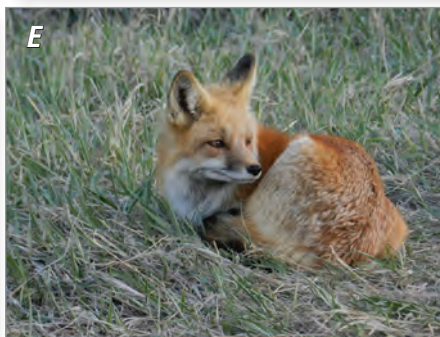
Conservation planning and acquisition efforts should consider the landscape context in which grassland fragments under consideration are embedded (Niemuth and others, 2008). Because patch size might be less relevant to grassland passerines when fragments are located in treeless landscapes, the size requirements 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.

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The Effects of Management Practices on Grassland Birds—Long-Billed Curlew (*Numenius americanus*)

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Chapter G of

The Effects of Management Practices on Grassland Birds

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Figure

- G1. Map showing breeding distribution of the Long-billed Curlew (*Numenius americanus*) in the United States and southern Canada, based on North American Breeding Bird Survey data, 2008–12.....2

Table

- G1. Measured values of vegetation structure and composition in Long-billed Curlew (*Numenius americanus*) breeding habitat by study.....12

Conversion Factors

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)

Abbreviations

DDE	dichlorodiphenyldichloroethylene
n.d.	no date
ppm	parts per million
spp.	species (applies to two or more species within the genus)
VOR	visual obstruction reading

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The Effects of Management Practices on Grassland Birds—Long-Billed Curlew (*Numenius americanus*)

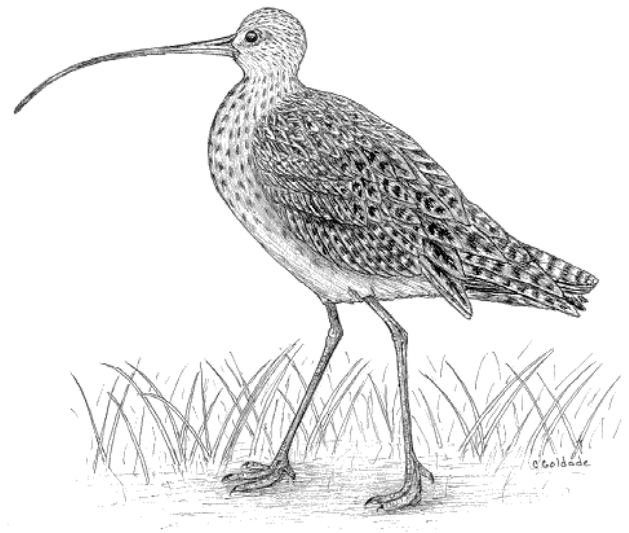
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Capsule Statement

Keys to Long-billed Curlew (*Numenius americanus*) management include providing large, open, level to gently rolling grasslands with short vegetation, and tailoring grazing regimes to local conditions. Long-billed Curlews have been reported to use habitats with 3–75 centimeters (cm) average vegetation height, less than or equal to 27 cm visual obstruction reading (VOR), 20–71 percent grass cover, 4–50 percent forb cover, 2–12 percent shrub cover, 7–40 percent bare ground, and less than (<) 3 cm litter depth. Descriptions of key vegetation characteristics are provided in table G1 (after the “References” section). Vernacular and scientific names of plants and animals follow the Integrated Taxonomic Information System (<https://www.itis.gov>).

Breeding Range

Long-billed Curlews breed from interior British Columbia and southern Alberta through southern Manitoba; south to central California; and east to southwestern North Dakota, central South Dakota, central Nebraska, western Kansas, northeastern New Mexico, and northern Texas (National Geographic Society, 2011). The relative densities of Long-billed Curlews in the United States and southern Canada, based on North American Breeding Bird Survey data (Sauer and others, 2014), are shown in figure G1 (not all geographic places mentioned in report are shown on figure).



Long-billed Curlew. Illustration by Christopher M. Goldade, U.S. Geological Survey.

Suitable Habitat

Long-billed Curlews use expansive, open, level to gently sloping or rolling grasslands with short vegetation such as shortgrass prairies or recently grazed mixed-grass prairies (Salt and Wilk, 1958; Bent, 1962; Graul, 1971; Stewart, 1975; Johnsgard, 1980; Bicak and others, 1982; Cochran and Anderson, 1987; Shackford, 1987; Eldridge, 1992; Clarke, 2006). They commonly nest in wet and dry prairies and in rangeland and occasionally nest in hayland, fallow fields, or stubble fields (Salt and Wilk, 1958; Bent, 1962; McCallum and others, 1977; Renaud, 1980; Cochran and Anderson, 1987; Shackford, 1994; Dugger and Dugger, 2002; Ackerman, 2007). Long-billed Curlews have been found nesting in cultivated land, such as fall-seeded winter wheat (*Triticum* species [spp.]) or spring-seeded barley (*Hordeum* spp.) (Devries and others, 2010), and will use cropland if native grasslands are not available (Saunders, 2001). Smith and Lomolino (2004) found a preference for shortgrass prairies with black-tailed prairie dog

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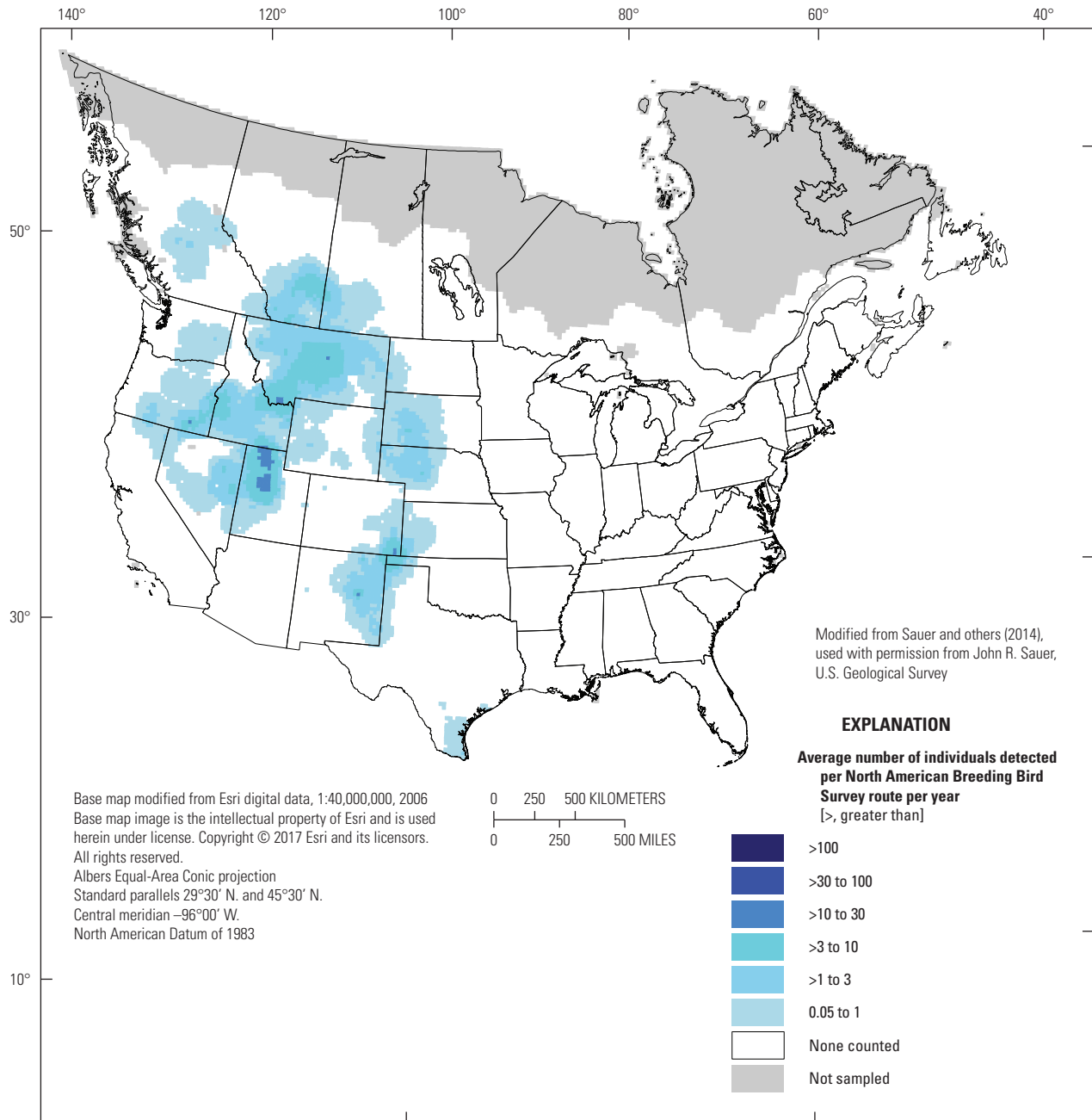


Figure G1. Breeding distribution of the Long-billed Curlew (*Numenius americanus*) in the United States and southern Canada, based on North American Breeding Bird Survey (BBS) data, 2008–12. The BBS abundance map provides only an approximation of breeding range edges.

(*Cynomys ludovicianus*) towns over shortgrass prairies without prairie dogs, fallow crop fields, scrub-sandsage, and Conservation Reserve Program grasslands. Long-billed Curlews occasionally inhabit grasslands enrolled in the Permanent Cover Program in Canada (McMaster and Davis, 1998).

Long-billed Curlews use a variety of vegetation types and prefer somewhat short vegetation. From a range-wide survey of Long-billed Curlews in the United States, Saalfeld and others (2010) determined that the species preferred short-grass prairies and rangeland, with vegetation height ranging

from 4 to 15 cm, within 400 meters (m) of survey stops along roads (excluding interstate highways and roads with two or more lanes). Numbers of curlews were negatively associated with shrub or scrub habitats. Within 800 m of stops, Long-billed Curlews were positively associated with rangeland and hayland and negatively associated with evergreen forests. In Alberta, the strongest predictor of curlew numbers was the percentage of each sampling unit that was native grassland, whereas curlew numbers were negatively associated with the percentage of each sampling unit that was cultivated land

or riparian area (Saunders, 2001). In Nebraska, the species used areas in which 75 percent of the total vertical vegetation density (number of plant contacts with a thin rod inserted vertically into the canopy) was at heights <10 cm, compared to 63 percent in unused areas (Bicak, 1977). Preference for areas in which vegetation density is concentrated near ground level may be important in terms of the feeding behavior of Long-billed Curlews or their ability to see potential predators. In Colorado, the species used shortgrass and mixed-grass prairies and weedy areas more than expected based on the availability of those habitats, and they used agricultural areas (for example, cropland, stubble fields, and bare ground) less than expected; they did not use areas dominated by sand sagebrush (King, 1978). In north-central Oregon, areas of shrubs or areas of downy brome (*Bromus tectorum*) intermixed with patches of Sandberg's bluegrass (*Poa secunda*) were preferred or used in proportion to availability (Pampush, 1980; Pampush and Anthony, 1993). Areas of dense forbs, antelope bitterbrush (*Purshia tridentata*), and bunchgrasses were used in proportion to their availability or were avoided. Bunchgrass habitats used by adults for brood rearing were contiguous with downy brome areas used as nesting sites.

Long-billed Curlews forage in grasslands, cultivated fields, stubble fields, wet meadows, prairie dog colonies, and occasionally along wetland margins (Silloway, 1900; Salt and Wilk, 1958; Johnsgard, 1980; Shackford, 1987; Prescott, 1997). During the incubation period in southwestern Idaho, Long-billed Curlew prey-capture rates were higher in areas with short grass even though prey density was higher in areas with tall grass (vegetation measurements, prey densities, and prey capture rates were not given) (Bicak and others, 1982; Bicak, 1983). Prelaying female curlews in western Idaho foraged in shortgrass pastures within their territories during years when vegetation was short (3.6–9.7 cm tall) (Redmond, 1986). However, during a year when vegetation was dense and tall (12–15.7 cm tall, with areas as high as 40 cm tall) owing to abundant precipitation, curlews flew as far as 10 kilometers (km) from their territories to forage. In south-central Washington, Long-billed Curlews preferred to forage in areas with higher topographic diversity (ridges and small dunes) and higher plant species diversity than in flatter areas with more homogeneous vegetation (Allen, 1980).

Nests often are located near cow dung or other conspicuous objects, possibly for concealment (Silloway, 1900; Bent, 1962; King, 1978; Johnsgard, 1979; Allen, 1980; Cochran and Anderson, 1987; Clarke, 2006). Additionally, nests often are placed on hummocks greater than or equal to 2.5 cm above the immediate surroundings, possibly to improve visibility of predators and to prevent flooding in otherwise level fields (Cochran and Anderson, 1987).

In North Dakota mixed-grass and shortgrass prairies, Long-billed Curlews prefer gently rolling terrain with gravelly soils (Stewart, 1975). In central Montana, Long-billed Curlews nested on dry portions of mixed-grass prairies, which were elevated above their surroundings and located near wet meadows (Silloway, 1900). Long-billed Curlews in Nebraska

nest on upland slopes of native vegetation near moist meadows that are used for foraging (Johnsgard, 1980). Grassy flood plains adjoining a creek provided nesting habitat in southeastern Colorado (Davis, 1949). In the Oklahoma Panhandle, Long-billed Curlews usually were observed in areas with clay loam soils on <1 percent slopes (Shackford, 1987). In northern Utah, Long-billed Curlew nests were found in irrigated and nonirrigated grass pastures and on alkali flats (Sugden, 1933; Forsythe, 1972; Paton and Dalton, 1994). Nests in that area were built in bunchgrasses, clumps of sedges (*Carex* spp.), stands of inland saltgrass (*Distichlis spicata*), or red saltwort (*Salicornia rubra*) (Forsythe, 1972). In south-central Washington, Long-billed Curlew breeding density was higher in topographically diverse areas, although most nests were placed on somewhat flat ground (neither the proportion of nests nor the slope of the ground was given) (Allen, 1980). Of 59 nests, 37 percent were 30–100 cm from an object (for example, big sagebrush (*Artemisia tridentata*) branches, rocks, dirt mounds, horse manure, metal cans, bunchgrasses), 31 percent were less than or equal to 30 cm from an object, 27 percent were immediately adjacent to an object, and only 5 percent were greater than 100 cm from an object (Allen, 1980). Big sagebrush, antelope bitterbrush, trees, dried tumbleweeds (*Salsola* spp.), dirt mounds, rocks, tree stumps, and fences were used as perches.

Vegetation composition and structure play an important role in nest selection. In 1 of 2 years in South Dakota grazed mixed-grass prairies, Clarke (2006) found no difference among nest sites, brood points, and random points for VOR; coverage of grasses, forbs, and bare ground; and distance to water. Long-billed Curlews selected nest sites similar to random points, with an average of 55 percent grass cover, 47 percent forb cover, and an average VOR of 27 cm. The plant species around nest sites ranged from an average height of 10 to 45 cm. In the second year, nest sites were in shorter vegetation with lower VORs than random points (Clarke, 2006). Nest sites had lower shrub coverage than random points, and slope was steeper at random points than at nest sites. In the second year, nest sites had lower VOR, more bare ground, and less forb coverage than in the previous year, whereas brood points also had lower VOR and grass and forb coverage, but more bare ground. Daily nest survival rates were positively related to average VORs taken at nest sites. Daily nest survival rate was higher at nest sites dominated by forb cover than at nest sites dominated by grass cover. Distance to the nearest manure pile was shorter from nest sites than from random points.

At Crescent Lake National Wildlife Refuge in Nebraska, Gregory and others (2011) found strong evidence for a negative effect of large-scale VOR (mean of 16 samples within 2–25 m from nests) on nest survival, and a weak but negative effect of small-scale VOR (mean of 4 samples within 2 m of nest) and forb cover on nest survival. Bare ground had a weak but positive influence on nest survival. Grass cover, litter cover, vegetation depth, and height of tallest vegetation had no influence. In Wyoming, nest sites within hayfields and pastures were characterized by less bare ground and higher percentage

cover of grasses (values were not given) than random sites (Cochran and Anderson, 1987). Hayfields and pastures with nests had lower percentage of grass cover (mean of 20 compared to 32 percent), greater forb cover (mean of 16 compared to 4 percent), and were drier (45 compared to 3 percent of random locations characterized as “dry”) than hayfields and pastures without nests. In Colorado and Texas, mean vegetation height was 11 cm centered at seven nests and 20.6 cm at 3 m from nests (King, 1978).

In Utah, 10 habitat patches containing nests had shorter vegetation (mean of 5.6 cm) than random habitat patches (mean of 9.0 cm) and had more bare ground 6–15 m from the nest (mean of 34–36 percent) than random patches (mean of 38–39 percent) (Paton and Dalton, 1994). At nest sites, however, vegetation <3 m from the nest was taller (mean of 6.5 cm) than vegetation 6–15 m from the nest (mean of 4.9–5.5 cm). Percentage of bare ground <3 m from the nest was lower (mean of 18 percent) than greater than or equal to 6 m from the nest (mean of 28–39 percent). In north-central Oregon, several vegetation variables differed between nesting areas and non-nesting areas (Pampush, 1980; Pampush and Anthony, 1993). Compared to non-nesting areas, nesting areas had shorter vegetation (24 compared to 29 cm at non-nesting areas), grass with less variation in height, total vegetation with less variation in height, grass with higher vertical density (0.8 compared to 0.2 contacts per 5-cm height increment) in the 25–50-cm height increment, and shrubs with lower total vertical density (0.02 compared to 0.05 contacts per 5-cm height increment). Nest density within study areas was negatively correlated with vegetation height and vertical density; nest density was positively correlated with percentage cover of bare ground and with the evenness of forb height. Depredation of eggs and chicks was high in habitats other than downy brome, possibly indicating that predator densities were higher or nests were more vulnerable in those habitats.

Exotic or invasive vegetation may reduce or improve habitat quality, depending on plant species and region (Dugger and Dugger, 2002). In South Dakota, nest sites dominated by yellow sweet clover (*Melilotus officinalis*), junegrass (*Koeleria macrantha*), green needlegrass (*Nassella viridula*), and buffalograss (*Bouteloua dactyloides*) had 100-percent daily nest survival rate, whereas survival rates were lower at nest sites dominated by Japanese brome (*Bromus japonicus*) and western wheatgrass (*Pascopyrum smithii*) (Clarke, 2006). American vetch (*Vicia americana*), junegrass, and buffalograss represented a higher proportion of species composition at nest sites than at random points. In southeastern Washington, 71 percent of 21 nests were in areas dominated by a mixture of downy brome and Sandberg’s bluegrass and 29 percent were in areas dominated by downy brome alone (Allen, 1980). Nearly all areas containing downy brome and Sandberg’s bluegrass were used for nesting, whereas areas containing solely downy brome were not always used for nesting. Allen (1980) attributed preference for areas dominated by the two plant species to a lower percentage cover of live (7 percent) and dead (65 percent) downy brome in those areas than in areas

dominated by downy brome alone (live: 14 percent; dead: 92 percent). Plant communities dominated by downy brome but containing substantial amounts of tumbleweed mustard (*Sisymbrium altissimum*), as well as other grass communities (for example, wheatgrass communities), were not used for nesting. In north-central Oregon, mean nest density was highest in downy brome and Sandberg’s bluegrass, followed by bunchgrasses, dense forbs and shrubs, and antelope bitterbrush (Pampush, 1980; Pampush and Anthony, 1993). In Colorado and Texas, six of seven nests were in areas dominated by buffalograss and blue grama (*Bouteloua gracilis*), and one nest was in an area dominated by sand dropseed (*Sporobolus cryptandrus*) (King, 1978).

Adults and broods move to habitats surrounding nest sites for cover, shade, and food (Maher, 1973; King, 1978; Allen, 1980; Pampush, 1980; Pampush and Anthony, 1993). After eggs hatch, adults and broods continue to forage in shortgrass and mixed-grass habitats, but they increase their use of areas that have more vegetative cover (for example, cropland, stubble fields, and weedy areas) (Maher, 1973, 1974; King, 1978; Allen, 1980; Pampush, 1980; Pampush and Anthony, 1993), particularly if vegetation is sparse at the nest site (Maher, 1974). In South Dakota, brood habitat contained a greater proportion of sixweeks fescue (*Vulpia octoflora*), Indianwheat (*Plantago* spp.), junegrass, and American vetch than random points in 1 year and a greater proportion of creeping spikerush (*Eleocharis palustris*) and water than random points in another year (Clarke, 2006). Use of areas with tall, dense vegetation in the Texas Panhandle and north-central Oregon may have provided chicks with an important source of shade or concealment cover (King, 1978; Pampush, 1980; Pampush and Anthony, 1993). In central South Dakota, Long-billed Curlews with chicks were reported in grass that was 18 cm tall (Spomer, 1981). In Oklahoma, Long-billed Curlews with young were observed in cultivated fields, shortgrass prairie, and tame grassland (Shackford, 1994).

Proximity to water may be an important factor in habitat selection (Bent, 1962; McCallum and others, 1977; Cochran and Anderson, 1987; Shackford, 1987). From a range-wide survey of Long-billed Curlews in the United States, Saalfeld and others (2010) determined that numbers of Long-billed Curlews were positively associated with wetland habitats within 400 m of survey stops along roadsides. In South Dakota, both nest sites and brood locations were within 500 m of the nearest water source (Clarke, 2006). During a dry year, broods used habitats consisting of a greater proportion of water than random points, and broods were located almost 200 m closer to water than in a year of average precipitation. In the Platte River Valley of Nebraska, Long-billed Curlews nested at higher densities in wet meadows than in upland prairies (Faanes and Lingle, 1995). Within the Nebraska sandhills at Crescent Lake National Wildlife Refuge and surrounding grasslands, proximity of mixed-grass uplands to wet meadows was the most important criterion in nest-site selection (Bicak, 1977). Wet meadows were used for feeding, loafing, and fledging young and were aggressively defended. In southeastern

Colorado, 41 percent of 63 Long-billed Curlew observations were within 91 m of standing water, and 68 percent of observations were within 403 m of water (McCallum and others, 1977). In southeastern Colorado and northwestern Texas, 39 percent of 354 curlew observations occurred within 400 m of stock ponds or irrigation facilities (King, 1978). Shackford (1987) suggested that a drop in the water table in the panhandle of Oklahoma caused Long-billed Curlews to favor areas near irrigated fields over upland, shortgrass sites. In Utah, nests often were placed near the edges of alkali flats of the Great Salt Lake (Paton and Dalton, 1994). In southeastern Alberta, Long-billed Curlews were less common on wet transects (defined as having wetlands intersecting transects along greater than 5 percent of their length) than on dry transects (Gratto-Trevor, 1999). Because curlews are known to return to the same area to nest each year, regardless of whether water is still available, curlews may be found nesting far from water if water sources have disappeared between breeding seasons (McCallum and others, 1977). In Alberta, Sliwinski and Koper (2012) found that Long-billed Curlew abundance decreased by 25 percent within 0.31 km of wetland edges, possibly because vegetation density was highest near wetlands.

Seasonal moisture levels may affect the abundance, distribution, and brood-survival rates of Long-billed Curlews. In an assessment of North American Breeding Bird Survey data for the conterminous United States, O'Connor and others (1999) reported a negative relationship between Long-billed Curlew abundance and the mean annual precipitation and the 30-year average of January temperature. Hartman (2008) evaluated the influence of precipitation on brood and chick daily survival rate in northeastern Nevada. The cumulative precipitation from October of the previous year through May of the current year was included as a variable in analyses because it corresponded to the period of greatest precipitation and time during which snowpack used to irrigate Nevada hayfields in the current year accumulated in the nearby mountains. Brood survival rate was slightly greater in years with higher winter precipitation, and individual chick daily survival rate was greater in wet years.

Area Requirements and Landscape Associations

Territory size is highly variable across the species' breeding range (De Smet, 1992). In southwestern Idaho, curlew densities were positively correlated with size of the management unit and with amount of area within the management unit that contained vegetation <10 cm tall (Bicak and others, 1982). Territory size averaged about 14 hectares (ha) in the most densely populated areas, and there was an unoccupied buffer zone of 300–500 m around the edge of suitable habitat (Redmond and others, 1981). In southeastern Washington, areas with diverse topography and habitat (shrubby areas near the nest sites) supported smaller curlew territories (6–8 ha)

than did open, flat, less diverse habitat, which supported larger territories (20 ha) (Allen, 1980). An increase in the breeding population between years did not result in the reduction of territory size, but rather resulted in an increased use of marginal habitat. Allen (1980) indicated that the existing territories may have already reached a minimum size. In South Dakota, the average 95-percent home range for five territories in 1 year ranged from 70 to 490 ha and ranged from 52 to 100 ha for the brood-rearing period; in the second year, the 95-percent home range for 13 territories ranged from 115 to 2,910 ha (Clarke, 2006).

After eggs hatch, adults and their broods often leave the nesting area. In southern Saskatchewan, one pair of adults with a brood was recorded more than 6.5 km from the nest site 6 days after hatching (Maher, 1974; Sadler and Maher, 1976).

Little information exists about the effect of habitat fragmentation on Long-billed Curlews. Although Long-billed Curlews prefer large expanses of grasslands, the effects of fragmentation on curlews has been poorly studied but is considered a potential threat to breeding populations (Sedgwick, 2006). In Alberta, Sliwinski and Koper (2012) found no effect of road or cropland edges on curlew abundance. In British Columbia, Ohanjanian (1992) determined that breeding Long-billed Curlews used only grassland areas that were greater than 250 m.

Brood Parasitism by Cowbirds and Other Species

No known records of brood parasitism by Brown-headed Cowbirds (*Molothrus ater*) exist (Shaffer and others, 2019). Long-billed Curlews and Willets (*Tringa semipalmata*) will occasionally parasitize each other's nests (Sugden, 1933; Bent, 1962; Dugger and Dugger, 2002).

Breeding-Season Phenology and Site Fidelity

Long-billed Curlews arrive on the breeding grounds from about mid-March through May and depart for the wintering grounds from August to October (Silloway, 1900; Sugden, 1933; Salt and Wilk, 1958; Bent, 1962; Maher, 1974; Stewart, 1975; Allen, 1980; Pampush, 1980; Renaud, 1980; Redmond and others, 1981; Bicak and others, 1982; Paton and Dalton, 1994; Saunders, 2001; Clarke, 2006). In some areas, fall departure may begin as early as June or July (Maher, 1973; King, 1978; Allen, 1980; Clarke, 2006; Page and others, 2014), especially by unsuccessful breeders (Allen, 1980; Paton and Dalton, 1994). Peak breeding season in North Dakota is early May through early June (Stewart, 1975). A single re-nesting attempt following depredation of a first clutch was observed in south-central Washington (Allen, 1980). The

second nest also was depredated following completion of the clutch. Clarke (2006) reported that one of four radio-marked pairs with a failed first nesting attempt re-nested in 1 year; the second nest was placed 332 m from the original nest. Six of 13 pairs re-nested in a second year, with two of those re-nesting twice, and re-nest distance ranged from 0.9 to 6 km from the original nest. In Nevada rangeland and hayfields, Hartman and Oring (2009) reported a high proportion of re-nesting after initial nest attempts, but no cases of double brooding.

Historically occupied sites are reused by curlews every year, and some individual birds may reuse the same territories from year to year (McCallum and others, 1977; Allen, 1980; Redmond and Jenni, 1982, 1986; Cannings, 1999). In South Dakota, 15 of 26 radio-marked adults returned to their breeding site of the previous year; nests were placed 0.1 to 1.1 km from previous nest sites (Clarke, 2006).

Species' Response to Management

Burning can improve habitat for Long-billed Curlews by removing shrubs and increasing habitat openness (Pampush and Anthony, 1993). During the breeding season after a fall range fire, there was a 30-percent increase in the estimated curlew breeding density in western Idaho (Redmond and Jenni, 1986). However, in central South Dakota mixed-grass prairies, curlew density did not differ between fall-burned and unburned pastures (Clarke, 2006).

Haying can be used to provide the short vegetation preferred by nesting curlews (Cochran and Anderson, 1987). In Nevada, hayfields provide suitable habitat for nesting Long-billed Curlews by providing optimal brood-rearing habitat that results in high chick survival (Hartman, 2008). Long-billed Curlews preferred nesting in tame hayfields and open rangeland more so than in shrub-desert rangeland (Hartman and Oring, 2009). Hayfields were irrigated from melting snow, grazed with as many as 4 cattle per ha until early May, and then hayed in mid- to late July; rangeland and shrub-desert plots received low-intensity cattle grazing of <0.25 cattle per ha (Hartman, 2008). Mammal predation accounted for most nest failures, but raking of irrigated hayfields, trampling by cattle, cattle-induced nest abandonment, and flooding owing to irrigation also contributed to nest failures. No chicks were lost to mowing or other ranching activities (Hartman, 2008). Broods that hatched in rangeland moved to hayfields within days of hatching, and no chick mortality was attributed to agricultural activity (Hartman and Oring, 2009).

In Wyoming hay meadows, ranchers traditionally scattered cow dung from fall- and winter-pastured cattle using branches, logs, or harrows (Cochran and Anderson, 1987). This practice, termed "dragging," was detrimental to nesting birds because curlews often built nests near cow dung. The practice generally has declined since the 1960s but still can be common locally. In north-central Oregon, alfalfa (*Medicago sativa*) fields were used for foraging as long as vegetation

remained <30 cm tall (Pampush, 1980; Pampush and Anthony, 1993). In Alberta, however, Long-billed Curlews did not use haylands (Prescott, 1997).

Grazing can be beneficial if it provides suitably short vegetation, particularly during the prelaying period (Bicak and others, 1982; Cochran and Anderson, 1987). In Colorado, Montana, Nebraska, North Dakota, South Dakota, and Wyoming, Long-billed Curlews preferred lightly grazed areas with aridic ustoll and aridic borollic soils and heavily grazed areas with typic ustoll soils (Kantrud and Kologiski, 1982). In southern Alberta, Long-billed Curlews used only continuously grazed mixed-grass pastures and were absent from mixed-grass pastures grazed in early summer, spring-grazed tame pastures, and deferred-grazed (grazed after July 15) mixed-grass pastures (Prescott and others, 1993). In Nebraska, curlews were present on grazed areas and were absent from ungrazed areas (Cole and Sharpe, 1976). In Colorado, curlew response to grazing over large areas of mixed-grass and shortgrass prairies was variable, but response to grazing in shrubsteppe habitats was negative (Bock and others, 1993). In Wyoming, nests in areas that were grazed during the incubation period had lower hatching success rates than nests in ungrazed areas (Cochran and Anderson, 1987).

In central South Dakota mixed-grass prairies, curlew density was not related to American bison (*Bison bison*) or cattle density (densities ranged from 0 to 223 bison per square kilometer [km^2] and from 0 to 42 cattle per km^2), and there was no difference in curlew densities between pastures grazed by bison, by cattle, or ungrazed pastures (Clarke, 2006). However, risk of nest trampling was dependent on livestock density. In 1 year, 3 of 15 nests were trampled by bison, with nest trampling starting at a bison density of 218 bison per km^2 . In the second year, 5 of 27 nests were trampled by bison, and 3 of 27 nests were trampled by cattle, with nest trampling starting at 77 bison per km^2 and 33 cattle per km^2 . Daily nest survival rates were negatively related to density of bison grazed in pastures containing nests. Sugden (1933) cautioned that sheep are more likely to trample nests than cattle. Of 119 nests in western Idaho, 4.2 percent were lost to trampling by livestock (Redmond and Jenni, 1986). Of 25 nests that failed owing to ranching operations in Nevada, such as from raking and trampling, 18 were in irrigated hayfields (Hartman, 2008). Ten nests were trampled by cattle: 7 of 94 nests in hayfields and 3 of 30 nests in rangeland.

In southwestern Idaho, Long-billed Curlews preferred recently grazed areas and avoided areas that had not been grazed within the past year (Bicak and others, 1982). Rotational and deferred grazing may provide suitable habitat, but year-long grazing was not recommended. Areas grazed by sheep alone or sheep and cattle had higher densities of curlews than did areas grazed by cattle alone. Pastures that included sheep in the grazing regime had more area of short grass (32 percent of area sampled <10 cm tall) than pastures grazed by cattle alone (19 percent of area sampled <10 cm tall). Curlew density was negatively correlated with height and vertical density of vegetation, and height of vegetation was negatively

correlated with grazing intensity and animal stocking rates. Sheep, however, were less likely than cattle to follow established routes through the grassland, and thus sheep trampled and reduced the amount of dead vegetation to a greater extent than did cattle. Neither cattle nor sheep could graze dense stands of perennial wheatgrasses, such as crested wheatgrass (*Agropyron cristatum*), to a height that was suitable to curlews. In northwestern South Dakota, Long-billed Curlews were seen in pastures with cattle as well as in unoccupied pastures, but no curlews were observed in pastures with sheep (Timken, 1969).

Long-billed Curlews prefer grazed prairies but will forage and occasionally even nest in cropland, including fallow fields, forage crops, and grain crops (McCallum and others, 1977; Pampush, 1980; Renaud, 1980; Cochran and Anderson, 1987; Pampush and Anthony, 1993; Saunders, 2001; Devries and others, 2010). However, Renaud (1980) reported that Long-billed Curlews avoided large cultivated areas in Saskatchewan. In the Platte River Valley of Nebraska, conversion of upland prairies to cropland had a negative impact on curlews through the destruction of nesting habitat (Faanes and Lingle, 1995). Long-billed Curlews in the Oklahoma Panhandle frequently used areas with a mix of shortgrass pastures and cropland, which often was planted to wheat (Shackford, 1987). In Alberta, Long-billed Curlews were more common in mixed-grass prairies than in cultivated areas (Owens and Myres, 1973). In central South Dakota, Long-billed Curlew adults were observed in a bare, disked field (Spomer, 1981). The only two nests found in cropland during a 3-year Oklahoma study were destroyed by agricultural operations (Shackford, 1994). Researchers suggested that Long-billed Curlews may experience better nesting success in wheat fields than in fields that are being prepared for plowing. Cochran and Anderson (1987) suggested that, although hayfields in Wyoming that had been cultivated may provide suitable vegetation and bare ground, they lacked elevated mounds and hummocks preferred for nesting. Nests in hayfields and pastures that were fertilized had lower success rates than nests in unfertilized fields, presumably because of disturbances caused by mechanical field operations.

Pesticides can be detrimental to Long-billed Curlews (Blus and others, 1985). Three Long-billed Curlews suffering convulsions or displaying erratic behavior were collected in northeastern Oregon. One male curlew appeared to have died of dieldrin poisoning (5.9 parts per million (ppm) tissue fresh weight) and another of chlordane poisoning (4.8 ppm fresh weight heptachlor epoxide and 4.4 ppm fresh weight oxychlordane). The third, a female, may have sustained lethal injuries as a result of impairment from poisoning (2.2 ppm fresh weight heptachlor epoxide and 2.7 ppm fresh weight oxychlordane). Seven eggs collected in the same region all contained 4.26 ppm fresh weight of dichlorodiphenyldichloroethylene (DDE), and some (numbers not given) contained low concentrations of polychlorinated biphenyls and chlordane metabolites. Blus and others (1985) suggested that concentrations of contaminants in the eggs were too low to influence

the reproductive success of Long-billed Curlews substantially. Peakall (1976) reported that one individual from Alberta had 14 ppm wet weight of DDE and 0.05 ppm of polychlorinated biphenyls.

Management Recommendations from the Literature

Preventing conversion of upland prairie to cropland is important in maintaining suitable habitat for Long-billed Curlews during the breeding season (Faanes and Lingle, 1995). To maintain healthy breeding populations, Saalfeld and others (2010) emphasized the importance of providing grassland habitats of short stature, free of woody plants, and embedded in a landscape dominated by other grasslands. Habitat areas need to be at least three times as large as a Long-billed Curlew territory, which averages about 14 ha, because the species requires an unoccupied buffer strip 300–500 m wide around the boundary of a territory (Redmond and others, 1981).

Tall, dense residual vegetation should be removed before the prelaying period (March to April) so that adults do not have to leave their territories to forage (Redmond, 1986; R.L. Redmond, University of Montana, written commun. [n.d.]). Removal of residual vegetation from previous growing seasons is especially important after years of above-normal precipitation. Burning may improve habitat in some areas by reducing shrub coverage and increasing habitat openness (Redmond and Jenni, 1986; Pampush and Anthony, 1993). Haying and grazing can be used to provide the short vegetation and reduced vertical plant density preferred by nesting curlews, but these disturbances should be timed so that short vegetation is available early in the season and active nests are not destroyed (Cochran and Anderson, 1987).

In west-central Wyoming, it is not advisable to drag hayfields to break up cow dung; Long-billed Curlews prefer to nest near cow dung (Cochran and Anderson, 1987). However, in Idaho, curlews did not show a preference for nesting near cow dung, and R.L. Redmond (University of Montana, written commun. [n.d.]) suggested that dragging may be acceptable if it occurs after the breeding season when eggs or chicks are no longer vulnerable. Hayfields in Nevada have high conservation value because they provide optimal brood-rearing habitat and thus, high productivity (Hartman and Oring, 2009). The practice of irrigation may be necessary to create the vegetation growth needed for brood-rearing habitat, yet may cause some nests to be flooded, so care must be taken (Hartman, 2008). In Nevada, land raking should be completed before the peak of nest initiation, which is late April (Hartman, 2008). Removing cattle from hayfields earlier than mid-April might alleviate some nest failures caused by cattle trampling and disturbance. Coyote control may be the single best factor in increasing curlew populations. In Nevada hayfields, predation, especially by large mammalian predators such as coyotes, was the greatest cause of nest failure (Hartman and Oring, 2009). In the

final year of the study, nest success (52 percent) was more than twice the average from the previous 3 years after the removal of six coyotes from one of the focal ranches.

Grazing is beneficial in providing the short vegetative structure preferred by Long-billed Curlews, although timing and intensity of grazing treatments may need to be adjusted to environmental conditions and biological factors (Bicak and others, 1982; Cochran and Anderson, 1987; Bock and others, 1993; Clarke, 2006). For northern mixed-grass prairies, Clarke (2006) recommended reducing grazing pressure from April 10 to June 25 to reduce nest trampling by livestock. This recommendation may mean reducing cattle density below 33 cattle per km² and bison density below 220 bison per km², and, during years of drought or following a fire, reducing bison density below 77 bison per km². Grazing during the curlew incubation period should be avoided; in Wyoming, nests in areas that were grazed during incubation had lower hatching success rates than nests in other areas (Cochran and Anderson, 1987). Native grasslands need grazing to provide shorter cover for broods, but not too short so as to limit use for escape cover and shade; moderate grazing provides the patchy vertical distributions required for chicks to survive (Clarke, 2006). Rotational and deferred grazing may provide suitable habitat, but year-long grazing is not recommended (Bicak and others, 1982).

Curlews may nest in cropland, and where they have been known to do so, agricultural producers can incorporate fall-seeded crops, such as winter wheat and fall rye (*Secale* spp.), into their rotations to provide cropland habitats with reduced disturbances for nesting curlews (Devries and others, 2010).

Breeding habitat and nesting curlews should be protected from detrimental human activities, such as vehicular use, researcher disturbance, and shooting (Sugden, 1933; Redmond and Jenni, 1986). In Saskatchewan, abandonment of breeding sites by Long-billed Curlews was attributed to researcher disturbance (Maher, 1973, 1974).

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Table G1. Measured values of vegetation structure and composition in Long-billed Curlew (*Numenius americanus*) breeding habitat by study. The parenthetical descriptors following authorship and year in the “Study” column indicate that the vegetation measurements were taken in locations or under conditions specified in the descriptor; no descriptor implies that measurements were taken within the general study area.

[cm, centimeter; %, percent; --, no data; ≤, less than or equal to; <, less than; spp., species (applies to two or more species within the genus)]

Study	State or province	Habitat	Management practice or treatment	Vegetation height (cm)	Vegetation height-density (cm)	Grass cover (%)	Forb cover (%)	Shrub cover (%)	Bare ground cover (%)	Litter cover (%)	Litter depth (cm)
Ackerman, 2007 (nests)	North Dakota	Mixed-grass prairie, tame grassland	--	2.8–13.1	7.4–10	--	--	--	--	--	0.1–0.5
Allen, 1980 (nests)	Washington	Tame grassland	Idle	≤20 ^a	--	--	--	--	--	--	--
Bicak and others, 1982	Idaho	Shortgrass prairie, tame grassland	Grazed	<10	--	--	--	--	--	--	--
Clarke, 2006 ^b (nests)	South Dakota	Mixed-grass prairie	Grazed	13	10.2–27.0	45.6–55.1	10.8–46.9	0–0.03	8.5–36.2	--	--
Clarke, 2006 ^b (broods)	South Dakota	Mixed-grass prairie	Grazed	12	10.4–20.3	34.6–59.4	5.3–10.8	0.1–1.1	16.1–53.2	--	--
Cochran and Anderson, 1987 (study plots)	Wyoming	Tame grassland	Multiple	--	--	71.4 ^c	10.1	--	9.3	--	--
Cochran and Anderson, 1987 (study plots)	Wyoming	Shortgrass prairie	Multiple	--	--	57.9 ^c	7.8	--	22.7	--	--
Cochran and Anderson, 1987 (nest fields)	Wyoming	Shortgrass prairie, tame grassland	Multiple	--	--	19.9	15.5	--	--	--	--
Cochran and Anderson, 1987 (nests)	Wyoming	Shortgrass prairie, tame grassland	Multiple	--	--	61	--	--	7	--	--
Gregory and others, 2011 (nests)	Nebraska	Mixed-grass prairie	--	5–75	0.5–6.8	--	--	--	--	--	0–2.3
King, 1978 (nests)	Colorado, Texas	Multiple	Grazed	11	--	--	--	--	--	--	--
Pampush, 1980	Oregon	Multiple	Grazed	25–33	--	30–53	4–20	2–12	26–40	--	--
Paton and Dalton, 1994 (nests)	Utah	Multiple	--	6.5	--	--	--	--	18	--	--
Redmond, 1986	Idaho	Shortgrass prairie	Grazed	3.6–15.7	--	--	--	--	--	--	--
Spomer, 1981 (broods)	South Dakota	Mixed-grass prairie	--	18 ^d	--	--	--	--	--	--	--

^aHeight of Sandberg’s bluegrass (*Poa secunda*) and downy brome (*Bromus tectorum*).

^bThe sum of the percentages is greater than 100%, based on methods described by the authors.

^cValue includes grasses, sedges (*Carex* spp.), and rushes (*Juncus* spp.).

^dGrass height.

The Effects of Management Practices on Grassland Birds—Marbled Godwit (*Limosa fedoa*)

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Chapter H of

The Effects of Management Practices on Grassland Birds

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Figure

H1. Map showing breeding distribution of the Marbled Godwit (<i>Limosa fedoa</i>) in the United States and southern Canada, based on North American Breeding Bird Survey data, 2008–12.....	2
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Table

H1. Measured values of vegetation structure and composition in Marbled Godwit (<i>Limosa fedoa</i>) breeding habitat by study.....	9
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Conversion Factors

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		
square meter (m ²)	0.0002471	acre
hectare (ha)	2.471	acre
square kilometer (km ²)	247.1	acre
square meter (m ²)	10.76	square foot (ft ²)
hectare (ha)	0.003861	square mile (mi ²)
square kilometer (km ²)	0.3861	square mile (mi ²)

Abbreviations

n.d. no date

PPR Prairie Pothole Region

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The Effects of Management Practices on Grassland Birds—Marbled Godwit (*Limosa fedoa*)

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Capsule Statement

Keys to Marbled Godwit (*Limosa fedoa*) management include providing large expanses of short, sparsely to moderately vegetated landscapes that include native grasslands and wetland complexes. Optimal wetland complexes should contain a diversity of wetland classes and sizes, such as ephemeral, temporary, seasonal, semipermanent, permanent, and alkali wetlands (wetland classifications based on Stewart and Kantrud, 1971), as well as intermittent streams. Marbled Godwits use wetlands of various salinities. The species has been reported to use habitats with less than or equal to 70 centimeters (cm) average vegetation height, 4–23 cm visual obstruction reading, and 1–9 cm litter depth. The descriptions of key vegetation characteristics are provided in table H1 (after the “References” section). Vernacular and scientific names of plants and animals follow the Integrated Taxonomic Information System (<https://www.itis.gov>).

Breeding Range

Marbled Godwits breed from central Alberta through central Manitoba and along James Bay; south through Montana, North Dakota, east-central South Dakota, and north-central Nebraska; and east to north-central Minnesota (National Geographic Society, 2011). The relative densities of Marbled Godwits in the United States and southern Canada, based on North American Breeding Bird Survey data (Sauer and others, 2014), are shown in figure H1 (not all geographic places mentioned in report are shown on figure).



Marbled Godwit. Illustration by Patsy D. Renz, used with permission.

Suitable Habitat

Breeding Marbled Godwits require large expanses of short, sparsely to moderately vegetated uplands for nesting and foraging, and wetland complexes for foraging (Stewart, 1975; Ryan, 1982; Ryan and others, 1984; Kantrud and Higgins, 1992; Niemuth and others, 2012). Marbled Godwit territories are characterized by a high percentage of grass cover, many wetlands, and high wetland diversity (Stewart, 1975; Ryan, 1982; Kantrud and Higgins, 1992). In upland and wetland habitats, tall, dense cover is avoided (Nowicki, 1973; Higgins and others, 1979; Ryan, 1982; Renken, 1983; Ryan and others, 1984; Renken and Dinsmore, 1987). In Manitoba, Montana, North Dakota, and South Dakota, Marbled Godwits used areas with less than (<) 40 percent dead vegetation, and they avoided areas with 100 percent visual obstruction above 10 cm and areas with greater than (>) 35 cm effective cover height (average maximum height of leaf canopy) (Kantrud and Higgins, 1992). Marbled Godwits with broods use somewhat taller (15–60 cm) and denser grass cover than do nesting pairs (Ryan and others, 1984). Foraging occurs in water 5–13 cm deep (Gratto-Trevor, 2000).

¹U.S. Geological Survey.

²South Dakota Game, Fish and Parks (current).

³U.S. Fish and Wildlife Service (current).

2 The Effects of Management Practices on Grassland Birds—Marbled Godwit (*Limosa fedoa*)

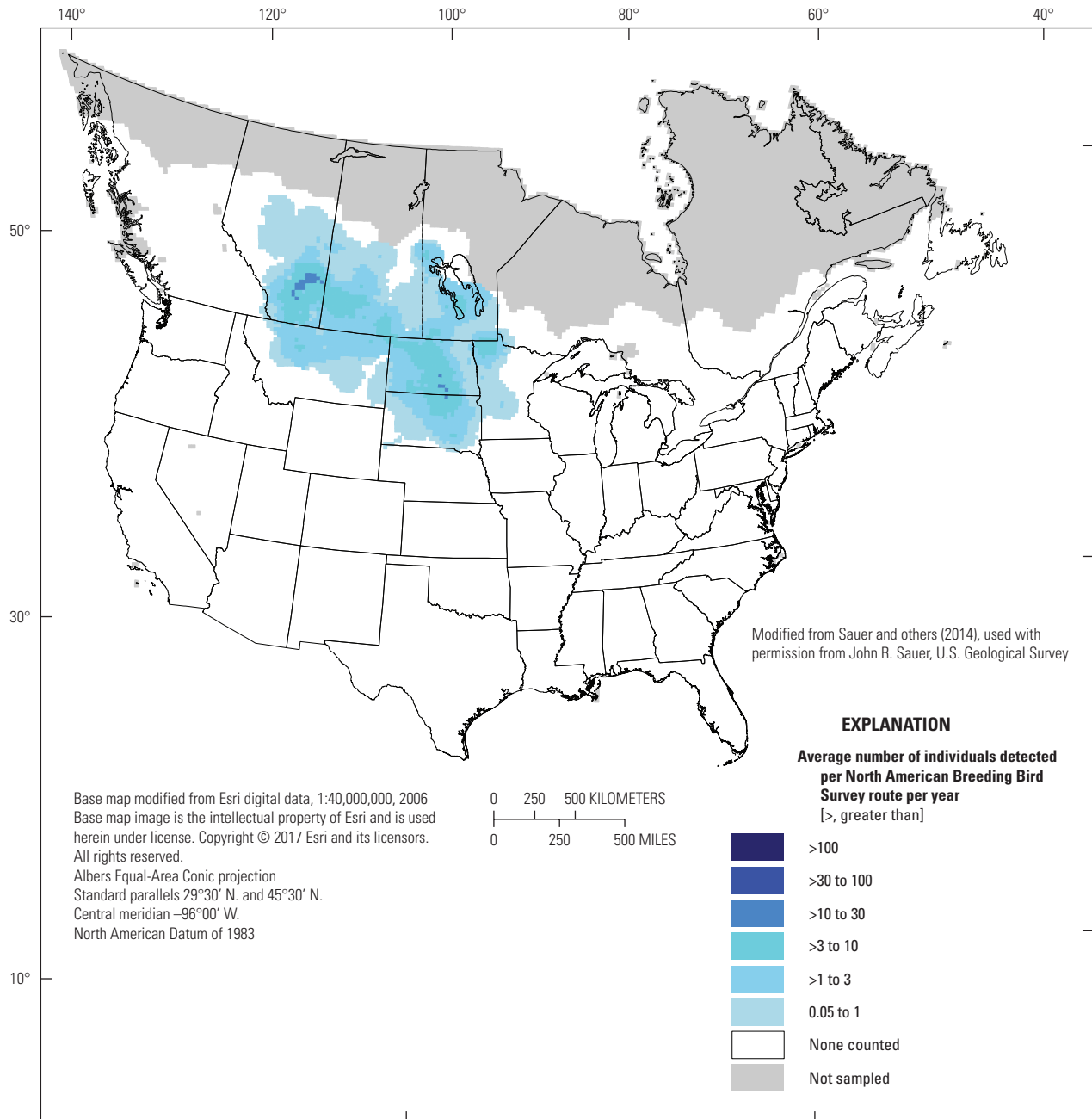


Figure H1. Breeding distribution of the Marbled Godwit (*Limosa fedoa*) in the United States and southern Canada, based on North American Breeding Bird Survey (BBS) data, 2008–12. The BBS abundance map provides only an approximation of breeding range edges.

Marbled Godwits nest on the ground, often in grasslands and well away from water edges (Gratto-Trevor, 2000). In southern Alberta, average distance between 62 nest sites and water was 239 meters (m) near managed wetlands and 258 m for three nest sites near natural wetlands (Gratto-Trevor, 2000). In the same study area in southern Alberta, Gratto-Trevor (2006) reported that Marbled Godwits nested in managed wetlands more than natural wetlands, possibly owing to low water levels in the natural wetlands. In Saskatchewan,

Marbled Godwits nested in uplands and wetland margins with denser, taller, and more homogeneous vegetation than random sites (Colwell and Oring, 1990). In Alberta, Saskatchewan, and Manitoba, Marbled Godwits selected nesting habitat in proportion to what was available, with cropland and idle native prairies selected more often than grazed prairies or wetlands (Garvey and others, 2013). In North Dakota, Marbled Godwits nested in wet and dry areas of wet meadow, upland areas of short (<30 cm) grass, and idle mixed-grass hayland;

they foraged in dry uplands, wet and dry areas of wet meadow, roadside ditches, and open water (Nowicki, 1973). In another North Dakota study, hatching success was similar between nests in cultivated fields and nests in native grasslands (Higgins and others, 1979).

Marbled Godwits prefer native grass cover to tame vegetation (Stewart, 1975; Ryan, 1982; Ryan and others, 1984; Kantrud and Higgins, 1992; Prescott and others, 1995; Prescott, 1997). Pastures, idle grasslands, and haylands often are used for nesting (Higgins and others, 1979; Ryan and others, 1984; Kantrud and Higgins, 1992; Garvey and others, 2013). Although tilled lands usually are avoided (Weber, 1978; Ryan and others, 1984), nests also have been reported in cropland, including small grains, common flax (*Linum usitatissimum*), and stubble fields (Stewart, 1975; Higgins and others, 1979; Kantrud and Higgins, 1992; Garvey and others, 2013). In the northern prairie and aspen parkland regions of Alberta, Marbled Godwits were most abundant in idle mixed-grass pastures, followed by sandhills (mixed-grass prairie containing sandy soils), hayland (planted to unspecified grasses or alfalfa [*Medicago sativa*]), fallow cropland, and tame pastures (Prescott and others, 1995; Prescott, 1997). In the aspen parkland uplands, Marbled Godwits were most abundant on idle mixed-grass pastures, followed by mixed-grass pastures grazed season long (Prescott and others, 1995). They were not found in idle tame grassland, tame dense nesting cover, tame pastures, deferred (mowed after July 15) tame hayland, deferred mixed-grass pastures (grazed after July 15), idle parkland, season-long grazed parkland, native dense nesting cover, hayland, or cropland.

Soil types may affect the availability of preferred habitats. In North Dakota, Marbled Godwits were associated with silty range, thin upland range, and shallow-to-gravel range sites (Messmer, 1990; Sedivec, 1994). Silty range and thin upland range sites were characterized by thin topsoil, loamy soil, 1–25-percent slope, grassy cover, low shrub cover, and moderate-to-high litter cover. Shallow-to-gravel range sites were characterized by sparse cover and reduced litter.

Niemuth and others (2012) observed seasonal shifts in habitat use between wetlands and uplands in that the detections of Marbled Godwits over a 7-week survey period spanning mid-May to late June were initially high in upland habitats but decreased with concomitant increases in wetland habitats. Shifts in wetland use occur seasonally and during climatic extremes, as breeding Marbled Godwits use less-permanent wetlands early in the breeding season and move to semipermanent and alkali wetlands later in summer or during drought (Ryan and others, 1984; Gratto-Trevor, 2000).

Within wetland habitats, Marbled Godwits avoid dense emergent vegetation, preferring shallow water areas with short and sparse vegetation or moderately vegetated shorelines (Ryan, 1982; Ryan and others, 1984; Eldridge, 1992). Suitable wetlands range in salinity from fresh to highly saline and vary widely in size and permanence (Stewart and Kantrud, 1965; Stewart, 1975; Ryan and others, 1984; Eldridge, 1992; Prescott and others, 1995). In a survey of 1,190 wetlands

throughout the Prairie Pothole Region (PPR) of North Dakota and South Dakota, Marbled Godwits were observed in a higher proportion of alkali or permanent wetlands than in temporary, seasonal, or semipermanent wetlands (Igl and others, 2017). Marbled Godwits were observed in 44 wetlands, which were characterized as having an average of 59 percent open water, 18 percent emergent vegetation, 16 percent wet meadow, and 5 percent shore/mudflat. Within wetlands in the PPR of North Dakota, Kantrud and Stewart (1984) most frequently observed breeding Marbled Godwits in seasonal wetlands, followed by semipermanent, temporary, and alkali wetlands, but their density was highest on temporary wetlands. In east-central North Dakota, semipermanent wetlands were used most often, but ephemeral, alkali, and temporary wetlands were used relative to their availability (Ryan and others, 1984).

Niemuth and others (2008) devised a conceptual model to predict Marbled Godwit habitat quality in the northern Great Plains. The model indicated that high-quality habitat consisted of at least 1.6 hectares (ha) of temporary or saturated wetlands per 130-ha patch, with a patch size that is at least 130 ha and at least 400 m wide (800 m preferred), >100 m from trees, in a landscape (3.2 kilometer [km] radius) with at least 10–30 percent grassland (>30 percent better), and with less than or equal to 3 percent average slope within a 535-m radius. The essential elements from this model were then formalized into rules. From these rules, maps were developed to depict areas of high predicted occurrence of Marbled Godwits in North Dakota and South Dakota. High occurrences of Marbled Godwits generally coincided with areas of high potential waterfowl densities, and waterfowl densities increased from south to north and east to west for all wetland classes examined, with higher waterfowl densities occurring on smaller wetlands. In the PPR of eastern Montana, North Dakota, and South Dakota, Niemuth and others (2012) evaluated wetland characteristics that influenced the detection and number of Marbled Godwits. The detection of Marbled Godwits exhibited a curvilinear relationship with wetland perimeter, and detections increased as the proportion of wetland surrounded by a grass buffer increased. Moreover, detections of Marbled Godwits were positively related to characteristics indicative of wetlands with low amounts of emergent vegetation: the amount of open water or bare soil covering >95 percent of the wetland area, the proportion of wetland covered by water, and the width of mudflats. Detections and number of Marbled Godwits were positively related to brackish or saline wetlands. In North Dakota and South Dakota, Niemuth and others (2013) reported that Marbled Godwit detections increased curvilinearly with the proportion of the wetland basin containing water. However, in South Dakota, Marbled Godwit presence was positively associated with wetlands containing dense stands of emergent vegetation, with open water or bare soil covering <5 percent of the wetland, and with adjacent uplands of alfalfa hayland; presence was negatively associated with wetlands that had adjacent tilled fields (Weber, 1978; Weber and others, 1982).

Area Requirements and Landscape Associations

Marbled Godwit territories are large and include feeding and nesting areas. Areas must be large enough to provide upland habitat and a diverse range of wetland types (Kantrud and Stewart, 1984; Ryan and others, 1984; Colwell and Oring, 1988a). In North Dakota, mean territory size was 90 ha (Ryan and others, 1984). Marbled Godwits may be area sensitive, rarely occurring on blocks of contiguous grassland <50 ha in the northern Great Plains (Johnson and Igl, 2001). Of 44 wetlands in the PPR of North Dakota and South Dakota in which Marbled Godwits were observed, average wetland size was 21 ha (Igl and others, 2017). Landscape composition within 800 m of these wetlands was 61 percent grassland, 18 percent wetland, 16 percent agricultural, and 5 percent other; average number of wetlands within 800 m was 22. In tallgrass prairies in southeastern North Dakota, occurrence of Marbled Godwit was positively associated with wetland cover at the 100-m scale, negatively associated with woodland cover at the 100-m scale, and negatively associated with tree cover at the 400-m and 800-m scales (Cunningham and Johnson, 2006). In the PPR of Montana, North Dakota, and South Dakota, detections of Marbled Godwits were positively related to the percentage of an 800-m buffer around survey points consisting of a mixture of native grass, forb, or scattered low shrub species on untilled prairie and to the percentage of area within the buffer consisting of temporary, seasonal, semipermanent, and permanent wetlands (Niemuth and others, 2012). In a preliminary effort to model Marbled Godwit occurrence and habitat associations in the Prairie Habitat Joint Venture region in Canada, Marbled Godwit occurrence was positively related to the presence of grassland and wetland areas in the landscape within 1,200 square meters of godwit observations; Marbled Godwit occurrence was negatively associated with the presence of roads and trees (S. Davis, pers. commun. [n.d.] in Melcher and others, 2010).

Brood Parasitism by Cowbirds and Other Species

The Marbled Godwit is an unsuitable host of the Brown-headed Cowbird (*Molothrus ater*), and no known records of brood parasitism exist (Shaffer and others, 2019).

Breeding-Season Phenology and Site Fidelity

The Marbled Godwit breeding season extends from mid-April through late July (Maher, 1973; Stewart, 1975; Kantrud and Higgins, 1992; Sedivec, 1994; Gratto-Trevor, 2000). The

earliest reported nest with eggs was April 17 (Stewart, 1975), with most nests initiated during mid- to late May (Maher, 1973; Kantrud and Higgins, 1992; Sedivec, 1994). Kantrud and Higgins (1992) reported a late hatching date of June 27, and Stewart (1975) observed a dependent brood on July 18. One brood is produced per season (Gratto-Trevor, 2000). Although Higgins and others (1979) reported that Marbled Godwit pairs appeared to make only one nesting attempt per breeding season, Ryan and others (1981) and Gratto-Trevor (2000) reported that renesting occurred after failure of the initial nest. Large postbreeding flocks of Marbled Godwits begin forming in late June and early July in central North Dakota (L.D. Igl, U.S. Geological Survey, unpub. data) to mid- to late July in Saskatchewan (Maher, 1973), and most flocks depart for the wintering grounds by late August (Ryan and others, 1984). In Saskatchewan and Alberta, Marbled Godwits exhibited breeding-site fidelity (Colwell and Oring, 1988b; Gratto-Trevor, 2000).

Species' Response to Management

Although burning, mowing, or grazing are necessary to maintain suitable habitat for Marbled Godwits (Ryan and others, 1984), few studies have examined the influence of burning or mowing on this species. In North Dakota mixed-grass prairies, Marbled Godwit densities were highest during the first 2 years after a burn (Johnson, 1997). Ryan and others (1984) indicated that fall burning or haying could provide suitable nesting habitat the following spring, and the denser, taller regrowth (15–60 cm) could provide suitable habitat for broods. Haylands are readily used by breeding Marbled Godwits (Ryan and others, 1984; Kantrud and Higgins, 1992). Direct or indirect adult mortalities associated with haying, mowing, and other land-management operations are likely minimal because godwits are not known to make frequent use of habitats likely to be mowed during the breeding season (C.L. Gratto-Trevor, pers. commun. [n.d.] in Melcher and others, 2010). More information is needed to determine whether these activities represent a significant source of godwit nest failure or chick mortality.

Grazing can be used in upland and wetland habitats to maintain the short, moderately dense vegetation preferred by Marbled Godwits (Ryan and others, 1984). Grazed or recently grazed uplands often are more attractive to breeding Marbled Godwits than are other land-use types (Ryan and others, 1984; Renken and Dinsmore, 1987; Kantrud and Higgins, 1992; Sedivec, 1994). In Saskatchewan, no significant difference in godwit abundance was found between lightly grazed mixed-grass pastures and lightly grazed stands of crested wheat-grass (*Agropyron cristatum*) (Sutter and Brigham, 1998). In Manitoba mixed-grass prairies, Marbled Godwits occurred in season-long (grazed from May through October) and twice-over rotational-grazed (grazed from June to mid-October with cattle rotated between 3 and 6 pastures) pastures, but avoided

idle pastures (Ranellucci, 2010). In North Dakota, density of Marbled Godwits did not differ among season-long, short-duration pastures (rotated through a grazing schedule of about 1 week grazed and 1 month ungrazed, repeated throughout the season), twice-over rotation pastures (grazing twice per season, with about a 2-month rest between grazing), and idle pastures (Messmer, 1990). In south-central North Dakota mixed-grass prairies, Marbled Godwits only occurred in heavily and extremely grazed pastures (20–35 percent of forage produced in an average year remained, equating to an average grazing rate of 4.2–6.8 animal unit months per ha) and not in lightly or moderately grazed pastures (50–65 percent, 1.1–2.4 animal unit months per ha) (Salo and others, 2004). Occurrence and densities increased as grazing intensity increased. In South Dakota mixed-grass prairies, Ahlering and Merkord (2016) reported no relationship between grazing intensity or burning activity and Marbled Godwit abundance; abundance did increase with greater variability in litter depth unrelated to grazing intensity.

Marbled Godwits may be affected by energy development and habitat edges. Niemuth and others (2013) examined the influence of two wind facilities in North Dakota and South Dakota on Marbled Godwit for 3 years. The species did not appear to avoid wetland basins within 805 m of wind turbines at either facility, although occurrence was slightly and consistently lower at one facility, possibly because that facility was located primarily in cropland and the other facility in grassland. In Alberta, Marbled Godwit abundance decreased by 25 percent within 0.1 km of roads and within 1 km of wetland edges; no effect was found for distance to cropland edges (Sliwinski and Koper, 2012). Godwit injury and mortality have been reported where powerlines bisect shallow wetlands (Melcher and others, 2010).

Management Recommendations from the Literature

Melcher and others (2010) identified habitat loss and fragmentation attributed to agricultural conversion of native prairies and wetlands as the greatest threat to Marbled Godwit populations in their midcontinental breeding range. Habitat protection is thus the highest conservation priority for this species. Protecting and restoring wetlands that are part of large, contiguous grasslands are important for maintaining suitable breeding habitat for Marbled Godwits. Providing a diverse complex of wetlands may be beneficial to breeding Marbled Godwits (Kantrud and Stewart, 1984; Ryan and others, 1984; Colwell and Oring, 1988a). Marbled Godwits use wetlands of widely varying types and salinities and may need to utilize larger, more-permanent wetlands during droughts or late in summer (Ryan and others, 1984; Melcher and others, 2010). Shallow-water ponds with little or no emergent vegetation are useful for pre- and postbreeding flocks, and shallow-water ponds with margins of emergent vegetation are useful for

broods (Gratto-Trevor, 2000). Natural wetlands should be protected from drainage (Ryan and others, 1984), and drained wetlands should be restored (Berkey and others, 1993; Johnson, 1996). Managed wetlands are important, especially in dry years during the breeding season, when they could be the only suitable habitat in the grassland for nesting or foraging (Gratto-Trevor, 2006). Protected habitats should be extensive enough (larger than 100 ha) to provide both upland habitat and a diverse range of wetland types (Stewart, 1975; Colwell and Oring, 1988a; Kantrud and Higgins, 1992; Gratto-Trevor, 2000; Melcher and others, 2010). Territories averaged 90 ha in North Dakota (Ryan and others, 1984), and Marbled Godwits may require large (>50 ha) blocks of contiguous grassland habitat (Johnson and Igl, 2001).

Native grassland habitat should be provided for upland nesting and foraging (Ryan and others, 1984; Eldridge, 1992; Kantrud and Higgins, 1992; Gratto-Trevor, 2000). Grassland restoration of agricultural fields through programs such as the Conservation Reserve Program are beneficial to Marbled Godwits, and the continuation and expansion of agricultural policies with grassland wildlife components are essential for maintaining populations of Marbled Godwits (Niemuth and others, 2008). Efforts to conserve waterfowl also will benefit Marbled Godwits owing to preferences for similar landscapes (Niemuth and others, 2008). Cunningham and Johnson (2006) recommended removal of trees to improve grassland habitats.

Habitat loss and degradation attributed to agricultural conversion is a significant threat to breeding populations of Marbled Godwits (Melcher and others, 2010). Upland and wetland habitats should be protected from tilling (Ryan and others, 1984; Melcher and others, 2010). Encouraging no-tillage and minimum-tillage practices on cropland may benefit Marbled Godwits (Kantrud and Higgins, 1992). Burning, mowing, or grazing can be used to provide areas of shorter, sparser vegetation (Ryan and others, 1984; Eldridge, 1992; Berkey and others, 1993). Fall burning or mowing of upland sites and wetland edges may produce suitable cover during the following spring (Ryan and others, 1984). Moderate-to-dense regrowth in burned areas may be too dense for nesting but may provide the denser, taller cover used by broods (Ryan and others, 1984).

Marbled Godwits tolerate a range of habitat disturbances, including burning, mowing, and grazing, but no clear management guidelines are evident from current studies. Burning or mowing prior to the breeding season may be preferable so that nests will not be destroyed by mechanical equipment. Results of studies on the effect of grazing on Marbled Godwits vary from no effect of grazing intensity (Ahlering and Merkord, 2016), to no differences among grazing systems (Messmer, 1990), to the species preferring extremely grazed pastures (Salo and others, 2004). The short, sparsely to moderately vegetated landscapes preferred by the species could be created by short-term grazing (2 to 4 weeks) in May, prior to the onset of breeding (Berkey and others, 1993), allowing birds to settle before implementing season-long grazing in mid-June (for example, Sedivec, 1994), or by deferring grazing until late

May or late June for rotational grazing (Sedivec, 1994; Gratto-Trevor, 2000), although none of these suggestions have been rigorously examined for their effect on Marbled Godwits.

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Table H1. Measured values of vegetation structure and composition in Marbled Godwit (*Limosa fedoa*) breeding habitat by study. The parenthetical descriptors following authorship and year in the “Study” column indicate that the vegetation measurements were taken in locations or under conditions specified in the descriptor; no descriptor implies that measurements were taken within the general study area.

[cm, centimeter; %, percent; --, no data; <, less than; DNC, dense nesting cover]

Study	State or province	Habitat	Management practice or treatment	Vegetation height (cm)	Vegetation height-density (cm)	Grass cover (%)	Forb cover (%)	Shrub cover (%)	Bare ground cover (%)	Litter cover (%)	Litter depth (cm)
Garvey and others, 2013 (nests)	Alberta, Manitoba, Saskatchewan	Multiple	Multiple	--	7 ^a	--	--	--	--	--	--
Higgins and others, 1979 (nests)	North Dakota	Mixed-grass prairie	Multiple	<15	--	--	--	--	--	--	--
Kantrud and Higgins, 1992 (nests)	Manitoba, Montana, North Dakota, South Dakota	Multiple	Multiple	--	4 ^a , 17 ^b	--	--	--	--	36 ^c	--
Nowicki, 1973 (nests)	North Dakota	Multiple	Multiple	<30	--	--	--	--	--	--	--
Renken, 1983 ^d	North Dakota	Tame grass-land (DNC)	Idle, grazed	--	7 ^a	49.9	18.8	5.5	0.7	99	2
Ryan and others, 1984 (adults)	North Dakota	Multiple	Multiple	<15	--	--	--	--	--	--	--
Ryan and others, 1984 (adults with broods)	North Dakota	Multiple	Multiple	15–60	--	--	--	--	--	--	--
Salo and others, 2004	North Dakota	Mixed-grass prairie	Heavy grazing intensity	27.1 ^e	22.9 ^a	--	--	--	--	--	2
Salo and others, 2004	North Dakota	Mixed-grass prairie	Light grazing intensity	17.5 ^e	7.9 ^a	--	--	--	--	--	0.9
Sedivec, 1994 ^f	North Dakota	Mixed-grass prairie	Multiple	50–70	6 ^a	--	--	--	--	--	3.8–9.1

^aVisual obstruction reading (Robel and others, 1970).

^bEffective vegetation height.

^cStanding dead vegetation.

^dThe sum of the percentages is greater than 100%, based on the modified point-quadrat technique of Wiens (1969).

^eMean grass height.

^fRange of average values.

The Effects of Management Practices on Grassland Birds—Mountain Plover (*Charadrius montanus*)

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Christopher M. Goldade,^{1,2} Melvin P. Nenneman,^{1,3} Travis L. Wooten,^{1,4} and
Betty R. Euliss¹

Chapter E 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^{1,5}

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DAVID BERNHARDT, Secretary

U.S. Geological Survey
James F. Reilly II, Director

U.S. Geological Survey, Reston, Virginia: 2019

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Figure

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Table

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Conversion Factors

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)
Area		
square meter (m ²)	0.0002471	acre
hectare (ha)	2.471	acre
square meter (m ²)	10.76	square foot (ft ²)
hectare (ha)	0.003861	square mile (mi ²)

Abbreviations

BBS Breeding Bird Survey

spp. species (applies to two or more species within the genus)

Acknowledgments

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The Effects of Management Practices on Grassland Birds—Mountain Plover (*Charadrius montanus*)

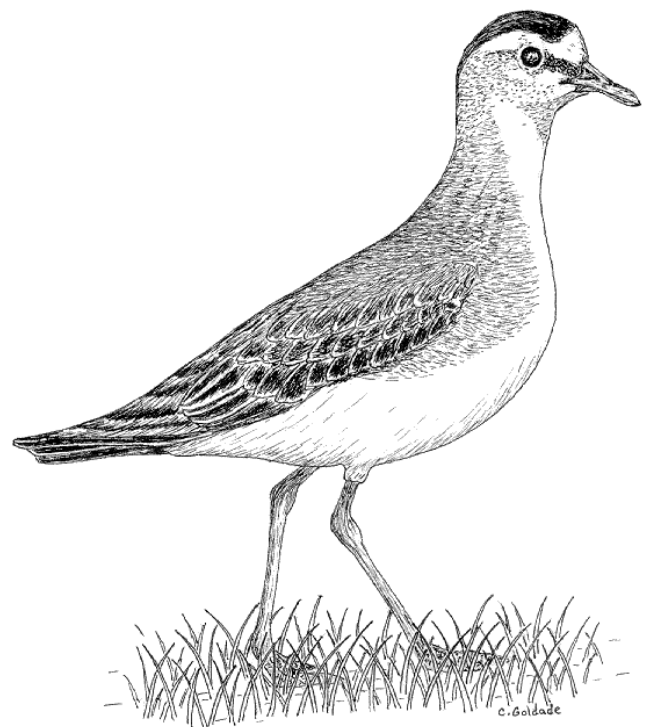
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Capsule Statement

The key to Mountain Plover (*Charadrius montanus*) management is maintaining sparsely vegetated grasslands. Grasslands can be made suitable for breeding Mountain Plovers by preserving large prairie dog (*Cynomys* species [spp.]) towns, conducting prescribed burns, or implementing heavy grazing in some situations. Mountain Plovers have been reported to use habitats with 2–38 centimeters (cm) average vegetation height, 14–87 percent grass cover, 2–14 percent forb cover, 4–55 percent shrub cover, 9–72 percent bare ground, 2 percent litter cover, and 4–6 cm litter depth. The descriptions of key vegetation characteristics are provided in table E1 (after the “References” section). Vernacular and scientific names of plants and animals follow the Integrated Taxonomic Information System (<https://www.itis.gov>).

Breeding Range

Mountain Plovers breed from southeastern Alberta and southwestern Saskatchewan through central Montana; south to south-central Wyoming, the southwestern portion of the Nebraska panhandle, east-central Colorado, and northeastern New Mexico; and east to northern Texas, northwestern Oklahoma, and western Kansas (Bly and others, 2008; McConnell and others, 2009; National Geographic Society, 2011). The relative densities of Mountain Plovers in the United States and southern Canada, based on North American Breeding Bird Survey (BBS) data (Sauer and others, 2014), are shown in figure E1 (not all geographic places mentioned in report are shown on figure). Although the Mountain Plover’s entire



Mountain Plover. Illustration by Christopher M. Goldade, U.S. Geological Survey.

breeding range is within in the area covered by the BBS, the species is considered to be poorly monitored by the BBS, with imprecise results largely owing to small sample sizes, a restricted distribution, and poor coverage of their habitats from along roadsides (Sauer and others, 2013). As such, the breeding range map does not depict locations with known but sparse breeding populations of Mountain Plovers in the northern part of the species’ range in Montana, southeastern Alberta, and southwestern Saskatchewan. In recent years, the species has expanded its breeding range into southwestern Nebraska and northwestern Oklahoma (Bly and others, 2008; McConnell and others, 2009).

¹U.S. Geological Survey.

²South Dakota Game, Fish and Parks (current).

³U.S. Fish and Wildlife Service (current).

⁴San Diego Zoo Institute for Conservation Research (current).

2 The Effects of Management Practices on Grassland Birds—Mountain Plover (*Charadrius montanus*)

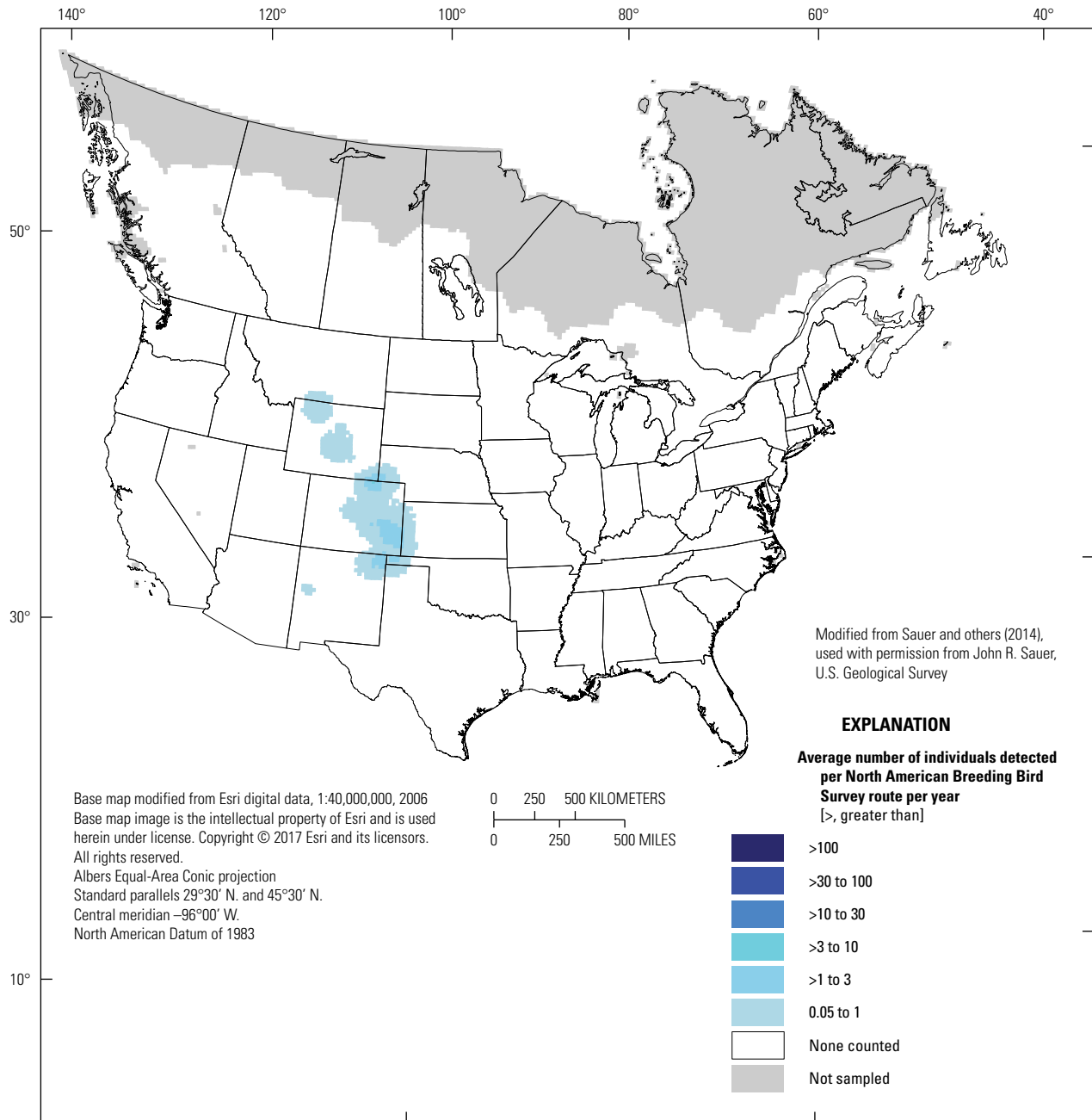


Figure E1. Breeding distribution of the Mountain Plover (*Charadrius montanus*) in the United States and southern Canada, based on North American Breeding Bird Survey (BBS) data, 2008–12. The BBS abundance map provides only an approximation of breeding range edges.

Suitable Habitat

Mountain Plovers prefer large, flat grassland expanses with sparse, short vegetation and bare ground within short-grass and mixed-grass prairies and shrubsteppe (Giezentanner, 1970; Graul, 1973, 1975; Knowles and others, 1982; Olson, 1984; Olson and Edge, 1985; Shackford, 1987; Wershler and Wallis, 1987; Leachman and Osmundson, 1990; Parrish and others, 1993; Knopf and Miller, 1994; Shackford and Leslie, 1994; Knowles, 1996; Ellison Manning and White, 2001;

Beauvais and Smith, 2003; Goguen, 2012). Within semidesert grasslands, Mountain Plovers use flat, sparsely vegetated habitats with cacti, low shrubs, and an open understory (Tolle, 1976; Day, 1994). Areas disturbed by prairie dogs, fire, or heavy grazing provide suitable habitat (Bradbury, 1918; Finzel, 1964; Giezentanner, 1970; Wallis and Wershler, 1981; Knowles and others, 1982; Knowles and Knowles, 1984; Olson, 1984; Shackford, 1987; Wershler and Wallis, 1987; Leachman and Osmundson, 1990; Knowles, 1996; Tipton and others, 2008; Augustine, 2011; Goguen, 2012). Mountain

Plovers frequently use cultivated fields for nesting and brood rearing (Shackford, 1987, 1991, 1996; Shackford and Leslie, 1994; Knopf, 1996; Knopf and Rupert, 1999; Shackford and others, 1999; Dreitz and others, 2005; Bly and others, 2008; McConnell and others, 2009).

Mountain Plovers tend to place nests in areas of sparse vegetation. The species often nests near rocks, cow dung, or clumps of vegetation (Graul, 1975; Wallis and Wershler, 1981; Olson and Edge, 1985; Parrish, 1988; Parrish and others, 1993; Knopf and Miller, 1994; Knopf, 1996). In Colorado, nest sites had a lower percentage of vegetative cover, more cow dung, and fewer pricklypear (*Opuntia* spp.) plants than nearby random sites (Knopf and Miller, 1994). In Utah shrubsteppe, Mountain Plover nest sites had shorter vegetation than random sites (Ellison Manning and White, 2001). In Montana, mean vegetation height at nest sites was lower than that in the surrounding habitat, although litter cover was higher (Olson, 1984; Olson and Edge, 1985). In northeastern Wyoming, Mountain Plovers nested in areas with shorter grasses (average height of 8.4 cm), forbs (average height of 4.3 cm), shrubs (average height of 3.7 cm), and distances to animal or wheel tracks; higher shrub density (average density of 12.3 shrubs per square meter [m²]); and taller cacti (average height of 6.7 cm) than random sites (Parrish, 1988; Parrish and others, 1993). Shrubs, primarily birdfoot sagebrush (*Artemisia pedatifida*), in the nesting habitat were small (2–20 cm) in diameter and grew in a mat-like fashion (Parrish and others, 1993). Nest sites had lower forb density and more grass cover than sites where chicks were located (Parrish, 1988; Parrish and others, 1993).

In Colorado, brood rearing occurred in areas with more bare ground and less grass cover than areas used for nesting (Knopf and Rupert, 1999). Mean percentage of bare ground and grass cover on brood rearing areas was 15 and 84 percent, respectively, compared with 9 and 87 percent on nesting areas, respectively. Differences between the two areas were statistically significant. Cover of forbs, pricklypear, and cow dung were similar between the two areas.

Mountain Plovers often are associated with grasslands dominated by blue grama (*Bouteloua gracilis*) or buffalo-grass (*Bouteloua dactyloides*) (Bradbury, 1918; Laun, 1957; Finzel, 1964; Giezentanner, 1970; Graul, 1973, 1975; Graul and Webster, 1976; Wallis and Wershler, 1981; Parrish, 1988; Parrish and others, 1993; Childers and Dinsmore, 2008), but Knopf and Rupert (1999) reported that none of 147 nests in a Colorado study were in buffalograss. In Alberta, the dominant vegetation at nest sites was blue grama, Sandburg's bluegrass (*Poa secunda*), junegrass (*Koeleria macrantha*), and thread-leaf sedge (*Carex filifolia*) (Wershler and Wallis, 1987). In Montana, typical Mountain Plover habitat consisted of a plant community dominated by saltbush (*Atriplex* spp.) (Knowles, 1996). Plovers also were found in wild buckwheat (*Eriogonum* spp.) with 70 percent bare ground including glacial till and rocks. In Wyoming, the species nested in shortgrass pastures dominated by needleandthread (*Hesperostipa comata*), junegrass, and Sandburg's bluegrass (Laun, 1957).

Mountain Plovers occupy areas with moderately flat topography during the breeding season, but soil types vary among nesting areas. In western Wyoming, Beauvais and Smith (2003) reported a strong negative relationship between Mountain Plover presence and slope; at slopes greater than 8 percent, the probability of Mountain Plover presence was zero. In another Wyoming study, Parrish (1988) and Parrish and others (1993) reported that the species occurred on slopes of less than (<) 3 percent and on clay soil. Of 154 nests in Colorado, 65 percent were on slopes of <1 degree (slope of 1.7 percent), 25 percent on slopes of <2 degrees (slope of 3.5 percent), and one nest was on a slope of 5 degrees (slope of 8.7 percent) (presumably, the remaining percentages of nests were on slopes between 2 and 5 degrees) (Graul, 1975). Knowles and others (1982) observed Mountain Plovers in Montana on slopes of <12 percent. Ellison Manning and White (2001) observed plovers in Utah on slopes of less than or equal to 10 percent. In New Mexico, Mountain Plovers used grasslands with slopes of <1.5 degrees (slope of 2.6 percent) (Goguen, 2012). In Oklahoma, Mountain Plovers nested in bare, cultivated fields of <2 percent slope, and their distribution was closely tied to the presence of clay loam soils (McConnell and others, 2009). In Alberta, the species nested on flat mixed-grass pastures with well-drained sandy soil but avoided nesting in areas with poorly drained soil and rolling hills (Wershler and Wallis, 1987). In the northern Great Plains, Mountain Plovers used heavily grazed areas with aridic ustoll soils (Kantrud and Kologiski, 1982). In Montana, the soils within an area designated as critical habitat for Mountain Plovers consisted of hardpan clay and bentonite (Childers and Dinsmore, 2008). Within cultivated fields in Colorado, Kansas, Oklahoma, and Wyoming, soil near nests contained an average of 55 percent sand and 43 percent silt and clay, whereas soil of cultivated fields unoccupied by plovers contained an average of 80 percent sand and 20 percent silt and clay (Shackford and Leslie, 1994).

Prairie dog towns are an important component of Mountain Plover habitat (Childers and Dinsmore, 2008; Augustine and Skagen, 2014). Prairie dog towns may provide greater food resources and more available prey for Mountain Plovers than areas outside of towns (Olson, 1985). In mixed-grass prairies and other areas where vegetation is otherwise too tall, thick, or shrubby, prairie dog towns that are moderately to heavily grazed by livestock provide the mixture of short-statured grass and bare ground suitable for Mountain Plovers (Knowles and Knowles, 1984; Olson, 1984; Olson and Edge, 1985; Leachman and Osmundson, 1990). Prairie dogs within pastures with shrubs can control sagebrush (*Artemisia* spp.) growth and provide suitable nesting habitat (Olson, 1984; Olson and Edge, 1985).

In Montana, Mountain Plovers rarely were observed outside of prairie dog towns, and towns <10 hectares (ha) were considered marginal habitat (Knowles and others, 1982; Olson, 1984). The species used active prairie dog towns in upland areas characterized by heavy grazing and sparse shrub cover (Knowles and others, 1982). Nest sites on prairie

dog towns in Montana had lower mean cover of plants, bare ground, grass, and big sagebrush (*Artemisia tridentata*), and lower vegetation height and density of plains pricklypear than on adjacent areas outside of prairie dog towns (Olson, 1984). Compared to random sites within prairie dog towns, Mountain Plover nest sites in prairie dog towns were characterized by shorter vegetation; lower cover of total plants, grass, and pricklypear; higher cover of bare ground, fringed sagewort (*Artemisia frigida*), and big sagebrush; and higher forb density than at random sites within prairie dog towns (Olson 1984; Olson and Edge, 1985). In another Montana study, the densities of Mountain Plovers were much higher on prairie dog towns than on grasslands without towns (Childers and Dinsmore, 2008). In Colorado shortgrass prairies, prairie dog towns within moderately grazed pastures provided suitable habitat for Mountain Plovers, whereas grasslands without prairie dog towns did not (Augustine and Derner, 2012). The species was most common in burned areas and grasslands that contained active prairie dog towns, was less common in shortgrass prairies with inactive prairie dog towns, and was not found on unburned grasslands with no prairie dog towns (Augustine, 2011). In another Colorado study, Tipton and others (2008) reported that Mountain Plover occupancy was higher in prairie dog towns than in grasslands outside prairie dog towns or agricultural fields.

Prairie dog towns occupied by Mountain Plovers often are associated with stock ponds (Knowles and others, 1982; Olson and Edge, 1985). In Colorado, Knopf and Rupert (1999) examined the presence of Mountain Plovers around stock tanks used by cattle and stock tanks not used by cattle. Plovers were located near 11 of the 28 tanks where cattle were present and were absent at all 28 tanks without cattle. Plovers probably were attracted either to cattle presence or recent site disturbance created by cattle. Cattle-watering tanks were generally devoid of vegetation within 20 meters (m) around the tank.

Mountain Plovers frequently nest in cultivated fields. In Nebraska, Mountain Plovers nested almost exclusively in agricultural fields (98 percent of 278 nests) (Bly and others, 2008). Similarly, 90 percent of known Mountain Plover breeding populations in Oklahoma occurred in agricultural fields (McConnell and others, 2009). In Colorado, the species nested in cultivated fields, especially if native prairie was nearby; in Wyoming, the species nested in plowed river-bottom fields (Shackford, 1996). Of 52 nests within cultivated fields in Colorado, Kansas, Oklahoma, and Wyoming, 26 were located on bare or fallow ground, 13 were in fields of growing wheat (*Triticum* spp.), 7 were in fields of milo (*Sorghum* spp.), 4 were in fields of forbs or forb stubble, and 2 were in fields of sprouting corn (*Zea mays*) (Shackford and others, 1999). Mountain Plovers continued to nest in cropland as the crop increased in height to about 35 cm, including one nest that remained active until vegetation was 38 cm high (Shackford and Leslie, 1994).

Cultivated land also may be used by adults with broods (Shackford, 1987, 1991, 1996; Shackford and Leslie, 1994; Knopf, 1996; Knopf and Rupert, 1999; Shackford and others,

1999; Dreitz and others, 2005; Bly and others, 2008). In Colorado, Knopf and Rupert (1999) determined that adults with broods moved from cultivated land to grazed shortgrass prairie when vegetation in the field reached 20 cm tall.

Area Requirements and Landscape Associations

In Colorado, three male Mountain Plovers defended territories averaging 16 ha, and the minimum area needed to raise a brood was at least 28 ha (Graul, 1973; Knopf and Rupert, 1996). Areas used by different broods often overlapped (Graul, 1973; Knopf and Rupert, 1996). In another Colorado study, Dreitz and others (2005) reported average home-range estimates for adults with broods in three landscapes: 146.1 ha for 12 broods in grasslands, 131.6 ha for 13 broods in agricultural fields, and 243.3 ha for 10 broods in prairie dog towns. The larger home-range estimate for prairie dog towns compared to the other two habitats was attributed to two broods that were raised on a prairie dog complex rather than in isolated towns. Movements (that is, the distance moved by adults and their brood between two consecutive locations collected at 1- or 2-day intervals) were similar across habitats. Adult plovers that nested in grasslands and adults that nested in prairie dog towns did not move their broods to different habitats but rather stayed in the habitat in which they nested. Four adults that nested in agricultural fields remained with their broods in those fields, four adults moved their broods to grasslands, and five adults moved back and forth between these two habitats. In Colorado, Kansas, and Oklahoma, plovers were attracted to large (greater than 30 ha), cultivated fields (Shackford and Leslie, 1994).

The size of prairie dog towns is an important factor that affects Mountain Plover presence and abundance. In New Mexico, Goguen (2012) determined that the probability of Mountain Plovers occupying a prairie dog town increased with increasing area, and in Montana, Dinsmore and others (2005) concluded that the population trend of Mountain Plovers closely matched the trend in the area occupied by black-tailed prairie dogs (*Cynomys ludovicianus*). Mountain Plovers in Montana occurred at highest densities on towns that were 6–50 ha and were less abundant on smaller towns (Knowles and others, 1982; Olson, 1984; Olson-Edge and Edge, 1987). The average size of towns used by Mountain Plovers in north-central Montana was 57.5 ha (Knowles and Knowles, 1984).

Tipton and others (2008) examined the relationship between Mountain Plover occupancy and the amount of prairie dog town, grassland, and dryland agriculture at three spatial scales (a 25-ha plot and its surrounding 203.48 ha; 1,031.35 ha; or 2,064.30 ha). The sampling frame consisted of 500×500-m (25-ha) grid cells of each habitat. Mountain Plover occupancy was positively correlated with increasing amounts of prairie dog town in the immediate surrounding landscape, regardless of plot type (prairie dog town, grassland, or agriculture). The amount of grassland at the largest landscape scale positively

influenced Mountain Plover occupancy in prairie dog towns and agricultural plots, but negatively influenced plover occupancy in grassland plots. The authors surmised that the plot-specific response may reflect Mountain Plover use of one habitat type for nesting and another habitat type for brood rearing.

In Colorado, anthropogenic edges (defined as fence lines, roads, and perimeters of agricultural fields) had no effect on nest success of Mountain Plovers (Mettenbrink and others, 2006). Nest success relative to the distance from the nearest edge was similar between rangeland and cropland, and year-to-year differences were insignificant. In Oklahoma, McConnell and others (2009) found no evidence to indicate that Mountain Plovers were influenced by roads. The 34 plover detections within 200 m of the observer surveying from the road were similar to the 38 plover detections beyond 400 m; no Mountain Plovers were encountered during additional searches in interiors of roadless sections.

Brood Parasitism by Cowbirds and Other Species

No studies have documented brood parasitism by Brown-headed Cowbirds (*Molothrus ater*) (Shaffer and others, 2019). Mountain Plovers are not suitable hosts for Brown-headed Cowbirds because young plovers are precocial and nidifugous. Jojola-Elverum and Giesen (2000) reported a Killdeer (*Charadrius vociferus*) incubating a nest with three Killdeer eggs and three Mountain Plover eggs.

Breeding-Season Phenology and Site Fidelity

Generally, Mountain Plovers arrive on the breeding grounds from mid-March to mid-April and depart for the wintering grounds from early August to late October (Graul, 1973, 1975; Wallis and Wershler, 1981; Olson, 1984; Leachman and Osmundson, 1990; Knopf, 1996; Knopf and Wunder, 2006; Bly and others, 2008). In Colorado, the peak breeding period is mid-April to mid-July (Ball, 1996).

If the first nest fails before June, the female may attempt to renest (Knopf and Wunder, 2006). Multiple nesting attempts (that is, the male incubates a first clutch while the female incubates a second clutch simultaneously) have been reported (Graul, 1973; Knopf, 1996). Mountain Plovers exhibit fidelity to nest sites used the previous year (Graul, 1973, 1975; Skrade and Dinsmore, 2010).

Species' Response to Management

Burning can benefit Mountain Plovers when used to maintain areas of shorter grass within mixed-grass prairies (Wallis and Wershler, 1981; Knopf and Wunder, 2006). In

shortgrass prairies, dormant-season prescribed burns may provide suitable breeding habitat. In Colorado, Mountain Plover densities in shortgrass prairies that were burned during the dormant season were similar to Mountain Plover densities in shortgrass prairies with active prairie dog towns (Augustine, 2011). Densities in these two habitats were higher than in shortgrass prairies with inactive prairie dog towns, and no Mountain Plovers occurred on unburned grasslands that had no active prairie dog towns owing to an outbreak of epizootic plague. Augustine and Skagen (2014) determined that Mountain Plover densities were similar between grasslands burned during the preceding dormant season and active prairie dog towns, but nest survival was higher on active prairie dog towns than burned sites. Increased time since burning decreased plover densities; densities declined by 78 percent at burned sites between the first and second season post-burn growing season. In comparison, removal of prairie dog disturbance owing to sylvatic plague reduced plover density by 70 percent relative to active prairie dog colonies after 1 year.

Grazed grasslands, especially those containing prairie dog towns, are vital to Mountain Plovers (Ryder, 1980; Kantrud and Kologiski, 1982; Knowles and others, 1982; Wershler and Wallis, 1987; Bock and others, 1993; Knowles, 1996; Dinsmore and others, 2005; Childers and Dinsmore, 2008). Grazing activities that maintain short vegetation and low litter attract Mountain Plovers; the species has been reported inhabiting areas around stock tanks and increasing in numbers where sheep herds and American bison (*Bison bison*) are pastured (Knowles, 1996). In Alberta, heavy grazing in summer or late winter improved habitat for Mountain Plovers by providing short-statured grass in mixed-grass areas (Wallis and Wershler, 1981; Wershler and Wallis, 1987). In Colorado, shortgrass pastures grazed heavily in summer were used by Mountain Plovers for foraging and nesting (Giezen-tanner, 1970). However, Mountain Plovers may be excluded by extreme or long-term overgrazing (Laun, 1957; Wallis and Wershler, 1981). Within Montana shrub grasslands (Olson and Edge, 1985) and Colorado shortgrass prairies (Augustine and Derner, 2012), intensive livestock grazing alone (1.2 animal unit months per ha, which was double the usual stocking rate), without prairie dog towns or burning, did not provide suitable habitat.

Cultivated land may provide suitable habitat in areas where prairie dog towns in grasslands are unavailable. In Oklahoma, Mountain Plovers preferred plowed fields over shortgrass prairies, unless prairie dog towns were present on the prairies (Shackford, 1991, 1996). In Colorado in April, the number of Mountain Plovers within paired plots at a native-prairie and cultivated-field interface did not differ. Mountain Plovers preferred a cultivated field over shortgrass prairie after April (Knopf and Rupert, 1999). After the cultivated field had been planted and chemically treated for weed control in early May, detections of Mountain Plovers were significantly higher on the cultivated field than on the shortgrass prairie. Although a few nests were found, Mountain Plovers appeared to be mainly foraging in the prairie and were nesting primarily in

the field. However, planting and weed control operations probably destroyed nests, because Mountain Plovers began courtship activities for a second time. Broods moved from prairie to fields, as did adults that were unsuccessful breeders.

Mountain Plovers are present in saltbush-dominated habitats in central Montana; Mountain Plovers abandoned an area after the seeding of saltbush habitats to crested wheatgrass (*Agropyron cristatum*) and the development of weedy mudflats owing to damming (Knowles, 1996).

In Montana, Dinsmore (2013) reported a strong negative effect of a pyrethroid insecticide on nest survival of Mountain Plovers. The insecticide contained 0.05 percent deltamethrin and was used to control fleas on prairie dogs to help limit the spread of plague, a flea-borne disease caused by the bacterium *Yersinia pestis*. Although there was no evidence of direct toxicity to nesting Mountain Plovers, Dinsmore (2013) indicated that the insecticide caused lower insect availability for the plovers, resulting in more time off nests or a switch to a less desirable insect prey, which in turn lowered nest survival.

Some oil and gas extraction activities may be compatible with Mountain Plover habitat needs. In Utah, disturbed areas around oil well pads created open habitat with high amounts of bare ground suitable for Mountain Plovers (Day, 1994). Mountain Plovers in southeastern Wyoming did not seem to be disturbed by nearby mining activity (Parrish, 1988).

Management Recommendations from the Literature

Disturbances, such as those created by livestock grazing, burning, or prairie dog activity, provide critical breeding habitat for Mountain Plovers (Finzel, 1964; Wallis and Wershler, 1981; Knowles and Knowles, 1984; Knopf and Wunder, 2006; Augustine and Derner, 2012; Augustine and Skagen, 2014). Because some plovers return to the same breeding sites year after year, several authors emphasized the importance of protecting areas traditionally used by Mountain Plovers, especially within large and active prairie dog towns (Graul, 1973, 1975; Knowles and others, 1982; Olson, 1984; Olson and Edge, 1985; Olson-Edge and Edge, 1987; Dinsmore and others, 2005; Goguen, 2012). Livestock grazing in these areas should be encouraged, because prairie dog towns often are associated with grazed areas (Knowles and others, 1982; Olson and Edge, 1985). Maintaining complexes of prairie dog towns will be more advantageous for successful brood rearing than maintaining isolated prairie dog towns (Dreitz and others, 2005). Management that mitigates or prevents plague outbreaks on large prairie dog towns will help maintain plover habitat (Augustine and Skagen, 2014).

Wershler (1987, 1991) recommended protecting large, unfragmented native grasslands from tillage and from seeding to exotic grasses. Knopf and Rupert (1999) highlighted the importance of using native seed mixtures on Conservation Reserve Program areas. In Colorado, shortgrass prairies

should be managed to maintain vegetation <25 cm in height (Ball, 1996).

Prescribed burns can improve Mountain Plover habitat in native grasslands (Wallis and Wershler, 1981; Eldridge, 1992; Knopf and Wunder, 2006). Burning, however, provides suitable habitat for only about 1 year; in Colorado, plover densities declined by 78 percent between the first and second post-burn growing season (Augustine and Skagen, 2014). Timing of burns and consideration of geographical location and annual precipitation are important. In Alberta, burns conducted in late summer or early fall improved the vegetation structure and composition that is associated with Mountain Plover habitat (Wershler, 1991). In Colorado, Knopf and Rupert (1999) found that Mountain Plovers were attracted to grasslands that had been burned the previous winter or spring. In another Colorado study, Augustine and Skagen (2014) reported that plover density did not differ between sites burned in fall compared to late winter or with burn size (11- to 500-ha burns), suggesting that managers have flexibility in selecting the timing and size of burns. Burning also may help maintain Mountain Plover habitat without adversely affecting forage availability for cattle; in Colorado shortgrass prairies during years with average or above-average precipitation, burning <50 percent of a pasture provided plover habitat (Augustine and Milchunas, 2009; Augustine and others, 2010).

A management approach that combines prairie dog conservation with the strategic use of prescribed burning can be beneficial to Mountain Plovers (Augustine and Skagen, 2014). Conducting burns near prairie dog towns that were recently affected by an outbreak of epizootic plague may help the towns to recover, as well as increase nest survival rate (Augustine, 2011; Augustine and Derner, 2012). In addition, prescribed burning can be used to create nesting habitat in landscapes where disturbances such as prairie dog towns are limited in distribution and size (Augustine and Skagen, 2014). Vegetation structure within prairie dog towns was more stable in space and time than within burned areas; hence, Augustine and Derner (2012) advised that the importance of maintaining active prairie dog towns should not be overlooked.

Suitable Mountain Plover habitat can be created or maintained by varying livestock grazing intensities, ranging from heavily grazed to ungrazed, which may simulate historic grazing pressure by American bison (Wallis and Wershler, 1981). For example, in shortgrass or mixed-grass pastures, grazing at moderate-to-heavy intensities will improve Mountain Plover habitat (Knowles and others, 1982; Eldridge, 1992). Grazing at heavy intensities, however, should occur in summer or late winter (Wallis and Wershler, 1981; Wershler, 1987). Wallis and Wershler (1981) recommended that long-term overgrazing should be avoided in Alberta and Saskatchewan because it may exclude Mountain Plovers. To enhance nesting habitat in Colorado, Knopf and Rupert (1999) suggested moving livestock into native pastures by early May to provide intensive, long-term grazing on grasslands adjacent to cultivated fields.

Cultivated landscapes can provide suitable habitat, but changes in cropland type may alter the suitability of fields

from one year to the next and hence represent population sinks (McConnell and others, 2009). Furthermore, cultivated fields in the northern part of the species' breeding range differ from fields in the southern part with regard to crop type, timing of cultivation, number of disturbances, and length of growing season (Shackford and others, 1999). These factors influence suitability and nest productivity of these habitats for Mountain Plovers, and more research is needed to evaluate the effectiveness of different agricultural practices in conserving populations of plovers. Some general management recommendations can be made. Knopf and Wunder (2006) recommended avoiding plowing operations of nesting areas during the breeding season. Shackford and others (1999) reported that 67 percent of 46 nests were lost to plowing operations. To minimize length of disturbance, Knopf and Rupert (1999) recommended that preparation of cropland fields for planting should be done right before planting rather than a month in advance. To control weeds between May 1 and July 15, Knopf and Rupert (1999) suggested that chemical control, rather than physical types of disturbance, should be used on fields.

Although oil and gas extraction activities may be compatible with Mountain Plover habitat needs, Ball (1996) recommended that extraction-related activities be restricted near Mountain Plover habitat during the peak breeding season (April–July), that production facilities be located in ways that minimize disturbance to plover habitat, and that travel be restricted to existing roads.

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Table E1. Measured values of vegetation structure and composition in Mountain Plover (*Charadrius montanus*) breeding habitat by study. The parenthetical descriptors following authorship and year in the “Study” column indicate that the vegetation measurements were taken in locations or under conditions specified in the descriptor; no descriptor implies that measurements were taken within the general study area.

[cm, centimeter; %, percent; --, no data; <, less than; ≥, greater than or equal to; ≤, less than or equal to]

Study	State or province	Habitat	Management practice or treatment	Vegetation height (cm)	Vegetation height-density (cm)	Grass cover (%)	Forb cover (%)	Shrub cover (%)	Bare ground cover (%)	Litter cover (%)	Litter depth (cm)
Augustine and Derner, 2012 (nests)	Colorado	Shortgrass prairie	Multiple	3.9	--	--	--	--	35.9	--	--
Augustine and Derner, 2012 (foraging locations)	Colorado	Shortgrass prairie	Multiple	3.5	--	--	--	--	34.1	--	--
Ellison Manning and White, 2001 (nests)	Utah	Shrubsteppe	--	23.1	--	31.4	13.6	54.9	--	--	--
Goguen, 2012 (prairie dog towns)	New Mexico	Shortgrass prairie	Grazed	2.3	--	28.5	--	9.2	61.5	--	--
Goguen, 2012 (nests)	New Mexico	Shortgrass prairie	Grazed	1.6	--	29.5	--	3.6	66.1	--	--
Graul, 1973, 1975 (nests)	Colorado	Shortgrass prairie	Grazed	<10	--	--	--	--	--	--	--
Knopf, 1996 (nests)	Colorado	Shortgrass prairie	Multiple	--	--	68	--	--	32	--	--
Knopf and Miller, 1994	Colorado	Shortgrass prairie	Multiple	--	--	--	--	--	≥30	--	--
Knopf and Rupert, 1999 (nests)	Colorado	Shortgrass prairie	Grazed	--	--	87	--	--	9	--	--
Knopf and Rupert, 1999 (broods)	Colorado	Shortgrass prairie	Grazed	--	--	84	--	--	15	--	--
Knowles, 1996	Montana	Shortgrass prairie	Grazed	--	--	--	--	--	70	--	--
Olson, 1984; Olson and Edge, 1985 ^a (nests)	Montana	Shortgrass prairie	Grazed	4.4	--	14.5	--	--	26.9 ^b	31.8	4.4
Olson, 1984; Olson and Edge, 1985 ^a (prairie dog towns)	Montana	Shortgrass prairie	Grazed	6.4	--	14.5	--	--	42.5 ^b	24.7	6.4
Parrish and others, 1993 (nests)	Wyoming	Reclaimed short-grass prairie	Grazed	≤9	--	13.9	2.1	10.4	71.8	1.9	--
Shackford, 1991 (nests)	Oklahoma	Cropland	--	20	--	--	--	--	--	--	--
Shackford and Leslie, 1994 (nests)	Colorado, Kansas, Oklahoma	Cropland	--	<38	--	--	--	--	--	--	--
Wershler and Wallis, 1987 (nests)	Alberta	Mixed-grass prairie	Burned, grazed	--	--	--	--	--	45–50	--	--
Wershler and Wallis, 1987 (nests)	Alberta	Mixed-grass prairie	Unburned, grazed	--	--	--	--	--	15–25	--	--

^aThe sum of the percentages is greater than 100%, based on methods described by the author.

^bErosion pavement cover.

The Effects of Management Practices on Grassland Birds—Upland Sandpiper (*Bartramia longicauda*)

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Chapter F of

The Effects of Management Practices on Grassland Birds

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Conversion Factors

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		
square meter (m ²)	0.0002471	acre
hectare (ha)	2.471	acre
square meter (m ²)	10.76	square foot (ft ²)
hectare (ha)	0.003861	square mile (mi ²)
Mass		
kilogram (kg)	2.205	pound (lb)
Luminance		
candle per square meter (cd/m ²)	0.8361	candles per square yard (cd/yd ²)

Abbreviations

AUM	animal unit month
BBS	Breeding Bird Survey
CRP	Conservation Reserve Program
DNC	dense nesting cover
spp.	species (applies to two or more species within the genus)

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The Effects of Management Practices on Grassland Birds—Upland Sandpiper (*Bartramia longicauda*)

By Jill A. Shaffer,¹ Lawrence D. Igl,¹ Douglas H. Johnson,¹ Meghan F. Dinkins,^{1,2} Christopher M. Goldade,^{1,3} Barry D. Parkin,¹ and Betty R. Euliss¹

Capsule Statement

The key to Upland Sandpiper (*Bartramia longicauda*) management is providing grasslands of various heights with few shrubs. In general, Upland Sandpipers forage within short vegetation and nest and rear broods within taller vegetation. Upland Sandpipers have been reported to use habitats with less than (<) 93 centimeters (cm) vegetation height, 5–75 cm visual obstruction reading, greater than or equal to (\geq) 33 percent grass cover, less than or equal to (\leq) 50 percent forb cover, \leq 13 percent shrub cover, 3–12 percent bare ground, 11–30 percent litter cover, and <13 cm litter depth. The descriptions of key vegetation characteristics are provided in table F1 (after the “References” section). Vernacular and scientific names of plants and animals follow the Integrated Taxonomic Information System (<https://www.itis.gov>).

Breeding Range

Upland Sandpipers breed from northeastern British Columbia to southwestern Ontario; south to northeastern Oregon, Idaho, Wyoming, Colorado, and Oklahoma; east to Virginia; and north to New Brunswick (National Geographic Society, 2011). The relative densities of Upland Sandpipers in the United States and southern Canada, based on North American Breeding Bird Survey data (Sauer and others, 2014), are shown in figure F1 (not all geographic places mentioned in report are shown on figure). The species also breeds in southeastern Alaska, northern British Columbia, and southwestern Yukon Territory.



Upland Sandpiper. Illustration by Christopher M. Goldade, U.S. Geological Survey.

Suitable Habitat

In general, Upland Sandpipers use areas with moderate grass cover, low-to-moderate forb cover, moderate-to-high litter cover, and sparse woody cover and bare ground (Buss and Hawkins, 1939; Rotenberry and Wiens, 1980; Renken, 1983; Skinner and others, 1984; Buhnerkempe and Westemeier, 1988; Sample, 1989; Kantrud and Higgins, 1992; Hull and others, 1996). Display perches, such as fence posts, may be important components of suitable habitat (Bent, 1962; Salt and

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2 The Effects of Management Practices on Grassland Birds—Upland Sandpiper (*Bartramia longicauda*)

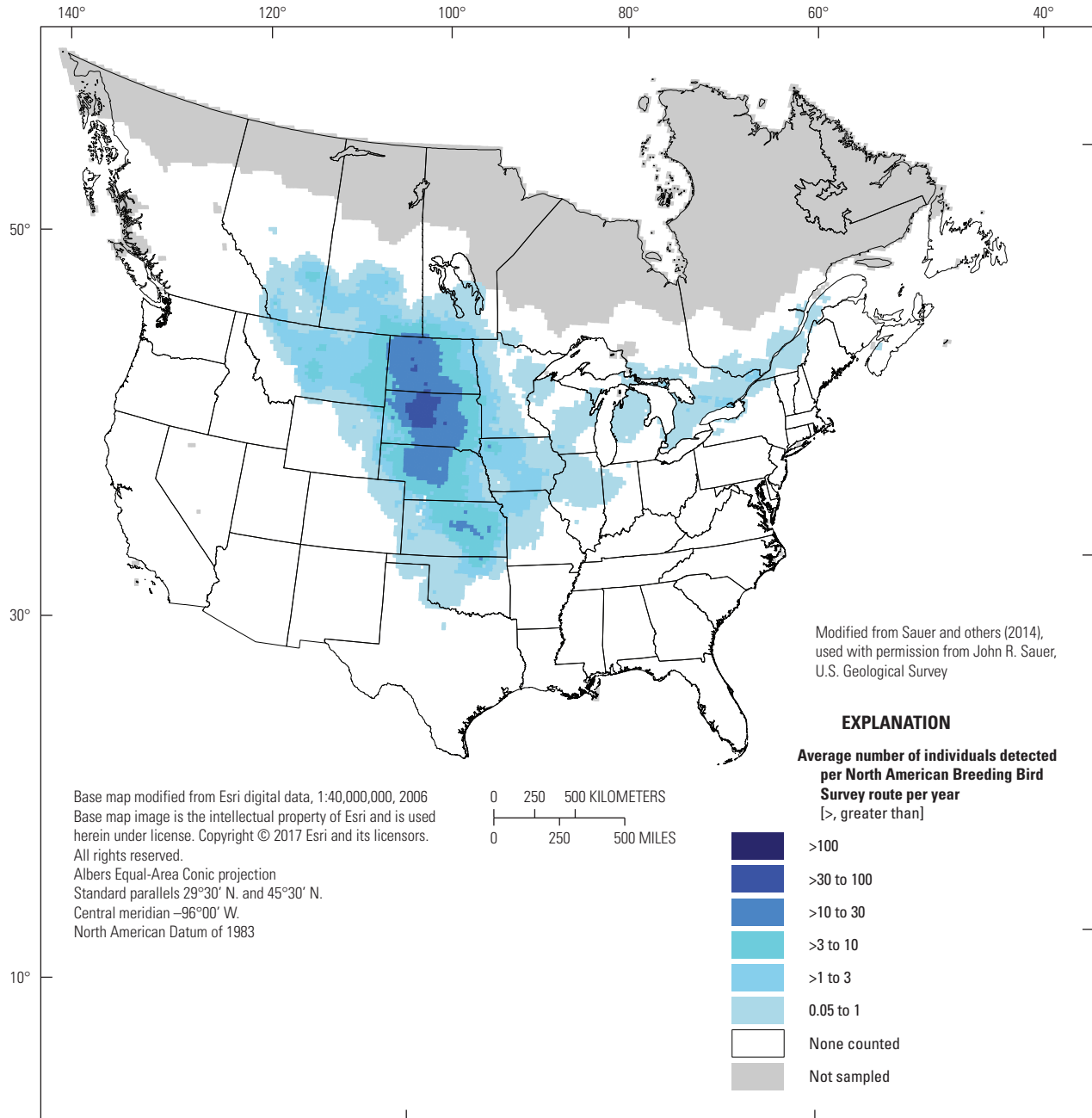


Figure F1. Breeding distribution of the Upland Sandpiper (*Bartramia longicauda*) in the United States and southern Canada, based on North American Breeding Bird Survey (BBS) data, 2008–12. The BBS abundance map provides only an approximation of breeding range edges.

Salt, 1976; White, 1983; Snyder and others, 1987). Upland Sandpipers breed in a variety of habitats, including shortgrass, mixed-grass, and tallgrass prairies that are idle, burned, hayed, or grazed (Bent, 1962; Goering, 1964; Stewart, 1975; Salt and Salt, 1976; Johnsgard, 1980; White, 1980; Skinner and others, 1984; Kantrud and Higgins, 1992; King and Savidge, 1995; Houston and others, 2011; Garvey and others, 2013), as well as wet meadows and jack pine barrens (Dorio and Grewe, 1979; Sample, 1989; Faanes and Lingle, 1995; Kim and others, 2008; Korte, 2013; Corace and others, 2016). The species

also uses tame grasslands and grassed waterways (Buss and Hawkins, 1939; Goering, 1964; Oetting and Cassel, 1971; Ailes and Toepfer, 1977; Sample, 1989; Bolster, 1990; Bryan and Best, 1991; Kantrud and Higgins, 1992). Planted cover, such as Conservation Reserve Program (CRP) fields, dense nesting cover (DNC), and Waterfowl Production Areas may provide suitable habitat (Renken and Dinsmore, 1987; Luttswager and Higgins, 1992; Johnson and Schwartz, 1993a, 1993b; Faanes and Lingle, 1995; Johnson and Igl, 1995; King and Savidge, 1995; Hull and others, 1996; Roth and others,

2005). Upland Sandpipers inhabit cropland, such as wheat (*Triticum* species [spp.]) stubble, fallow fields, grains, and rowcrops (Bates, 1907; Bent, 1962; Oetting and Cassel, 1971; Higgins, 1975; Kirsch and Higgins, 1976; Ailes and Toepfer, 1977; Dorio and Grewe, 1979; Ducey and Miller, 1980; Buhnerkempe and Westemeier, 1988; Bolster, 1990; Hultquist and Best, 2001). The species uses open fields at airports (White, 1980; Snyder and others, 1987).

In North Dakota, South Dakota, Montana, Wyoming, Colorado, and Nebraska, densities of Upland Sandpipers were highest in areas with moderately grazed typic ustoll soils (Kantrud and Kologiski, 1982). In North Dakota mixed-grass prairies, Upland Sandpipers were associated with silty range and thin upland range sites; these sites were characterized by thin topsoil, loamy soil, 1–25-percent slope, grassy cover, low shrub coverage, and moderate-to-high litter coverage (Messmer, 1990). In the same area, Sedivec (1994) found Upland Sandpipers more frequently on overflow range sites (areas receiving more than normal soil moisture because of runoff from higher land or from flooding) than silty range sites. In Kansas, Upland Sandpipers appeared to prefer clay upland range sites and to avoid loamy upland range sites and limestone breaks (Bowen, 1976). The species used claypan and shallow range sites in proportion to their abundance; relative abundance of clay upland was a good predictor of use by Upland Sandpipers. In Wisconsin, Upland Sandpipers were found on areas of Clyde silt loam and peat but not on areas of Miami silt loam, possibly because these areas supported trees (Buss and Hawkins, 1939). In Michigan, soil types of areas where Upland Sandpipers had reliably bred for at least the previous 25 years were glacial outwash, coarse glacial till, and lacustrine sand and gravel (Korte, 2013; Corace and others, 2016).

Upland Sandpipers use native and tame grasslands but show no clear preference for either grassland type. In a study encompassing grasslands throughout the Great Plains, areas considered to be the best habitat for Upland Sandpipers were dominated by wheatgrass (formerly *Agropyron* spp.) and Kentucky bluegrass (*Poa pratensis*), followed by green needlegrass (*Nassella viridula*), buffalograss (*Bouteloua dactyloides*), western snowberry (*Symphoricarpos occidentalis*), and slimspike three-awn (*Aristida longespica*) (Kantrud and Kologiski, 1982). Dominant vegetation at nest sites in Manitoba, North Dakota, Montana, and South Dakota were Kentucky bluegrass, smooth brome (*Bromus inermis*), needle and thread (*Hesperostipa comata*), and quackgrass (*Elymus repens*), although most nests were located within native mixed-grass prairies. Within those same States and Province, Upland Sandpipers readily nested in stands of tame grasses; forbs and shrubs were dominant at very few nests (Kantrud and Higgins, 1992). In Saskatchewan, Upland Sandpipers were present in low abundance in native mixed-grass prairies and in tame grasslands dominated by crested wheatgrass (*Agropyron cristatum*) (Sutter and Brigham, 1998). In Manitoba, numbers of Upland Sandpipers were positively correlated with presence of native vegetation and negatively correlated with presence of

tame vegetation (Wilson and Belcher, 1989). In northern North Dakota mixed-grass prairies, Upland Sandpiper occurrence was not related to coverage of Kentucky bluegrass, smooth brome and quackgrass, native grasses and forbs, or tame legumes (Grant and others, 2004). Within ungrazed grasslands in North Dakota and South Dakota, 93 percent of 41 nests were in either native or tame grasses located in idle fields or in rights-of-way (Higgins and others, 1969). Nests were primarily placed in little bluestem (*Schizachyrium scoparium*), needle and thread, porcupinegrass (*Hesperostipa spartea*), green needlegrass, junegrass (*Koeleria macrantha*), blue grama (*Bouteloua gracilis*), smooth brome, Kentucky bluegrass, quackgrass, and crested wheatgrass; one nest was in alfalfa (*Medicago sativa*) and brome (*Bromus* spp.). The three nests not found in idle fields or in rights-of-way were in pastures. Within grazed grasslands in North Dakota, vegetation within 1 meter (m) of nests consisted of native grasses (needle and thread, green needlegrass, western wheatgrass [*Pascopyrum smithii*]) and small amounts of tame grasses such as Kentucky bluegrass and smooth brome (Bowen and Kruse, 1993). In South Dakota, Upland Sandpipers nested only in native prairie; 97 percent of the 33 nests were in prairies classified as in good or excellent range condition (Kaiser, 1979). The species preferred to nest in mixed-grass or tallgrass prairies, although nests also were found in Kentucky bluegrass. Nest success did not differ between nests in matted and upright residual vegetation or among nests in tallgrass prairies, in the transition zone between mixed-grass and tallgrass, or in mixed-grass prairies. In northwestern Minnesota, 91 percent of 22 nests were in native grasses (little bluestem, junegrass, and muhly [*Muhlenbergia* spp.]); the other two nests were in alfalfa (Lindmeier, 1960). Likewise, in southeastern Wisconsin, nests were placed in tame and native vegetation (junegrass, reed canary grass [*Phalaris arundinacea*], quackgrass, and timothy [*Phleum pratense*]), and two nests were found in legumes (alfalfa and sweetclover [*Melilotus* spp.]) (Buss and Hawkins, 1939). Nests initiated earlier in the nesting season were in pastures, whereas nests initiated later in the nesting season were in idle native grasslands. In central Minnesota and central Wisconsin, Upland Sandpipers nested in tame vegetation; study areas, however, may have contained little or no native vegetation (Ailes, 1976; Dorio, 1977). Dorio (1977) noted that the species nested in smooth brome, quackgrass, yarrow (*Achillea millefolium*), goldenrod (*Solidago* spp.), and in wet meadows (annually mowed sedge [*Carex* spp.], timothy, and Canada bluegrass [*Poa compressa*]). In Illinois, Upland Sandpipers preferred stands of Kentucky bluegrass and other tame grass species as opposed to tallgrass prairie and preferred older (greater than [$>$] 5 years old) plantings of tame grasses and forbs (Birkenholz, 1973; Buhnerkempe and Westemeier, 1988). Musselman (1935) noted that Upland Sandpipers in Missouri and Illinois nested within an idle clover (*Trifolium* spp.) field and an idle grassland. In Kansas, Upland Sandpipers nested in tallgrass prairies as well as in tame grasses (Goering, 1964). Nests were within dense stands of ungrazed big bluestem (*Andropogon gerardii*) and little bluestem, in spring-burned three-awn, in

heavily grazed smooth brome, and in clumps of yellow sweet-clover (*Melilotus officinalis*) within weedy, ungrazed brome. In Nebraska, King and Savidge (1995) observed Upland Sandpipers in CRP fields seeded with warm-season grasses and native tallgrass prairies.

Upland Sandpipers prefer grasslands with minimal coverage of woody vegetation. In North Dakota mixed-grass prairies, Upland Sandpipers were present in grasslands with a lower percentage cover of shrubs >1 m tall than in unoccupied grasslands (Grant and others, 2004). Occurrence was not related to the percentage cover of shrubs <1 m tall. In mixed-grass prairies in South Dakota, Upland Sandpipers generally were more abundant in early seral stage areas than in late seral stage areas; seral stage was defined by percentage cover of three grass species and their frequency of occurrence (Fritcher and others, 2004). In Wisconsin, Upland Sandpipers avoided sites with woody vegetation (Sample, 1989). Sandpiper density was highest in areas with a medium density of prostrate residual vegetation; medium density was defined as <3 cm deep with >50 percent coverage. Density of Upland Sandpipers was negatively correlated with total percentage of woody cover, total number of dead stems, maximum vegetation height, and vegetation height-density. In another Wisconsin study, Upland Sandpipers were more numerous in nonforested areas with level terrain and with large agricultural fields, preferably hay, oats (*Avena* spp.), or pasture, but not corn (*Zea mays*) (White 1980, 1983).

In a multi-State study, the abundance of Upland Sandpipers was positively correlated with the total number of vertical vegetation hits and negatively correlated with the percentage of bare ground (Rotenberry and Wiens, 1980). Within DNC fields in North Dakota, the species used plots with less grass, less forb coverage, shorter and less dense vegetation, and a thinner litter layer than unused plots (Renken, 1983; Renken and Dinsmore, 1987). In North Dakota mixed-grass prairies, Upland Sandpipers were present in grasslands with lower maximum vegetation height and lower percentage cover of live vegetation than in unoccupied grasslands (Grant and others, 2004). Occurrence was not related to litter depth or year. In Colorado, the species used lightly to moderately grazed pastures and bare ground in proportion to the availability of these habitat features (Bolster, 1990). Upland Sandpipers preferred medium-height vegetation in shortgrass prairies and preferred grass-like vegetation more than tangled vegetation. Before incubation, the species used heavily grazed fields more often and weedy fields less often than expected. In North Dakota tallgrass prairies, Upland Sandpiper abundance was affected by the interaction between vegetation height-density (visual obstruction reading) and litter depth; abundance generally increased with variability in litter depth but decreased with high levels of litter depth and vegetation height-density (Ahlering and Merkord, 2016).

Vegetation structure, time of day, daily activities, and phase of nesting cycle influence habitat use (Dorio, 1977; Dorio and Grewe, 1979; Bolster, 1990). For example, in Colorado, Upland Sandpipers were encountered most frequently

on heavily grazed (average vegetation <10 cm tall) pastures and on cut and baled alfalfa fields during the morning; during the evening, the species preferred bare ground and small-grain fields in which vegetation was <27 cm tall (Bolster, 1990). In Wisconsin, territories included loafing, nesting, and feeding sites; suitable loafing areas were pastures with low-growing clumps, sparsely vegetated grassland, and open hayland (Buss and Hawkins, 1939). Loafing and feeding sites were near nesting sites and were shared by several sandpiper pairs. In Kansas, Upland Sandpipers used brood rearing sites characterized by short vegetation, low grass density, high bare ground coverage, and a mix of forbs, woody vegetation, and grass coverage (Mong, 2005).

Upland Sandpipers prefer to forage in short vegetation. Upland Sandpipers exhibited seasonal use of foraging habitats in Minnesota (Dorio, 1977; Dorio and Grewe, 1979). Upon first arriving on the breeding grounds in spring, Upland Sandpipers used plowed and seeded fields; in May, sedge-grass meadows were used until vegetation was 30 cm tall; in May and June, overgrazed pastures were used; and in late summer, mowed fields of red clover (*Trifolium pratense*) were used when vegetation was 2.5–15 cm tall. In Wisconsin, a few Upland Sandpipers were observed foraging in plowed and fallow (previously pasture, but plowed and left idle) fields and recently seeded corn fields, in which corn was 5–10 cm tall (Ailes, 1976; Ailes and Toepfer, 1977). Corn fields were no longer used by the species when the corn was >15 cm tall. Overall, however, idle fields, plowed fields, and cropland were used infrequently for foraging, and feeding occurred mostly in grazed pastures, followed by ungrazed pastures and hayfields (Ailes, 1976). In Nebraska, Upland Sandpipers foraged in pastures and corn fields before corn emerged or while corn was still short (Ducey and Miller, 1980). In another Nebraska study, Upland Sandpipers foraged in wheat stubble that contained grain (Bates, 1907). Graber and Graber (1963) suggested that open and idle fields and cropland were used for foraging in Illinois. In Minnesota, both young and adult birds preferred to feed in vegetation <10 cm tall (Dorio, 1977; Dorio and Grewe, 1979). Zimmerman (1993) surmised that the Upland Sandpiper was most abundant in annually burned grasslands in Kansas because the species used these areas as foraging habitat. In another Kansas study, Goering (1964) reported that Upland Sandpipers foraged in areas with short grass, especially burned pastures, upon their arrival in spring. In Indiana, the species foraged in mowed areas and in idle tallgrass prairies (Snyder and others, 1987).

Upland Sandpipers nest in a variety of habitats, ranging from idle prairies with dense, homogeneous vegetation to wet meadows, oldfields (idle or neglected arable lands that have naturally reverted back to perennial cover), pastures, hayland, cropland, tame vegetation, burned areas, and sandy areas with sparse vegetation (Musselman, 1935; Buss and Hawkins, 1939; Bent, 1962; Graber and Graber, 1963; Goering, 1964; Higgins, 1975; Ailes, 1976; Dorio, 1977; Dorio and Grewe, 1979; White, 1980; Snyder and others, 1987; Buhnerkempe and Westemeier, 1988; Colwell and Oring, 1990; Kantrud and

Higgins, 1992; Faanes and Lingle, 1995). Nests may be placed in depressions covered by grass arching over the top, in grass clumps, in dense vegetation, or at the base of forbs or shrubs (Buss and Hawkins, 1939; Lindmeier, 1960; Bent, 1962; Ailes, 1976; Kirsch and Higgins, 1976; Salt and Salt, 1976; Skinner and others, 1984). In Alberta, Saskatchewan, and Manitoba, Upland Sandpipers preferred nesting in native grasslands, either grazed or idled, more than in cropland, hayland, planted cover, wetland, or woodland habitats; however, daily nest-survival rates were not higher in the preferred habitat (Garvey and others, 2013). Of 41 nests located in the Missouri Coteau of North Dakota, 38 were in idle grasslands characterized by moderately tall grasses and abundant ground litter (Higgins and others, 1969). In another North Dakota study, Upland Sandpipers chose nest sites with less grass coverage than random locations (Wiens, 2007). In Wisconsin, choice of nesting sites changed as the season progressed; nests initiated early in the breeding season were located in pastures, whereas nests initiated later in the breeding season were in ungrazed prairies (Buss and Hawkins, 1939). In another Wisconsin study, 38 percent of 553 nests were in pastures, one-fifth of which were in burned pastures White (1983). An additional 28 percent of nests were in tallgrass prairies, 7 percent in hayfields, and the remainder in woody areas, cropland, wetlands, and idle fields adjacent to airport runways. In Colorado, nesting Upland Sandpipers appeared to prefer lightly grazed pastures (average vegetation 17–23 cm tall) and small-grain fields (vegetation <27 cm tall), and to use tall (>27 cm) alfalfa and small-grain fields (vegetation \geq 27 cm tall) less than expected (Bolster, 1990).

Vegetation height around Upland Sandpiper nests generally ranges from 10.2 to 63.5 cm (Lindmeier, 1960; Goering, 1964; Higgins and others, 1969; Ailes, 1976; Kaiser, 1979; Buhnerkempe and Westemeier, 1988; Eldridge, 1992). In Saskatchewan, Upland Sandpipers nested in tall, dense, homogeneous vegetation >15 cm tall (Colwell and Oring, 1990). In North Dakota, Upland Sandpipers most commonly nested in areas where grass accounted for \geq 50 percent canopy cover and forbs accounted for <50 percent canopy cover (Bowen and Kruse, 1993). Two other habitats used for nesting were those in which forbs accounted for \geq 50 percent canopy cover and grass for <50 percent canopy cover, or in which western snowberry with a grass understory accounted for <50 percent canopy cover. Habitats were avoided in which western snowberry with a grass understory accounted for \geq 50 percent canopy cover. In South Dakota, nest concealment in grazed prairie was measured from various angles: all nests had \geq 50 percent vertical concealment by residual and living vegetation, 33 percent of nests were concealed on all sides, 55 percent were concealed on two sides, and 12 percent had no side concealment (Kaiser, 1979). In northwestern Minnesota, vegetation height at nests measured within 10 days after the first egg was laid averaged 25.4 cm and consisted largely of residual vegetation (Lindmeier, 1960). Standing vegetation over Upland Sandpiper nests was fairly sparse, with an average light intensity of 222 candles per square meter (m^2) for

12 nests. In another study in northwestern Minnesota, mean vegetation measurements from 40 sampling points within four Upland Sandpiper territories were 79 cm vegetation height, 35 percent ground cover (coverage of live vegetation with a total height of \leq 10 cm), and 24 cm phanerophyte height (Niemi and Hanowski, 1983). Phanerophytes were defined as shrubs, forbs, or graminoids >40 cm tall and present each year. In Wisconsin, Upland Sandpipers did not initiate nests in vegetation >40 cm tall, although by the time the eggs hatched, vegetation was as tall as 70 cm (Ailes, 1976, 1980).

Sparse-to-moderate forb coverage may be an important component of suitable nesting habitat (Skinner, 1975; Renken, 1983; Buhnerkempe and Westemeier, 1988; Klute, 1994; Hull and others, 1996; Klute and others, 1997). In Minnesota, the predominant forbs within territories were bedstraw (*Galium* spp.), goldenrod, and clover (Niemi and Hanowski, 1983). In Greater Prairie-Chicken (*Tympanuchus cupido*) sanctuaries in Illinois, Upland Sandpipers preferred to nest in fields of seeded grasses that were being invaded by forbs (Buhnerkempe and Westemeier, 1988). In Missouri, Skinner (1974) found fewer Upland Sandpipers in fields that were predominantly forbs than in fields with scattered forbs or with no forbs. In Kansas tallgrass prairies, Upland Sandpipers were significantly more abundant in pastures than in CRP grasslands; grazed pastures had significantly greater coverage of total vegetation, live vegetation, grasses, and forbs than did CRP grasslands (Klute and others, 1997). In Kansas CRP fields, Upland Sandpipers were present in fields described as having a medium frequency of occurrence of forbs (Hull and others, 1996). In Oklahoma, Upland Sandpiper abundance increased with the percentage of forb cover in tallgrass pastures as well as with distance to rock outcrops (that is, any large [>2 m], upright, naturally occurring rocky structure) (Coppedge and others, 2008).

Brood rearing typically occurs in recently disturbed habitats and in areas with shorter, sparser vegetation. In Minnesota, broods used weedy fields, open areas within oldfields, and overgrazed pastures (Dorio, 1977; Dorio and Grewe, 1979). Marshy areas of sedge and cattails (*Typha* spp.) that had dried during drought were used as escape cover by broods. In Wisconsin, brood rearing occurred mostly in heavily grazed (vegetation <10 cm tall) pastures, followed by ungrazed pastures and hayfields (Ailes, 1976). Some broods were observed in idle fields, plowed fields, and cropland. Late-summer feeding occurred mainly in heavily and moderately grazed pastures; lightly grazed pastures were used infrequently (Ailes, 1980). In Illinois, broods were observed in wheat stubble, recently hayed legumes, redtop (*Agrostis gigantea*) intermixed with weeds, and moderately grazed pastures (Buhnerkempe and Westemeier, 1988). In Colorado, brood rearing occurred within short (\leq 27 cm tall) and cut alfalfa and small-grain fields (vegetation <27 cm tall) more often than expected, whereas small-grain fields (vegetation \geq 27 cm tall) were used less often than expected (Bolster, 1990). Bolster (1990) observed a noticeable movement of broods from pastures to alfalfa fields. Prior to migration, heavily grazed fields and cut and baled

alfalfa fields were used more often, and lightly grazed fields, weedy fields, tall alfalfa, and small-grain fields (vegetation ≥ 27 cm) were used less often than expected.

Moisture levels may affect the abundance of Upland Sandpipers, but as Niemuth and others (2017) stated, the biological meaning of climate variables in models characterizing bird-environment relationships is unclear; they are likely correlates of other factors (for example, plant community composition, primary and secondary productivity) that more directly influence species occurrence, likely in concert with other factors such as soils and landform. Using North American BBS data for four States within the Badlands and Prairies Bird Conservation Regions, Gorzo and others (2016) reported that Upland Sandpiper abundance was positively related to a within-year standardized temperature index, but not to the previous year's temperature index or to a standardized precipitation index. Using BBS data for seven States within the U.S. portion of the northern Great Plains, some of the same BBS routes used by Gorzo and others (2016), Niemuth and others (2017) reported that the occurrence of Upland Sandpiper exhibited a quadratic relationship with the means of long-term (30-year) precipitation and January temperatures, indicating that intermediate values of these climatic variables best explained the species' distribution.

Area Requirements and Landscape Associations

Territory sizes in Wisconsin ranged from 8 to 12 hectares (ha) (Wiens, 1969). In Kansas, home-range size during the breeding season averaged 199 ha for 21 males and 247.7 ha for 23 females (Mong, 2005). Males provided most of the care posthatching, and the average brood rearing home-range size of 200.8 ha based on nine males was three times as large as the nesting home-range size of 67.02 ha based on 14 males.

Upland Sandpipers are sensitive to habitat fragmentation (Herkert, 1991a; Herkert and others, 1993; Vickery, 1993; Winter, 1998; Ribic and others, 2009), and abundance may be positively correlated with patch size (Herkert, 1994; Vickery and others, 1994; Bollinger, 1995; Helzer, 1996; Thogmartin and others, 2006; Vos and Ribic, 2011). In Illinois, Upland Sandpipers were present in grasslands >30 ha (Herkert, 1991b, 1991c). In southwestern Missouri, Upland Sandpipers occurred only on tallgrass prairie fragments >75 ha (Winter, 1998). In Nebraska, Upland Sandpipers required a minimum area of 50–61 ha, with a perimeter-area ratio of 0.008, to reach 50 percent incidence (Helzer, 1996; Helzer and Jelinski, 1999). Occurrence of Upland Sandpipers was positively correlated with patch area and inversely correlated with perimeter-area ratio (Helzer and Jelinski, 1999). In Wisconsin,

Upland Sandpipers were found only on large grassland patches (>45 ha) and were absent from smaller patches (<10.5 ha) (Vos and Ribic, 2011). In a second Wisconsin study with patches ranging in size from 4 to 267 ha, Vos and Ribic (2013) reported that Upland Sandpipers occurred only on the largest prairie patch. However, in Canada, Garvey and others (2013) found no relationship between daily nest survival and patch size, proximity to an edge, amount of edge, distance to wetland edge, or to proportion of cropland or natural idled grasslands. In Maine, Upland Sandpipers were rare in areas <50 ha and reached 50 percent incidence in areas that were 200 ha (Vickery, 1993; Vickery and others, 1994).

Upland Sandpipers are generally intolerant of woody vegetation. In North Dakota mixed-grass prairies, Grant and others (2004) classified the Upland Sandpiper as a woodland-sensitive species. The species' maximum probability of occurrence never exceeded 30 percent within the study area, and the probability of occurrence declined to <20 percent at about 20 percent woodland cover. Upland Sandpipers were present in grasslands with a lower percentage of aspen woodland within 100 m and 500 m than in unoccupied grasslands. In North Dakota tallgrass prairies, Upland Sandpiper occurrence was negatively associated with grassland and woodland cover at the 100-m scale and with tree cover at the 400-m scale (Cunningham and Johnson, 2006). In Wisconsin, Upland Sandpiper abundance was highest in an 800-ha landscape with high grassland coverage and low forest coverage (Murray and others, 2008). Using BBS data from Minnesota, Wisconsin, and Michigan, Thogmartin and others (2006) reported Upland Sandpiper abundance was negatively associated with the percentage of forest in the landscape. Niemuth and others (2017) investigated the relationship between Upland Sandpiper occurrence and land use within an 800-m landscape of BBS points throughout the northern Great Plains; occurrence was positively associated with percent coverage of grasslands (native and tame), pasture and hayland (native and tame), shrubland, cropland, and emergent wetlands, but was negatively associated with percent coverage of forest, open water, and developed land.

Brood Parasitism by Cowbirds and Other Species

Brood parasitism by Brown-headed Cowbirds (*Molothrus ater*) is infrequent in Upland Sandpiper nests (Friedmann, 1963; Friedmann and Kiff, 1985; Houston and others, 2011). Upland Sandpipers are unsuitable cowbird hosts because their young are precocial and nidifugous. Rates of parasitism varied from 0 percent (several studies) to 8 percent of 13 nests (Beriman, 2007), as summarized in Shaffer and others (2019a).

Breeding-Season Phenology and Site Fidelity

Upland Sandpipers arrive on the breeding grounds from early April to early May (Buss and Hawkins, 1939; Lindmeier, 1960; Bent, 1962; Goering, 1964; Maher, 1973; Higgins and Kirsch, 1975; Ailes, 1976, 1980; Bowen, 1976; Dorio, 1977; Johnsgard, 1980; Buhnerkempe and Westemeier, 1988; Kantrud and Higgins, 1992). They depart from mid-July to late August (Buss and Hawkins, 1939; Bent, 1962; Goering, 1964; Wiens, 1969; Maher, 1973; Higgins and Kirsch, 1975; Ailes, 1976, 1980; Dorio, 1977; Johnsgard, 1980; Bolster, 1990).

Breeding-site fidelity has been observed (Ailes, 1976, 1980; Bowen, 1976; Dorio, 1977). In a Kansas population of radio-marked and color-banded birds, annual return rates varied from 20 to 50 percent; radio-harnessed birds had lower odds of returning than birds without radios (Mong and Sandercocock, 2007). Upland Sandpipers sometimes nest semicolonially (Buss and Hawkins, 1939; Bowen, 1976; Bowen and Kruse, 1993). Patterns of nest aggregation may reflect females preferentially settling near relatives, as in cases of joint settlement of female siblings or female-biased natal philopatry (Casey and others, 2011).

Time limitations within a nesting season make double-broodedness unlikely for Upland Sandpipers. However, renesting following failure of initial nests has been reported (Buss and Hawkins, 1939; Lindmeier, 1960; Dorio and Grewe, 1979).

Species' Response to Management

Because Upland Sandpipers use sites with a range of vegetation characteristics throughout the breeding season, management may have different effects on the species depending on the stage of the nesting cycle at the time of the disturbance.

Burning generally benefits Upland Sandpipers, especially by providing habitat for foraging. In Saskatchewan, Upland Sandpipers used a burned plot 2–3 years postburn during 3 years of postburn monitoring, but were not observed on an unburned plot (Pylypec, 1991). Bent (1962) suggested that burning and cultivation of mixed-grass prairie in Saskatchewan forced Upland Sandpipers to nest in cultivated fields. In Minnesota, a 75-percent reduction in nesting cover due to spring fire may have reduced the number of Upland Sandpipers in the year of the burn (Lindmeier, 1960); numbers of breeding pairs returned to preburn levels in the following year. Likewise, in Wisconsin, burning reduced or eliminated nesting attempts in fields where nesting had occurred in the year previous to the burn (Buss and Hawkins, 1939). In mixed-grass prairies in North Dakota, Upland Sandpipers were more abundant immediately following a burn and 1 year after a burn than 2–15 years postburn (Johnson, 1997). In tallgrass prairies of southeastern North Dakota, Upland Sandpipers were most abundant on burned pastures in the wettest year of a

3-year study (Ahlering and Merkord, 2016). In South Dakota, Upland Sandpipers used a burned native pasture significantly more than an unburned pasture (Huber and Steuter, 1984). In another South Dakota study, the species nested at high densities on idle, mowed, and burned prairies (Lokemoen and Duebbert, 1974). In Illinois tallgrass prairies, Upland Sandpipers were most abundant 2 years postburn, but were absent 3 years postburn (Herkert, 1994). In Illinois grasslands that were seeded to both native and tame grasses, Upland Sandpipers preferred nesting in fields 1 year after a burn (Buhnerkempe and Westemeier, 1988). Burned fields were not preferred in the following years, and number of years postburn did not affect nest density.

Many studies have evaluated Upland Sandpiper response to burning in the tallgrass prairies of the Kansas and Oklahoma Flint Hills. Upland Sandpipers appeared to use unburned grasslands for nesting and foraging and annually burned grasslands for foraging, but they nested in watersheds that were not burned in spring (Zimmerman, 1993). Goering (1964) found nests in burned native grasslands as well as in heavily grazed and ungrazed native and tame grasslands. Eddleman (1974) observed Upland Sandpipers on heavily grazed and annually burned pastures, moderately grazed and unburned pastures, and ungrazed and burned areas; Upland Sandpipers did not use unburned and ungrazed areas. Robel and others (1998) found that Upland Sandpipers were present only on spring-burned, seeded-native CRP fields and not on unburned fields. Bowen (1976) observed that Upland Sandpiper abundance did not differ between burned and unburned pastures in Kansas. Radio-marked Upland Sandpipers preferred sites that were recently burned and grazed, followed by burned and ungrazed sites, unburned and ungrazed sites, and unburned and grazed sites (Mong, 2005). The species preferred the most recently burned sites over areas that had not been burned for more than 1 year and showed no preference for grazed or ungrazed areas. Hovick and others (2015) established seven experimental pastures with varying levels of patchiness ranging from annually burned with spring-only fires to a 4-year fire-return interval to examine the interaction of fire and grazing; Upland Sandpiper density was positively influenced by number of patches (that is, increasing heterogeneity), was not related to fire-return interval, and was negatively related to vegetation height.

In the Kansas Flint Hills, Powell (2006) examined the effect of American bison (*Bison bison*) grazing and prescribed burns on grassland bird abundance. Upland Sandpipers were more abundant in pastures in the season of burn and ≥ 4 years postburn than 1–3 years postburn. Upland Sandpipers also increased with bison grazing. Bison were stocked at low intensity (5 ha per animal with the expected consumption of 25 percent of aboveground plant growth). Powell (2008) also examined the effect of cattle grazing and prescribed burns on grassland bird abundance. Upland Sandpiper abundance was higher during the year of burns than 1–3 years after the last burn and in areas grazed by cattle. Cattle were grazed at low intensity (3 ha per cow-calf pairs with the expected consumption of about 25 percent of aboveground plant growth). Upland

Sandpipers were significantly more abundant in burned idle grasslands than unburned idle or hayed grasslands (Powell and Busby, 2013).

In the Oklahoma Flint Hills, Upland Sandpipers were as abundant in annually burned pastures as in pastures burned in a patch-mosaic pattern (that is, portions of the pasture were burned on a 3-year fire-return interval; Coppedge and others, 2008). However, within the same tallgrass pastures, Upland Sandpiper abundance was five times higher in patches with recent disturbances (that is, patches averaging 100 ha burned once in either spring or autumn every 3 years with cattle stocked at 1.2 ha per 270-kilogram [kg] steer) than pastures not burned within the past 36 months, and 2.5 times greater in pastures with recent focal disturbances than in annually spring-burned pastures stocked from mid-April to mid-July (Fuhlendorf and others, 2006).

Upland Sandpipers readily use hayfields, although haying disturbances may cause nest failure (Ducey and Miller, 1980; Houston and others, 2011). Igl and Johnson (2016) assessed the effects of emergency and managed haying on grassland breeding birds in 483 CRP grasslands in nine counties in four States in the northern Great Plains between 1993 and 2008. Upland Sandpiper densities in CRP grasslands that had been idled for more than 5 years did not differ from sandpiper densities in CRP grasslands that had been hayed 1, 2, 3, or 4 years earlier. In North Dakota, Upland Sandpipers used previously idled areas only after the areas were mowed (Messmer, 1990). In Wisconsin, Upland Sandpipers occurred at higher densities in haylands than in pastures and wet prairies, although differences in densities were not statistically significant (Sample, 1989). Upland Sandpipers were located in annually mowed native prairies (dominated by porcupinegrass) in Iowa (Kendeigh, 1941). In Iowa and Wisconsin CRP fields planted to switchgrass (*Panicum virgatum*), Upland Sandpipers were more abundant in harvested plots than in unharvested plots (Murray and Best, 2003; Roth and others, 2005). In Wisconsin, the species nested in hayland the first year after mowing (Ailes, 1976). In Illinois grasslands that were seeded to native and tame grasses, Upland Sandpipers preferred nesting in fields 1 year after the fields were rotary mowed, whereas grass meadows harvested for seed the previous year were used as nesting habitat less frequently than were other grasslands (Buhnerkempe and Westemeier, 1988). In Missouri, hayfields were preferred over seed-combined fields and were used for foraging and loafing (Skinner, 1974). Skinner (1974, 1975) also compared Upland Sandpiper density between idle fields and fields subjected to haying, seed combining, or grazing at four intensities. Density of Upland Sandpipers was highest under moderate grazing (vegetation 10.2–30.4 cm tall, 20–40 percent grass and forb coverage at 25 cm tall) and heavy grazing (vegetation 0–10.2 cm tall, <20 percent grass and forb coverage at 25 cm tall) (Skinner, 1975, 1982). Upland Sandpipers were present in hayed, combined, and lightly grazed fields (vegetation >30.4 cm tall) but not in idle fields.

Upland Sandpipers use grazed areas for nesting, foraging, and brood rearing (Ailes, 1976; Dorio, 1977), although

the effects of grazing vary among studies. Nest loss occasionally occurs as a result of trampling by cattle (Buss and Hawkins, 1939; Ailes, 1976, 1980; Dorio, 1977; Bowen and Kruse, 1993). In Alberta, Upland Sandpipers were found only on deferred-grazed native areas (Prescott and Wagner, 1996). Treatments included tame pastures of crested wheat-grass grazed in spring from late April to mid-June, native grasslands grazed in early summer, and native grasslands grazed after July 15 (deferred); the control was a continuously grazed native pasture. In Saskatchewan, Upland Sandpipers were observed on grazed pastures but not on ungrazed areas (Dale, 1984). In Ontario, Upland Sandpipers preferred lightly grazed pastures over hayland, oldfields, and cropland (Speirs and Orenstein, 1967). In North Dakota, Upland Sandpipers were more attracted to heavily grazed, native grasslands than to lightly grazed, moderately grazed, or mowed grasslands, although densities were relatively high in all habitats compared to other bird species (Kantrud 1981). In south-central North Dakota, Upland Sandpipers only occurred in extremely grazed pastures (20 percent of forage produced in an average year remained, equating to an average grazing rate of 6.8 animal unit months [AUMs] per ha) and not in lightly, moderately, or heavily grazed pastures (35–65 percent forage removed, or 1.1–4.2 AUMs per ha) (Salo and others, 2004). In tallgrass prairies of southeastern North Dakota, grazing intensity of 0–4.57 AUMs per ha (1 cow-calf pair on range for 1 month, regardless of weight) did not negatively affect Upland Sandpiper abundance (Ahlering and Merkord, 2016). Over a broader geographic range (North Dakota, South Dakota, Montana, Wyoming, Colorado, Nebraska), however, Kantrud and Kologiski (1982) did not find any relationship between density of Upland Sandpipers and grazing intensity. Upland Sandpiper densities were significantly higher in idle and grazed mixed-grass prairies than in tame DNC fields; the species also occurred in areas the first year after grazing (Renken, 1983; Renken and Dinsmore, 1987). In Nebraska, Upland Sandpipers were present on areas grazed by cattle and areas that were grazed by American bison and that also were burned (Griebel and others, 1998). In another Nebraska study, Upland Sandpiper densities were similar on grazed and ungrazed plots, but densities in ungrazed plots were highest under moderate moisture levels (Kim and others, 2008). The species also preferred pastures that were grazed year-round by cows and calves than pastures that were grazed by steers. In a third Nebraska study, avian diversity and density were higher on grazed than ungrazed areas because of the presence of species, including the Upland Sandpiper, that were not present on ungrazed areas (Cole and Sharpe, 1976). In the Nebraska Sandhills, Kempema (2007) examined the effect of grazing system duration on Upland Sandpiper density. Average values during the growing season (May 1 to September 30) for short duration was a rotation of 3 days of grazing at 1.4 AUMs per ha (11 animals per ha); medium duration was 23 days at 1.3 AUMs per ha (2.5 animals per ha), and long duration was 78 days at 1.4 AUMs per ha (0.6 animals per ha). Upland Sandpiper densities were similar among grazing systems, with

the highest density occurring on the long-duration system. Shrub coverage provided the best explanation for variation in densities in that as shrub cover increased, densities decreased. In Kansas, Upland Sandpipers preferred grazed pastures more than ungrazed pastures (Bowen, 1976).

Several studies have evaluated the effects of grazing on nest productivity in mixed-grass prairies in south-central North Dakota. Messmer (1990) and Sedivec (1994) compared rotational grazing systems, specifically short-duration grazing and twice-over rotation grazing, to season-long grazing and idle grasslands. Short-duration grazing involves a system of pastures rotated through a grazing schedule of about 1 week grazed and 1 month ungrazed, repeated throughout the season (usually late May or early June to October). Twice-over rotation involves grazing a number of pastures twice per season, with about a 2-month rest in between grazing. Season-long grazing involves leaving cattle on the same pasture throughout the growing season. Research by Messmer (1985, 1990) revealed that nest density and nest success were higher on twice-over deferred and season-long grazing systems than on idle pastures, but that average density of breeding Upland Sandpipers was highest on the short-duration grazing system. As range conditions on the short-duration pastures improved and cover increased, sandpiper density decreased. In a continuation of Messmer's study, Sedivec (1994) reported that nest density was significantly higher on grazed than on idle grasslands. Both authors concluded that grazing is compatible with the breeding needs of Upland Sandpipers. Bowen and Kruse (1993) and Kirsch and Higgins (1976) examined seasonality and intensity of grazing, respectively. Bowen and Kruse (1993) compared nest density among five grazing treatments: autumn grazing, autumn-and-spring grazing, season-long grazing, spring grazing, and ungrazed. Nest densities were lower in pastures subjected to grazing during the nesting season (autumn-and-spring grazing, season-long grazing, and spring grazing) than in control fields or fields with autumn grazing. Nest densities did not differ between spring grazing with high stocking density (3.7 head of cattle per ha and grazing rate of 3.1 AUMs per ha) to that of season-long grazing with low stocking density (1.0 head of cattle per ha and grazing rate of 2.45 AUMs per ha). Nest densities were significantly lower in years after pastures had been subjected to season-long and autumn-and-spring grazing than in the year before grazing treatments occurred. Allowing mixed-grass prairies to remain idle for 2–3 years between grazing treatments was not detrimental to breeding Upland Sandpipers (Bowen and Kruse, 1993).

Within mixed-grass and tallgrass prairies in South Dakota, nest densities did not differ between idle sites and sites that were grazed in May at a grazing rate of 1–2.5 AUMs per ha and in which 20–80 percent of the current year's growth was removed (Kaiser, 1979). Fourteen nests were found within a 256-ha fragment of moderately grazed prairie in South Dakota (Lokemoen and Duebbert, 1974). In North Dakota, Kirsch and Higgins (1976) reported that mean nest productivity was lowest on tilled areas (where no nests were

observed), higher on grazed and idle areas, and highest on burned areas. The highest nest density of 6.8 nests per 40.5 ha was on a grassland area burned 2 years previously. In North Dakota, South Dakota, Montana, and Manitoba, nest success was higher in idle grasslands than in grazed pastures (Kantrud and Higgins, 1992). In eastern Kansas, Upland Sandpipers preferred native pastures to CRP grasslands seeded to native grasses, both of which were annually burned; nests were found only in pastures (Klute, 1994; Klute and others, 1997). In Missouri, nests were found on grazed tallgrass prairie (Skinner and others, 1984).

Upland Sandpipers have been reported as relatively uncommon in CRP and DNC grasslands and rowcrops compared to other habitats (Kantrud and Higgins, 1992; Klute, 1994; Best and others, 1997). The species may prefer cropland to CRP fields or idle fields (Skinner, 1975; Johnson and Igl, 1995; Best and others, 1997; Herkert, 2009), although Patterson (1994) and Patterson and Best (1996) reported that the species nested in Iowa CRP fields but not in rowcrops. In a multi-State study, abundance of Upland Sandpipers decreased after the establishment of CRP fields (Herkert, 2009). In North Dakota, densities of Upland Sandpipers were higher in idle and grazed native prairies than in DNC fields (Renken and Dinsmore, 1987). At a landscape scale, Uden and others (2015) evaluated four scenarios of land use change in Nebraska, and the influence of rowcrop, CRP, and switchgrass area on Upland Sandpiper abundance. The first scenario was a baseline condition in which some rowcrops were converted to switchgrass under current conditions of climate, irrigation limitations, commodity prices, ethanol demand, and continuation of the CRP. The second scenario converted more rowcrops to switchgrass. The third scenario converted all CRP to switchgrass, and the final scenario converted all CRP to rowcrops. Upland Sandpiper abundance increased 0.7–2 percent under the first two scenarios, increased little under scenario 3, and did not change under scenario 4, indicating that replacing rowcrops with switchgrass was more beneficial to Upland Sandpipers than replacing CRP with switchgrass or rowcrops. Conversely, Veech (2006) used BBS data to characterize the landscape within a 30-kilometer (km) radius of populations of Upland Sandpipers throughout the Great Plains that were increasing or decreasing; CRP comprised a greater proportion of the landscape for increasing populations than for decreasing populations, and urban land comprised a greater proportion for decreasing populations. The proportion of rangeland did not differ between increasing and decreasing populations.

Cultivation may negatively affect Upland Sandpipers (Bent, 1962; Ailes, 1976; Faanes and Lingle, 1995) by eliminating brood rearing areas and forcing broods to use edge habitats (Dorio, 1977). In Michigan, Upland Sandpipers preferred hayfields, pastures, and grasslands over rowcrop agricultural fields (Korte, 2013). In Nebraska, Upland Sandpipers preferred untilled areas, such as alfalfa fields and pastures, more than tilled areas; however, the species foraged in corn fields before and just after emergence of the corn plants (Ducey and Miller, 1980). In south-central North Dakota,

hatching success was lowest on annually tilled cropland (none of six nests hatched) and highest on burned and idle grasslands (Kirsch and Higgins, 1976). Also in south-central North Dakota, Upland Sandpipers seemed to prefer minimum-tillage (seeding into untilled or moderately tilled land) and organic farming (cultivation and crop rotation, but no chemicals, were used to control weeds) over conventional tillage (spring and fall tillage and use of herbicides); most nests were placed in wheat stands that were physically similar to grasslands (Lokemoen and Beiser, 1997). In another North Dakota study, Upland Sandpipers preferred nesting in untilled uplands (road rights-of-way, bands of vegetation around wetlands, heavily grazed grasslands, and idled grasslands) over fallow (bare ground), mulched or standing stubble fields, or growing small grain (Higgins 1975). In Illinois, the species preferred seeded grasses mixed with forbs, such as young, rotary-mowed seedings and older meadows harvested for grass seed, as nesting habitat (Buhnerkempe and Westemeier, 1988). Upland Sandpipers preferred nesting in fields that were >5 years postseeding, especially in those fields that were >8 years old. Fields that had homogeneous vegetation or that were planted to smooth brome were rarely selected. Grass meadows that were harvested for seed the previous year, brome, and wheat stubble-legume fields were not used for nesting. In Nebraska, woody encroachment into wet prairie and conversion of upland prairie to cropland negatively affected Upland Sandpipers (Faanes and Lingle, 1995). In Indiana, the species used oat fields in spring until the fields were harvested (Snyder and others, 1987).

Some pastures were used more frequently during years when they had been fertilized with nitrogen (Bowen, 1976). In Wisconsin, fertilizing with manure reduced or completely excluded nesting by Upland Sandpipers; however, Upland Sandpipers were found nesting in grass clumps formed around manure droppings (Buss and Hawkins, 1939).

Upland Sandpipers may avoid wind facilities. At two of three wind facilities in mixed-grass prairies in North Dakota and South Dakota, Upland Sandpipers exhibited displacement from areas within and surrounding wind-turbine facilities, with both immediate (1-year postconstruction) and delayed (2–5 years postconstruction) displacement occurring at one facility, and delayed displacement at a second facility (Shaffer and Buhl, 2016). Avoidance distances varied from within 100 m of turbines to overall displacement from the study area.

Management Recommendations from the Literature

Maintaining unbroken native prairie or converting agricultural fields to grasslands will be beneficial to Upland Sandpipers (Veech, 2006; Uden and others, 2015). Large (> 100 ha), contiguous tracts of prairie should be maintained to reduce edge habitats, to provide habitat heterogeneity, and to decrease nest depredation (Herkert and others, 1993;

Herkert, 1994; Klute, 1994; Helzer, 1996; Mong, 2005). Herkert and others (1993) recommended maintaining grassland blocks that are within 1.6 km of each other and that are contiguous with adjacent grassy habitats (for example, pastures, hayfields). Helzer and Jelinski (1999) highlighted the importance of considering shape and area of management units; perimeter-area ratio strongly influenced occurrence of Upland Sandpipers in Nebraska (Helzer and Jelinski, 1999).

Several studies have emphasized the importance of managing native prairies every 2–3 years by burning, grazing, haying, or idling (Kaiser, 1979; Kantrud, 1981; Bowen and Kruse, 1993; Ahlering and Merkord, 2016). Grazing provided habitat conditions for nesting to a lesser extent but was more compatible than cropland or tame-grass seedings. In Oklahoma tall-grass prairies, replacing annual burning and grazing with patch burning and grazing increased vegetative heterogeneity and Upland Sandpiper abundance (Fuhlendorf and others, 2006). In Wisconsin CRP fields of switchgrass, Upland Sandpipers used mowed fields but not unmowed fields; mowed fields had lower vegetation height-density and litter cover than unmowed fields (Roth and others, 2005).

Encroachment of woody vegetation into grasslands may be detrimental to Upland Sandpipers and other grassland birds (Herkert and others, 1993). Grant and Murphy (2005) recommended the reintroduction of fire and grazing on lands managed for grassland birds to stem the encroachment of woody vegetation in northern Great Plains grasslands. Grant and others (2004) suggested that managers focus initial restoration efforts on grasslands with <20 percent woodland encroachment because these grasslands would have the most immediate and lasting conservation benefit for grassland birds. Programs that encourage the planting of trees and tall shrubs within grasslands are discouraged (Grant and others, 2004). Cunningham and Johnson (2006) recommended removal of trees for improving habitats for grassland birds; however, perches, such as fence posts, rock piles, or tree stumps, may be important for displaying Upland Sandpipers (White, 1983).

Several authors have recommended that management disturbances (for example, burning, mowing, or plowing) should be avoided or delayed during the nesting season (Buss and Hawkins, 1939; Lokemoen and Beiser, 1997). For example, Bolster (1990) and Patterson (1994) recommended that mowing and spraying of pesticides in CRP grasslands should be delayed until after July to avoid disturbances during the peak nesting season. Buhnerkempe and Westemeier (1988) recommended delaying mowing of habitat for nesting and brood rearing until July 1 or later. Oetting and Cassel (1971) recommended delaying mowing on road rights-of-way until late July.

Upland Sandpipers require a mosaic of habitat types throughout the breeding season, including grasslands of various heights and densities as well as cropland (Bolster, 1990). Grazed, burned, and hayed fields provide suitable habitat for feeding, loafing, and brood rearing, but undisturbed fields are needed for nesting (Lindmeier, 1960; Bowen and Kruse, 1993).

Rotational burning of patches in pastures may benefit Upland Sandpipers by providing vegetation heterogeneity (Fuhlendorf and others, 2006). Herkert (1994) recommended that 20 to 30 percent of grassland fragments <80 ha in size should be burned annually. Small fragments should have <50 percent of their area burned at a time, and, if next to other fragments, should be burned on a rotating schedule that allows unburned fragments to be adjacent to burned fragments (Herkert, 1994). Herkert and others (1993) recommended that burns should occur from March to early April or from October to November to avoid disturbances during the nesting season.

Grazing can be used to improve habitat for Upland Sandpipers. For example, Skinner (1974) recommended moderate grazing levels to provide diverse grass heights and densities. Skinner (1974) also suggested that a rotational grazing system of two or more grazing units also can provide a diversity of grass heights and densities within and among units. Bowen and Kruse (1993) and Sedivec (1994) indicated that season-long grazing should be avoided, and where grazing is necessary, grazing should be delayed until mid- to late June to maintain nest densities. To provide more undisturbed cover, Sedivec (1994) recommended rotational grazing over season-long grazing during the nesting season. To benefit nesting sandpipers as well as to optimize weight gain of calves in rotational grazing systems, Sedivec (1994) recommended that grazing should be delayed until late May to early June. Sedivec (1994) suggested following the stocking-rate recommendations as outlined by the Soil Conservation Service (1984); rates may be slightly higher for rotational grazing (Sedivec, 1994).

Kantrud and Higgins (1992) and Lokemoen and Beiser (1997) highlighted the importance of maintaining areas of undisturbed habitat during the nesting season by encouraging no-till or minimum-tillage practices instead of conventional annual tillage practices. Nest productivity may be low on annually tilled cropland and former cropland planted to grass and legumes (Kirsch and Higgins, 1976). On farms that adopt organic farming practices, Lokemoen and Beiser (1997) recommended delaying the first tilling operations on organic fallow fields until late June or early July to reduce the destruction of nests.

Buhnerkempe and Westemeier (1988) emphasized the importance of maintaining heterogeneous fields of cool-season, tame grasses that are >5 years old; to obtain a mixture of forbs and grasses, fields should not be reseeded until they are 10–12 years old. Management of seeded grasses should include allowing them to idle, rotary mowing to a height of 15–30 cm on a 3-year rotation, or burning (Buhnerkempe and Westemeier, 1988). Moderate grazing may provide suitable habitat in native and tame grasses, but more research is needed (Buhnerkempe and Westemeier, 1988).

In some cases, management might involve the avoidance or reduction of impacts to habitat and avian populations from external stressors. Shaffer and others (2019b) developed the avian-impact offset method to help guide compensatory

mitigation of habitat loss associated with energy development. The avian-impact offset method calculates the biological value (measured in terms of avian numbers) lost when Upland Sandpipers avoid otherwise suitable breeding habitat because of energy development. The method's output (ha of grassland necessary to offset development) converts biological value to the traditional unit of measure in which land is purchased or sold, so that compensatory mitigation can be undertaken in the form of conservation easements or grassland reconstruction. The areal unit of measure also lends itself readily to mapping applications in which conservation delivery of offsetting measures can be viewed at local, regional, or landscape scales. To this end, Shaffer and others (2019b) used models developed from Niemuth and others (2017) to develop a decision-support tool that identifies locations for placement of compensatory offset sites with equivalent biological value as impact sites. Alternatively, the tool can be used prior to development of energy facilities to identify locations that would require little compensatory mitigation if developed, relative to other potential locations.

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Table F1. Measured values of vegetation structure and composition in Upland Sandpiper (*Bartramia longicauda*) breeding habitat by study. The parenthetical descriptors following authorship and year in the “Study” column indicate that the vegetation measurements were taken in locations or under conditions specified in the descriptor; no descriptor implies that measurements were taken within the general study area.

[cm, centimeter; %, percent; <, less than; --, no data; WPA, Waterfowl Production Area; ≥, greater than or equal to; >, greater than; CRP, Conservation Reserve Program; DNC; dense nesting cover]

Study	State or province	Habitat	Management practice or treatment	Vegetation height (cm)	Vegetation height-density (cm)	Grass cover (%)	Forb cover (%)	Shrub cover (%)	Bare ground cover (%)	Litter cover (%)	Litter depth (cm)
Ailes, 1976 (nests)	Wisconsin	Tame grassland	Multiple	<40	--	--	--	--	--	--	--
Ailes, 1976 (brood-rearing)	Wisconsin	Tame grassland	Multiple	<10	--	--	--	--	--	--	--
Bowen and Kruse, 1993 (nests)	North Dakota	Mixed-grass prairie (WPA)	Multiple	--	5–20 ^a	≥50	<50	--	--	--	--
Buhnerkempe and Westemeier, 1988 (nests)	Illinois	Tame grassland	Multiple	17–33	--	--	--	--	--	--	--
Colwell and Oring, 1990 (nests)	Saskatchewan	Mixed-grass prairie	Grazed, idle	>15	--	--	--	--	--	--	--
Dorio, 1977 (nests)	Minnesota	Multiple	Multiple	22.5–35	--	--	--	--	12	25.2	--
Dorio, 1977 (foraging)	Minnesota	Multiple	Multiple	2.5–30	--	--	--	--	--	--	--
Fritcher and others, 2004 ^{b,c}	South Dakota	Mixed-grass prairie	--	26.6–51.8	5.8–17 ^a	85.7–91.6	18–26.1	--	1.8–12.9	80.7–94.6	0.9–3.1
Fuhlendorf and others, 2006 ^d	Oklahoma	Tallgrass prairie	Annual complete burn and grazed	14.7	--	63	18	--	20.3	8	--
Fuhlendorf and others, 2006 ^d	Oklahoma	Tallgrass prairie	Patch burn and grazed	21.7	--	55.7	19	--	14.7	50.3	--
Garvey and others, 2013 (nests)	Alberta, Manitoba, Saskatchewan	Multiple	Multiple	--	7 ^a	--	--	--	--	--	--
Grant and others, 2004	North Dakota	Mixed-grass prairie	--	47	--	--	--	13.2	--	--	3.3
Higgins and others, 1969 (nests)	North Dakota, South Dakota	Mixed-grass prairie	Multiple	15–61	--	--	--	--	--	--	--
Hull and others, 1996	Kansas	Tame grassland (CRP)	Burned	--	--	--	50.1	--	--	--	--
Kaiser, 1979 (nests)	South Dakota	Mixed-grass prairie, tallgrass prairie	Grazed	12.7–63.5	--	--	--	--	--	--	--

Table F1. Measured values of vegetation structure and composition in Upland Sandpiper (*Bartramia longicauda*) breeding habitat by study. The parenthetical descriptors following authorship and year in the “Study” column indicate that the vegetation measurements were taken in locations or under conditions specified in the descriptor; no descriptor implies that measurements were taken within the general study area.—Continued

[cm, centimeter; %, percent; <, less than; --, no data; WPA, Waterfowl Production Area; ≥, greater than or equal to; >, greater than; CRP, Conservation Reserve Program; DNC; dense nesting cover]

Study	State or province	Habitat	Management practice or treatment	Vegetation height (cm)	Vegetation height-density (cm)	Grass cover (%)	Forb cover (%)	Shrub cover (%)	Bare ground cover (%)	Litter cover (%)	Litter depth (cm)
Kantrud and Higgins, 1992 (nests)	North Dakota, South Dakota, Montana, Manitoba	Multiple	Multiple	--	12 ^a ,26 ^c	--	--	--	--	36 ^f	--
Kirsch and Higgins, 1976 (nests)	North Dakota	Multiple	Multiple	15.4–30.8	--	--	--	--	--	--	--
Lindmeier, 1960 (nests)	Minnesota	Tallgrass prairie, tame grassland	Idle, burned	25.4	--	--	--	--	--	--	--
Messmer, 1990	North Dakota	Mixed-grass prairie	Multiple	50–70	--	--	--	--	--	--	3.8–9.1
Murray and Best, 2003	Iowa	Tame grassland (CRP)	Total-harvested switchgrass	80.9	71 ^a	51.6	19.6	0.4	5	23.2	1.9
Murray and Best, 2003	Iowa	Tame grassland (CRP)	Strip-harvested switchgrass	81.7	75 ^a	53.3	17.5	0.1	2.8	29.6	3.5
Murray and Best, 2003	Iowa	Tame grassland (CRP)	Unharvested switchgrass	78.1	71 ^a	32.9	25.4	2.1	2.9	22.9	5.5
Niemi and Hanowski, 1983 (territories)	Minnesota	Tallgrass prairie	--	79	--	--	--	--	--	--	--
Powell and Busby, 2013	Kansas	Tallgrass prairie	Unburned idle	93 ^f	--	--	--	--	--	--	12.5
Powell and Busby, 2013	Kansas	Tallgrass prairie	Burned idle	52.1 ^g	--	--	--	--	--	--	0.3
Powell and Busby, 2013	Kansas	Tallgrass prairie	Hayed	50.1 ^g	--	--	--	--	--	--	6.5
Powell and Busby, 2013	Kansas	Tallgrass prairie	Grazed	74 ^f	--	--	--	--	--	--	9.4
Renken, 1983 ^h	North Dakota	Tame grassland (DNC)	Idle, grazed	--	11 ^a	57.4	23.5	5.7	0.5	98.8	2.3
Roth and others, 2005	Wisconsin	Tame grassland (CRP)	Harvested warm-season	--	12.4 ^a	--	33.2	--	--	--	1.4
Salo and others, 2004	North Dakota	Mixed-grass prairie	Extreme grazing intensity	17.5 ⁱ	7.9 ^a	--	--	--	--	--	0.9
Sample, 1989	Wisconsin	Multiple	--	45.1	13.8 ^a	--	81.2 ^j	0.5	4.2	10.9	--

Table F1. Measured values of vegetation structure and composition in Upland Sandpiper (*Bartramia longicauda*) breeding habitat by study. The parenthetical descriptors following authorship and year in the “Study” column indicate that the vegetation measurements were taken in locations or under conditions specified in the descriptor; no descriptor implies that measurements were taken within the general study area.—Continued

[cm, centimeter; %, percent; <, less than; --, no data; WPA, Waterfowl Production Area; ≥, greater than or equal to; >, greater than; CRP, Conservation Reserve Program; DNC; dense nesting cover]

Study	State or province	Habitat	Management practice or treatment	Vegetation height (cm)	Vegetation height-density (cm)	Grass cover (%)	Forb cover (%)	Shrub cover (%)	Bare ground cover (%)	Litter cover (%)	Litter depth (cm)
Sedivec, 1994 (nests)	North Dakota	Mixed-grass prairie	Multiple	--	12.7 ^a	--	--	--	--	--	--
Skinner, 1974	Missouri	Tallgrass prairie	Moderately grazed	10–30.4	--	--	--	--	--	--	--
Skinner, 1974	Missouri	Tallgrass prairie	Heavily grazed	0–10.2	--	--	--	--	--	--	--
Wiens, 2007 (nests)	North Dakota	Tame grassland	Multiple	22	21 ^a	43	40	--	--	--	1.4
Wiens, 2007 (plots)	North Dakota	Tame grassland	Multiple	--	22 ^a	53	30	--	--	--	1.7

^aVisual obstruction reading (Robel and others, 1970).

^bRange of averages across seral stages within study area.

^cThe sum of the percentages is >100%, based on methods described by the authors.

^dThe sum of the percentages is >100%, based on the modified point-quadrat technique as described by the authors.

^eEffective vegetation height.

^fStanding dead vegetation.

^gLive vegetation height.

^hThe sum of the percentages is >100%, based on the modified point-quadrat technique of Wiens (1969).

ⁱMean grass height.

^jHerbaceous vegetation cover.

The Effects of Management Practices on Grassland Birds—Willet (*Tringa semipalmata inornata*)

By Jill A. Shaffer,¹ Lawrence D. Igl,¹ Douglas H. Johnson,¹ Marriah L. Sondreal,¹
Christopher M. Goldade,^{1,2} Barry D. Parkin,¹ Jason P. Thiele,^{1,3} and Betty R. Euliss¹

Chapter I 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^{1,4}

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Figure

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Table

11. Measured values of vegetation structure and composition in Willet (*Tringa semipalmata inornata*) breeding habitat by study.....9

Conversion Factors

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
hectare (ha)	0.003861	square mile (mi ²)

Abbreviations

DNC dense nesting cover

PPR Prairie Pothole Region

spp. species (applies to two or more species within the genus)

Acknowledgments

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The Effects of Management Practices on Grassland Birds—Willet (*Tringa semipalmata inornata*)

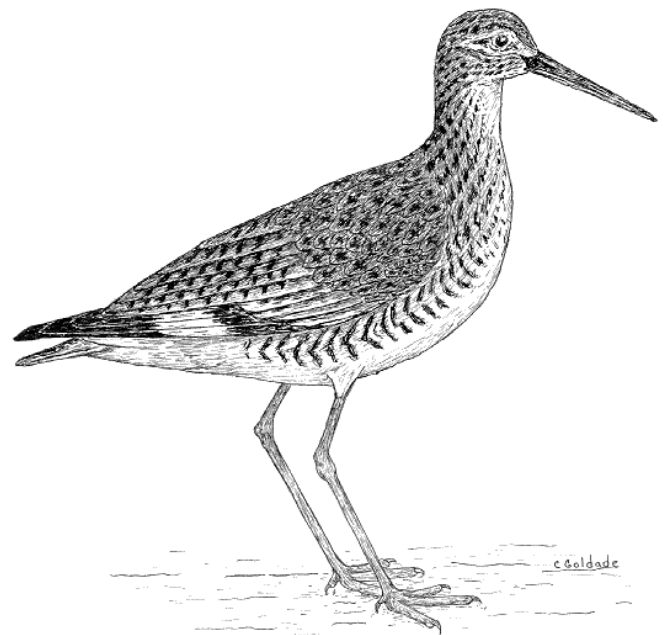
By Jill A. Shaffer,¹ Lawrence D. Igl,¹ Douglas H. Johnson,¹ Marriah L. Sondreal,¹ Christopher M. Goldade,^{1,2} Barry D. Parkin,¹ Jason P. Thiele,^{1,3} and Betty R. Euliss¹

Capsule Statement

Keys to Willet (*Tringa semipalmata inornata*) management include providing large expanses of native grasslands and wetland complexes. Wetland complexes should contain a diversity of wetland sizes and classes, such as ephemeral, temporary, seasonal, semipermanent, and permanent wetlands (wetland classifications based on Stewart and Kantrud, 1971), as well as intermittent streams. Willets use wetlands of various salinities. Willets require short, sparse upland grasslands for nesting and foraging and wetland complexes for foraging. Broods use taller, denser grass cover than do nesting adults. Willets have been reported to use habitats with less than or equal to 70 centimeters (cm) average vegetation height, 4–23 cm visual obstruction reading, 15 percent bare ground, 38 percent litter cover, and 1–9 cm litter depth. The descriptions of key vegetation characteristics are provided in table II (after the “References” section). Vernacular and scientific names of plants and animals follow the Integrated Taxonomic Information System (<https://www.itis.gov>).

Breeding Range

Two subspecies of Willets breed in North America: the Eastern Willet (*Tringa semipalmata semipalmata*) and the Western Willet (*Tringa semipalmata inornata*). This account deals only with the Western Willet, which breeds on the Great Plains, and not with the Eastern Willet, which breeds on the Atlantic and Gulf coasts of North America. Western Willets breed from central Alberta and Montana through southern Manitoba, North Dakota, western Minnesota, and South Dakota; south to south-central Oregon and central California; and east to northern Nevada, Idaho, northern Utah, Wyoming, northern Colorado, and western Nebraska (National Geographic Society, 2011). The relative densities of



Willet. Illustration by Christopher M. Goldade, U.S. Geological Survey.

both subspecies of Willets in the United States and southern Canada, based on North American Breeding Bird Survey data (Sauer and others, 2014), are shown in figure II (not all geographic places mentioned in report are shown on figure).

Suitable Habitat

During the breeding season, Willets require large expanses of short, sparse grasslands for nesting and foraging, and wetland complexes for foraging (Stewart, 1975; Weber, 1978; Higgins and others, 1979; Kantrud and Stewart, 1984; Ryan and Renken, 1987; Colwell and Oring, 1988a, 1990; Eldridge, 1992; Kantrud and Higgins, 1992; Prescott and others, 1995; Lowther and others, 2001). In upland and wetland habitats, adults with broods use somewhat taller, denser

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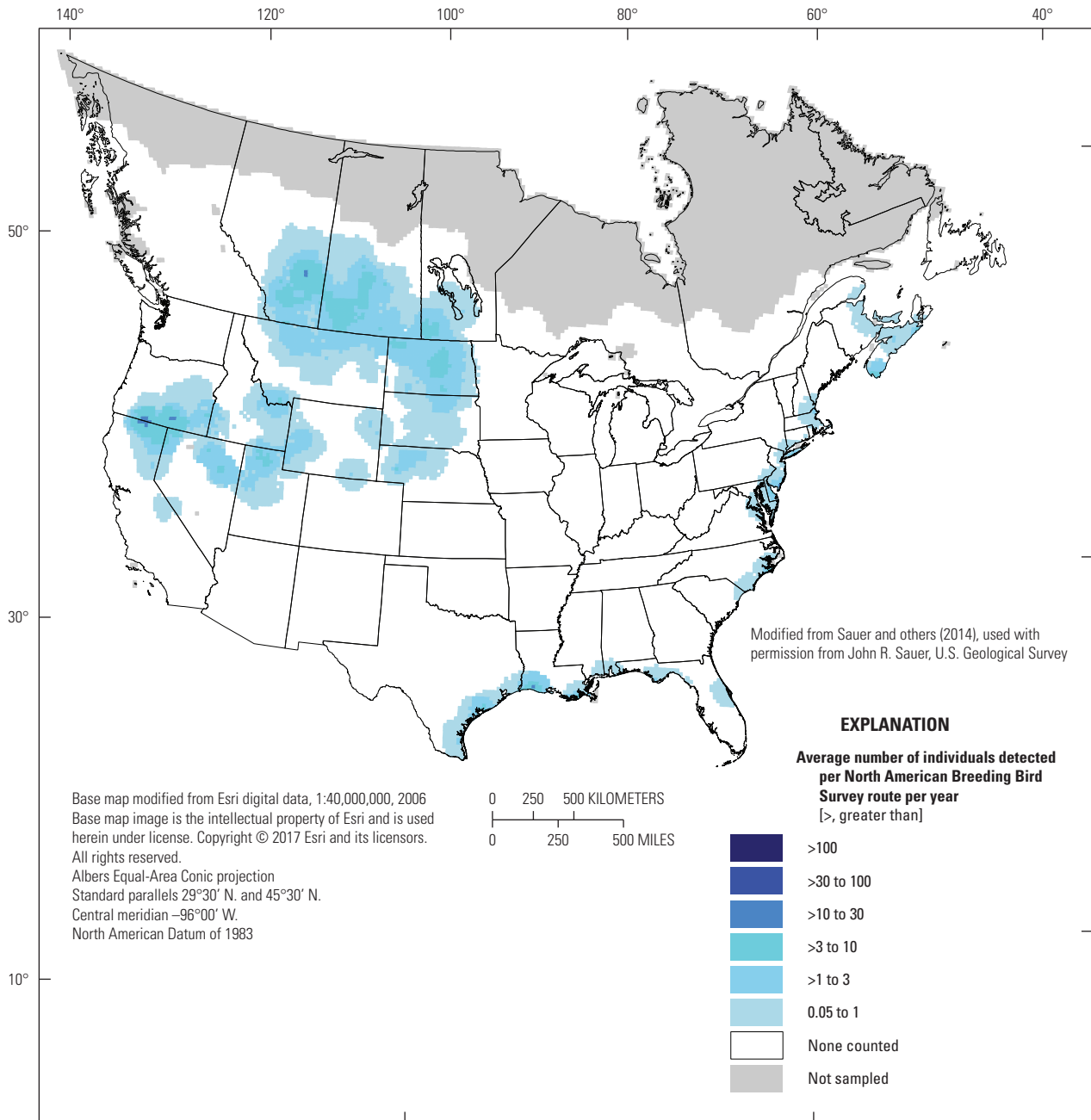


Figure 11. Breeding distribution of the Willet (*Tringa semipalmata*) in the United States and southern Canada, based on North American Breeding Bird Survey (BBS) data, 2008–12. The BBS abundance map provides only an approximation of breeding range edges.

grass cover than do breeding pairs during nesting (Ryan and Renken, 1987). In North Dakota, uplands used by Willets had a thinner litter layer than surrounding areas (Renken, 1983). Willets often nest near a conspicuous object, such as a piece of wood, dried cattle dung, or a stone (Higgins and others, 1979; Kantrud and Higgins, 1992).

Willets prefer native grass to tame vegetation (Stewart, 1975; Ryan and Renken, 1987; Eldridge, 1992; Kantrud and Higgins, 1992). In the prairie and aspen parkland regions of Alberta, mean number of birds per site was compared among several habitats (Prescott and others, 1995; Prescott, 1997). In the prairie region, Willets were most abundant in native

mixed-grass prairies, followed by coulees (that is, a valley containing an ephemeral creek or seepage that may contain other undescribed habitat types), upland shrublands, planted croplands, and haylands (planted to grasses [species not given] or alfalfa [*Medicago sativa*]) (Prescott, 1997). In the uplands of aspen parkland, Willets were most abundant in deferred native pastures grazed after July 15, followed by idle native grasslands, continuously grazed native parklands, and tame dense nesting cover (DNC) (Prescott and others, 1995). The species was not found in tame pastures, deferred tame pastures, idle tame uplands, idle tame grasslands, continuously grazed native grasslands, idle parklands, or native DNC. In North Dakota, the species was absent from tame DNC (Renken, 1983).

In Manitoba, Montana, North Dakota, and South Dakota, Willets nested in native and tame grasslands, pastures, hayland, and idle or burned areas (Kantrud and Higgins, 1992). Dominant vegetation at nest sites included green needlegrass (*Nassella viridula*), short sedges (*Carex* species [spp.]), and Kentucky bluegrass (*Poa pratensis*). In North Dakota, nesting pairs used upland sites characterized by short (less than 15 cm) vegetation (Ryan and Renken, 1987). The mean cover percentages at 15 nests were 77.7 percent for vegetation less than (<) 15 cm tall, 7.8 percent for vegetation 15–60 cm tall, and 0.7 percent for vegetation greater than (>) 60 cm tall. In another North Dakota study, the species nested in dry uplands in vegetation with low height-density (<6 cm) (Sedivec, 1994). Although tilled lands usually are avoided (Weber, 1978; Eldridge, 1992), nests have been reported in hayland and cropland, including small-grain, common flax (*Linum usitatissimum*), and stubble fields (Higgins and others, 1979; Kantrud and Higgins, 1992). In a North Dakota study, pairs nesting in native vegetation had higher apparent hatching success than pairs nesting in cultivated fields (Higgins and others, 1979).

In wetlands, Willets avoid dense, emergent vegetation, preferring shallow-water areas with short, sparse shoreline vegetation (Weber, 1978; Ryan and Renken, 1987; Colwell and Oring, 1988a; Eldridge, 1992; Lowther and others, 2001). In Alberta, Willets were present in areas with shallow water, a condition that was provided by created wetlands but not natural wetlands, which were dry during the study (Gratto-Trevor, 1999). In another Alberta study, Willet abundance increased by 25 percent within 0.94 kilometer (km) of wetland edges (Sliwinski and Koper, 2012). In Saskatchewan, Willets waded in shallow water within 10 meters (m) of the wetland edge (Colwell and Oring, 1988a).

Suitable wetlands range from fresh to saline and vary widely in size and permanence (Stewart and Kantrud, 1965; Stewart, 1975; Kantrud and Stewart, 1984; Ryan and Renken, 1987; Eldridge, 1992; Prescott and others, 1995; Lowther and others, 2001). In Alberta, Willets were most abundant in large saline wetlands and rare in small freshwater wetlands; the species also was found in large and medium freshwater and medium saline wetlands (Prescott and others, 1995). In North Dakota, the highest densities of Willets were found in brackish and saline semipermanent potholes with closed stands of

emergent cover, with clumps of emergent cover interspersed with open water, or with peripheral bands of emergent cover encircling expanses of open water (Stewart and Kantrud, 1965). In another North Dakota study, the highest densities of Willets were found in brackish and subsaline semipermanent ponds and lakes (Stewart, 1975). In that same study, 47 percent of 219 breeding pairs of Willets were recorded using semipermanent wetlands, 43 percent seasonal wetlands, 4 percent permanent wetlands, 3 percent alkali wetlands, and 3 percent intermittent streams (Stewart, 1975). In a third North Dakota study, semipermanent wetlands were used most often by Willets in North Dakota, but seasonal, temporary, ephemeral, and alkali ponds were preferred relative to their availability (Ryan and Renken, 1987). In a study of 1,190 wetlands throughout the Prairie Pothole Region (PPR) of North Dakota and South Dakota, Willets were observed in natural and restored wetlands, and were observed in a higher proportion of the alkali or permanent wetlands than in the semipermanent, seasonal, or temporary wetlands (Igl and others, 2017). Willets were observed in 78 wetlands characterized by an average of 52 percent open water, 22 percent wet-meadow vegetation, 20 percent emergent vegetation, and 6 percent shore/mudflat (Igl and others, 2017). In South Dakota, Willets occurred in many types of wetlands, including permanent, semipermanent, seasonal, temporary, and ephemeral ponds; stock ponds; dugouts; and intermittent streams. Willets were absent from permanent streams, ponds in tilled fields, drainage and road ditches, oxbows, and gravel pits (Weber, 1978; Weber and others, 1982).

The use by spring-migrating Willets of temporary and seasonal wetlands within agricultural fields in the Drift Prairie of eastern North Dakota was evaluated by Niemuth and others (2006). Willets were more likely to use temporary than seasonal wetlands and to use wetlands with sparse amounts of tall, emergent vegetation than wetlands with dense stands of tall, emergent vegetation; presence of Willets was positively related to the number of times a wetland basin contained water during 9 or 10 visits. The species was less likely to use wetlands showing evidence of a drainage history. In the PPR of eastern Montana, North Dakota, and South Dakota, Niemuth and others (2012) evaluated wetland characteristics that influenced the detection and number of Willets. Detections of Willets were negatively related to excavated, ditched wetlands and positively related to brackish wetlands. Detections increased as the proportion of wetlands surrounded by a grass buffer increased. Moreover, detections were positively related to characteristics indicative of wetlands with low amounts of emergent vegetation: the amount of open water or bare soil covering >95 percent of the wetland area, the proportion of wetland inundated by water, and the width of mudflats. Detections were negatively related to wetlands characterized by scattered-to-closed stands of tall (>25 cm) emergent and with open water or bare soil covering less than or equal to 95 percent of the wetland area. However, in a study in North Dakota and South Dakota, Willet detections were positively associated with wetlands characterized by scattered-to-open water and

with open water or bare soil covering from 5 to >95 percent of the wetland area (Niemuth and others, 2013).

Seasonal shifts in habitat use between wetlands and uplands may occur, as Niemuth and others (2012) reported that the detections of Willets over a 7-week survey period spanning mid-May to late June were initially high in upland habitats but decreased with concomitant increases in wetland habitats. Shifts in wetland use occur seasonally and during climatic extremes (Ryan and Renken, 1987; Gratto-Trevor, 1999, 2006). In North Dakota, semipermanent wetlands were used more often later in the summer than other wetland types, and semipermanent and permanent wetlands were used during drought years (Ryan and Renken, 1987).

Area Requirements and Landscape Associations

Willet territories are large and include feeding and nesting areas. Areas must be large enough to provide upland habitat and a diversity of wetland types (Kantrud and Stewart, 1984; Ryan and Renken, 1987; Colwell and Oring, 1988a). In North Dakota, mean territory size was 44.3 hectares (ha) (Ryan and Renken, 1987). Of 78 wetlands in the PPR of North Dakota and South Dakota in which Willets were observed, average wetland size was 15 ha (Igl and others, 2017). In South Dakota, the presence of the species was positively associated with area of surface water (Weber, 1978). In the PPR of Montana, North Dakota, and South Dakota, Niemuth and others (2012) reported that the detections of Willets exhibited a curvilinear relationship with area of wetland surveyed; in North Dakota and South Dakota, detections increased curvilinearly with wet area of basin (Niemuth and others, 2013).

At a landscape level, detections of Willets in the PPR of Montana, North Dakota, and South Dakota were positively related to the percentage of grassland within 800 m of a survey point consisting of native grassland, forb, and scattered low shrubs; and to percentage of area within 800 m consisting of temporary, seasonal, semipermanent, and permanent wetlands (Niemuth and others, 2012). The number of Willets was positively related to the number of different water regimes for wetlands (that is, number of wetland types) in the landscape. In North Dakota's Drift Prairie, presence of spring-migrating Willets was positively related to the percentage of the landscape within 800 m of sampled wetlands that was occupied by semipermanent and permanent wetlands (Niemuth and others, 2006). In a survey of 1,190 wetlands in the PPR of North Dakota and South Dakota, Igl and others (2017) recorded Willets in 78 wetlands, where landscape composition within 800 m of the wetlands in which Willets were observed was 58 percent grassland, 21 percent agricultural, 17 percent wetland, and 4 percent other; average number of wetlands within 800 m was 22 (Igl and others, 2017). In South Dakota, the presence of the species was negatively associated with area of land under cultivation (Weber, 1978).

Brood Parasitism by Cowbirds and Other Species

The Willet is an unsuitable host of the Brown-headed Cowbird (*Molothrus ater*), and there are no known records of cowbird brood parasitism (Shaffer and others, 2019). Willet nests have been parasitized by Long-billed Curlews (*Numenius americanus*) (Sugden, 1947).

Breeding-Season Phenology and Site Fidelity

Willetts arrive on Saskatchewan breeding grounds from late April to mid-May and depart from mid-August to early September (Maher, 1974). In South Dakota, peak spring migration occurs from April 21 to May 10, and peak fall migration occurs from August 1 to August 15 (Gardner and others, 2008). In the northern Great Plains, Willets breed from early May through late July, with broods present from about early June until late July (Stewart, 1975; Kantrud and Higgins, 1992; Sedivec, 1994; Gardner and others, 2008). Ryan and others (1981) reported two cases of renesting after initial nests were destroyed. In Saskatchewan, adults of both sexes and yearling females exhibited breeding-site fidelity (Colwell and Oring, 1988b).

Species' Response to Management

Little information is available on the effects of prescribed burning or haying on Willets. Ryan and Renken (1987) recommended burning, mowing, or grazing of upland and wetland habitat to maintain the short, sparse vegetation and thin litter layer preferred by Willets. Ryan and others (1984) indicated that fall burning can provide dense, taller regrowth (15–60 cm) later in the summer, which may benefit broods because they use vegetation >15 cm in height (Ryan and Renken, 1987). In a North Dakota grassland study, Willet densities were unrelated to time since burning (Johnson, 1997). In another North Dakota study, Willets did not use idle areas even after they were hayed the previous year (Messmer, 1990).

Grazed uplands appear to be more attractive to breeding Willets than idle grasslands (Messmer, 1985, 1990; Renken and Dinsmore, 1987; Sedivec, 1994; Lowther and others, 2001; Gardner and others, 2008), although Kantrud and Higgins (1992) indicated that the species prefers pastures that are idle during the nesting season and, to a lesser extent, actively grazed pastures than other land-use types. In Alberta, Willet densities were higher (not statistically tested) on deferred-grazed native pastures than on native pastures grazed in early summer, but the species was not present on continuously grazed native pastures (Prescott and Wagner, 1996). However,

Willetts were present on tame pastures of crested wheatgrass (*Agropyron cristatum*) grazed in spring from late April to mid-June. In Saskatchewan, Willetts were observed on grazed and ungrazed areas (Dale, 1984). In North Dakota, Willetts were more common in grazed areas than in ungrazed areas (Sedivec, 1994). In North Dakota, densities of breeding Willetts were significantly higher on twice-over deferred (grazing a number of pastures twice per season, with about a 2-month rest between grazing) grazing systems than on season-long (leaving cattle on the same pasture for the entire growing season) or short-duration (pastures rotated through a grazing schedule of about 1 week grazed and 1 month ungrazed, repeated throughout the growing season [usually late May or early June until October]) grazing systems or on idle pastures (Messmer, 1990). The density of Willetts decreased as vegetative cover increased on a managed pasture. The twice-over deferred pastures consisted of silty range, thin upland range, and shallow-to-gravel range sites (Messmer, 1990; Sedivec, 1994). Silty range and thin upland range sites were characterized by thin topsoil, loamy soil, 1–25 percent slope, grassy cover, low shrub cover, and moderate-to-high litter cover. Shallow-to-gravel range sites were characterized by sparse cover and reduced litter. In south-central North Dakota mixed-grass prairies, Willetts only occurred in heavily and extremely grazed pastures (20–35 percent of forage produced in an average year remained, equating to an average grazing rate of 4.2–6.8 animal unit months per ha) and were not recorded in lightly or moderately grazed pastures (50–65 percent, 1.1–2.4 animal unit months per ha) (Salo and others, 2004).

Willetts appear to be fairly tolerant of anthropogenic activity associated with energy development, roads, and agriculture. Niemuth and others (2013) examined the influence of two wind facilities in North Dakota and South Dakota on Willetts for 3 years. The species did not appear to avoid wetland basins <805 m from wind turbines at either facility, although occurrence was slightly and consistently lower at one facility, possibly because that facility was located primarily in cropland and the other facility in grassland. In Alberta, Sliwinski and Koper (2012) detected no effect of cropland or road edges on Willet abundance.

Management Recommendations from the Literature

Protection of wetland complexes is important because Willetts use a variety of wetland sizes and types with various salinity levels (Kantrud and Stewart, 1984; Ryan and others, 1984; Ryan and Renken, 1987; Colwell and Oring, 1988a; Lowther and others, 2001; Niemuth and others, 2006, 2012). During migration, numerous small wetlands may be of equal

or greater importance to Willetts than a few large wetlands (Niemuth and others, 2006), but larger, more permanent wetlands provide vital habitat during droughts or in late summer (Ryan and Renken, 1987; Prescott and others, 1995; Niemuth and others, 2006).

Willetts will benefit from protecting wetlands from drainage (Ryan and others, 1984; Ryan and Renken, 1987; Lowther and others, 2001; Niemuth and others, 2006) and from restoring drained wetlands (Berkey and others, 1993; Johnson, 1996). Construction of wetlands may be especially beneficial to Willetts in locations where natural wetlands tend to go dry in the summer; Willetts in Alberta were more abundant in managed wetland basins than in natural wetland basins because natural basins usually lacked water by late summer (Gratto-Trevor, 1999, 2006).

Ryan and Renken (1987) emphasized the importance of providing wetland and grassland habitats that are large enough to support Willet territories, which averaged 44.3 ha in North Dakota. Willetts were not found in small (<100 ha) blocks of wetland and grassland habitat (Ryan and Renken, 1987). Areas also must be large enough to provide grassland habitat and a diverse range of wetland types and sizes (Stewart, 1975; Kantrud and Stewart, 1984; Ryan and Renken, 1987; Colwell and Oring, 1988a; Kantrud and Higgins, 1992). Native grassland habitat for upland nesting and foraging should be provided (Ryan and Renken, 1987; Eldridge, 1992; Kantrud and Higgins, 1992).

Burning, mowing, and grazing can be used to provide areas of shorter, sparser vegetation in uplands and wetlands (Kantrud and Stewart, 1984; Messmer, 1985; Ryan and Renken, 1987; Eldridge, 1992; Berkey and others, 1993). Fall burning or mowing of upland sites and wetland edges can produce suitable cover during the following spring (Ryan and others, 1984). Moderate-to-dense regrowth in burned areas may be too dense for nesting but may provide the denser, taller cover used by broods (Ryan and others, 1984).

Twice-over deferred grazing is preferable to season-long grazing (Messmer, 1985, 1990; Sedivec, 1994). Berkey and others (1993) indicated that short-term grazing (2–4 weeks in May) may be beneficial to Willetts in North Dakota. Willetts prefer previously grazed areas that are idle during the current breeding season (Kantrud and Higgins, 1992). To benefit Willetts and other nesting grassland birds, the timing of grazing in spring may depend on the grazing regime; Sedivec (1994) recommended delaying grazing until late May to early June when implementing a rotational grazing system, and until mid-June when implementing season-long grazing.

Agriculture and mechanical disturbances on cropland may negatively affect Willetts during the breeding season. Willetts will benefit from the protection of grasslands from conversion to agriculture (Ryan and others, 1984; Ryan and Renken, 1987) and the adoption of no-tillage and minimum-tillage practices on cropland (Kantrud and Higgins, 1992).

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Table 11. Measured values of vegetation structure and composition in Willet (*Tringa semipalmata inornata*) breeding habitat by study. The parenthetical descriptors following authorship and year in the “Study” column indicate that the vegetation measurements were taken in locations or under conditions specified in the descriptor; no descriptor implies that measurements were taken within the general study area.

[cm, centimeter; %, percent; <, less than; > greater than; --, no data; DNC, dense nesting cover]

Study	State or province	Habitat	Management practice or treatment	Vegetation height (cm)	Vegetation height-density (cm)	Grass cover (%)	Forb cover (%)	Shrub cover (%)	Bare ground cover (%)	Litter cover (%)	Litter depth (cm)
Higgins and others, 1979 (nests)	North Dakota	Mixed-grass prairie	Multiple	<15	--	--	--	--	--	--	--
Kantrud and Higgins, 1992 (nests)	Manitoba, Montana, North Dakota, South Dakota	Multiple	Multiple	--	4 ^a , 11 ^b	--	--	--	--	38 ^c	--
Messmer, 1990	North Dakota	Mixed-grass prairie	Multiple	50–70	--	--	--	--	--	--	3.8–9.1
Renken, 1983 ^d	North Dakota	Tame grassland (DNC)	Idle, grazed	--	8 ^a	55.1	20.5	3.9	0.6	98.7	1.8
Ryan and Renken, 1987 (nests)	North Dakota	Mixed-grass prairie	--	<15	--	--	--	--	14.5	--	--
Ryan and Renken, 1987 (broods)	North Dakota	Mixed-grass prairie	--	>15	--	--	--	--	--	--	--
Salo and others, 2004	North Dakota	Mixed-grass prairie	Heavy grazing intensity	27.1 ^e	22.9 ^a	--	--	--	--	--	2
Salo and others, 2004	North Dakota	Mixed-grass prairie	Extreme grazing intensity	17.5 ^e	7.9 ^a	--	--	--	--	--	0.9
Sedivec, 1994 (nests)	North Dakota	Mixed-grass prairie	Multiple	--	5.8 ^a	--	--	--	--	--	--

^aVisual obstruction reading (Robel and others, 1970).

^bEffective vegetation height.

^cStanding dead vegetation.

^dThe sum of percentages is >100%, based on the modified point-quadrat technique of Wiens (1969).

^eMean grass height.

The Effects of Management Practices on Grassland Birds—Wilson’s Phalarope (*Phalaropus tricolor*)

By Jill A. Shaffer,¹ Lawrence D. Igl,¹ Douglas H. Johnson,¹ Christopher M. Goldade,^{1,2} Amy L. Zimmerman,¹ and Betty R. Euliss¹

Chapter J of

The Effects of Management Practices on Grassland Birds

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Figure

J1. Map showing breeding distribution of the Wilson's Phalarope (<i>Phalaropus tricolor</i>) in the United States and southern Canada, based on North American Breeding Bird Survey data, 2008–12.....	2
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Table

J1. Measured values of vegetation structure and composition in Wilson's Phalarope (<i>Phalaropus tricolor</i>) breeding habitat by study.....	10
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Conversion Factors

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
hectare (ha)	0.003861	square mile (mi ²)
square kilometer (km ²)	0.3861	square mile (mi ²)
Mass		
microgram (µg)	0.0000003527	ounce (oz)
gram (g)	0.03527	ounce (oz)

Abbreviations

CRP	Conservation Reserve Program
DNC	dense nesting cover
PPR	Prairie Pothole Region
spp.	species (applies to two or more species within the genus)

Acknowledgments

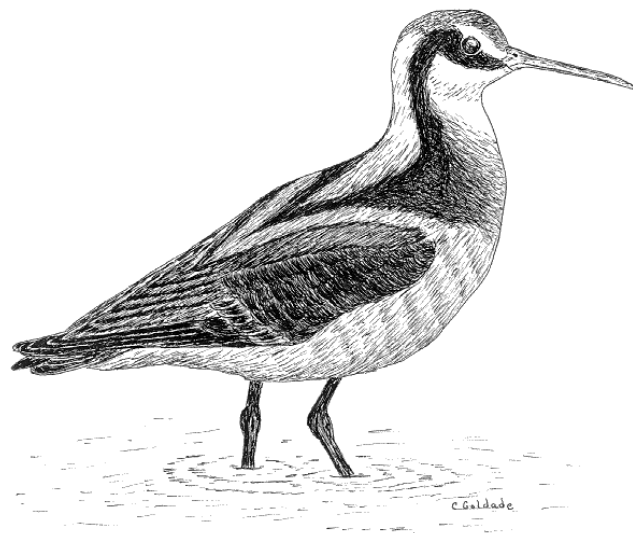
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The Effects of Management Practices on Grassland Birds—Wilson’s Phalarope (*Phalaropus tricolor*)

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Capsule Statement

The key to Wilson’s Phalarope (*Phalaropus tricolor*) management is providing wetland complexes containing suitable wetland characteristics (that is, open water, emergent vegetation, and open shoreline) and upland habitat (native grassland or tame hayland) throughout the breeding season. Wilson’s Phalaropes have been reported to use habitats with 15–32 centimeters (cm) average vegetation height, 8–18 cm visual obstruction reading, 45–53 percent grass cover, 19–22 percent forb cover, and less than (<) 3 cm litter depth. The descriptions of key vegetation characteristics are provided in table J1 (after the “References” section). Vernacular and scientific names of plants and animals follow the Integrated Taxonomic Information System (<https://www.itis.gov>).



Wilson’s Phalarope. Illustration by Christopher M. Goldade, U.S. Geological Survey.

Breeding Range

Wilson’s Phalaropes breed from the southern Yukon Territories through British Columbia, south-central Alberta and southern Manitoba; south to central California, southern Nevada, southern Colorado, northern New Mexico, and northern Texas; and east to central Kansas, northwestern Iowa, and northwestern Minnesota. They also breed from eastern Wisconsin and northeastern Illinois, east to Michigan, northern Ohio, eastern Ontario, and northern New York (National Geographic Society, 2011). The relative densities of Wilson’s Phalarope in the United States and southern Canada, based on North American Breeding Bird Survey data (Sauer and others, 2014), are shown in figure J1 (not all geographic places mentioned in report are shown on figure).

Suitable Habitat

Wilson’s Phalaropes use fresh and alkali wetlands with three characteristics: open water, emergent vegetation, and open shoreline (Saunders, 1914; Stewart and Kantrud, 1965; Hohn, 1967; Stewart, 1975; Prescott and others, 1995; Naugle, 1997). The species uses a wide variety of habitats, including wetlands, wet meadows, upland grasslands, standing stubble, and road rights-of-way (Roberts, 1932; Bent, 1962; Hohn, 1967; Higgins, 1975; Stewart, 1975; Murray, 1983; Bomberger, 1984; Colwell, 1987; Colwell and Oring, 1990; Einemann, 1991; Faanes and Lingle, 1995; Dinsmore and Schuster, 1997; Naugle and others, 2001). Wilson’s Phalaropes occasionally inhabit Conservation Reserve Program (CRP) fields, dense nesting cover (DNC), and Wildlife Management Areas (Svedarsky, 1992; Johnson and Schwartz, 1993; Prescott and others, 1993; Wiens, 2007). In Alberta, the species used grassy areas free of cattails (*Typha* species [spp.]) and sedges (*Carex* spp.) (Hohn, 1967). In the Midwest region of the United

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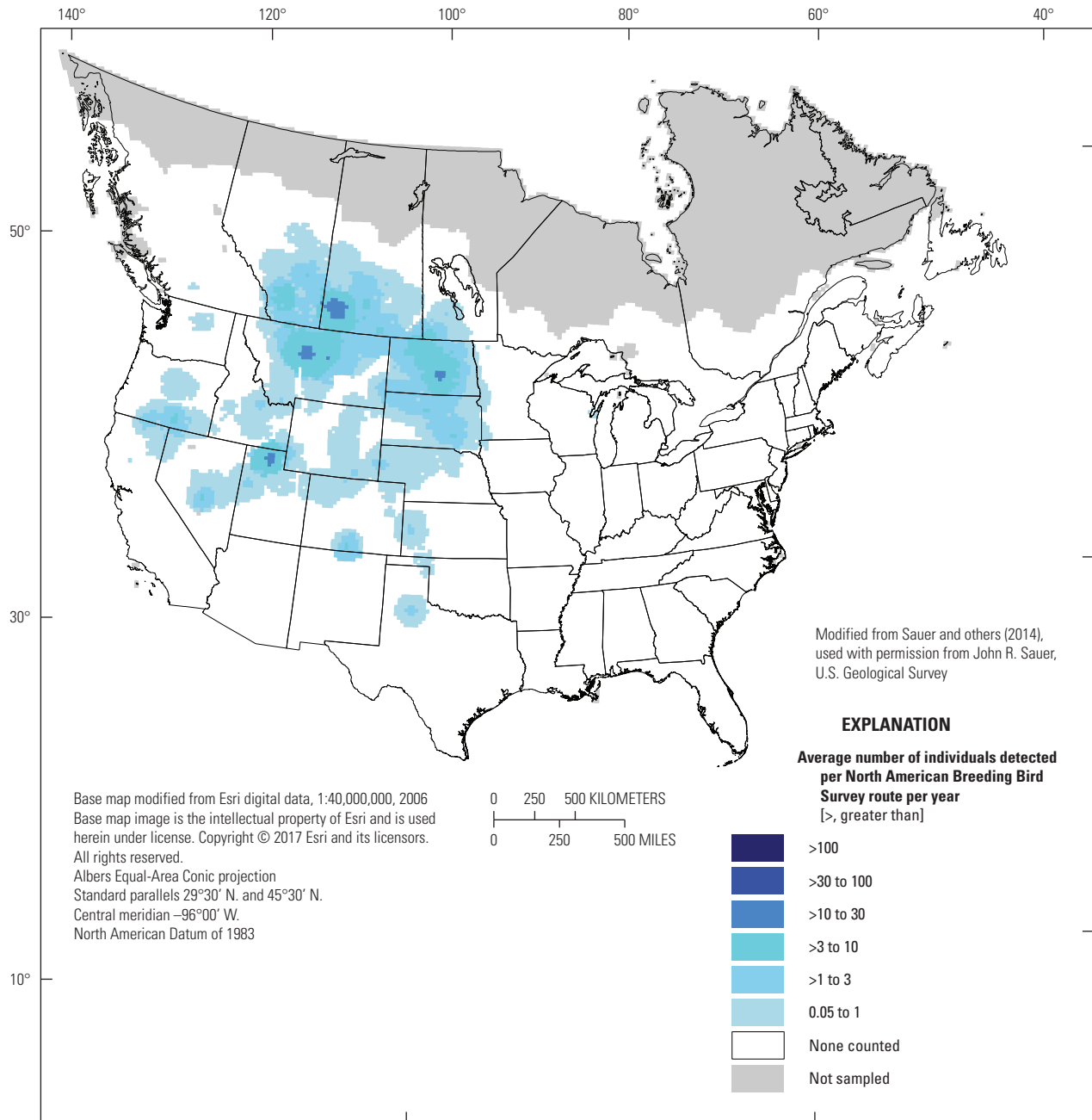


Figure J1. Breeding distribution of the Wilson’s Phalarope (*Phalaropus tricolor*) in the United States and southern Canada, based on North American Breeding Bird Survey (BBS) data, 2008–12. The BBS abundance map provides only an approximation of breeding range edges.

States, the species occurred in the wet-meadow zones of permanent or semipermanent wetlands and foraged in open water at depths up to about 30 cm (Eldridge, 1992).

Suitable wetland types range from fresh to saline and vary widely in size and permanence (Stewart and Kantrud, 1965; Stewart, 1975; Kantrud and Stewart, 1984). In the Prairie Pothole Region (PPR) of eastern Montana, North Dakota, and South Dakota, Niemuth and others (2012) evaluated wetland characteristics that affected the detection and the number of Wilson’s Phalaropes; the species was more likely

to be observed, and observed at higher numbers, on brackish or saline wetlands than on freshwater wetlands. In a study of wetlands throughout the PPR of North Dakota and South Dakota, Wilson’s Phalaropes were more likely to be observed in alkali or permanent wetlands than in temporary, seasonal, or semipermanent wetlands (Igl and others, 2017). Within the North Dakota PPR, Wilson’s Phalarope densities were highest in wetlands with frequently tilled soils, followed by temporary, seasonal, semipermanent, fen, alkali, and permanent wetlands (Kantrud and Stewart, 1984). Wilson’s Phalaropes often

occupied the peripheral low-prairie and wet-meadow areas of most classes of wetlands in North Dakota. In another North Dakota study, the highest densities of Wilson's Phalaropes were found on seasonal wetlands with closed stands of emergent cover, such as common spikerush (*Eleocharis palustris*), or with clumps of emergent cover interspersed with open water (Stewart and Kantrud, 1965). High densities also were found on brackish or saline semipermanent wetlands with closed stands of emergent cover, with clumps of emergent cover interspersed with open water, or with peripheral bands of emergent cover encircling expanses of open water. The use by spring-migrating Wilson's Phalaropes of temporary and seasonal wetlands within agricultural fields in the Drift Prairie of eastern North Dakota was evaluated by Niemuth and others (2006). Wilson's Phalaropes were more likely to use seasonal and undrained wetlands than temporary or drained wetlands, and to use wetlands with sparse amounts of tall, emergent vegetation than densely vegetated wetlands with tall, emergent vegetation; presence of Wilson's Phalaropes was positively related to the number of times a wetland basin contained water during 9 or 10 visits. In South Dakota, Wilson's Phalaropes most frequently occurred on temporary and seasonal wetlands and on stock ponds, but were also seen on ephemeral and semipermanent wetlands, intermittent streams, dugouts, and wetlands with tilled soil bottoms; no phalaropes were seen on permanent streams (Weber and others, 1982). In Colorado, Wilson's Phalaropes preferred seasonal wetlands and habitats dominated by baltic rush (*Juncus balticus*), sedges, and grasses <40 cm tall more than semipermanent wetlands, habitats dominated by cattail (*Typha* spp.) and softstem bulrush (*Schoenoplectus tabernaemontani*) greater than (>) 40 cm tall, inland saltgrass (*Distichlis spicata*) habitats, or upland shrub habitats (Laubhan and Gammonley, 2000; Gammonley and Laubhan, 2002).

Wilson's Phalarope occurrence and abundance are influenced by local wetland characteristics. Niemuth and others (2012) reported that detections and number of Wilson's Phalaropes were positively related to the proportion of wetland inundated by water and negatively related to certain wetland cover classes that characterized the amount and configuration of emergent vegetation. Detections and number of Wilson's Phalaropes were negatively related to the full spectrum of cover classes, from the one extreme indicating wetlands with closed stands of tall (>25 cm) emergent vegetation and with no open water or bare soil, to the other extreme of wetlands entirely inundated with water and with no emergent vegetation or bare soil. Detections were positively related to the percentage of shoreline with an upland vegetative buffer greater than or equal (\geq) to 25 meters (m) wide and to width of mudflats. In a study in North Dakota and South Dakota, Wilson's Phalarope probability of occurrence was positively associated with wetland cover classes characterized by scattered-to-open water and with open water or bare soil covering anywhere from 5 to >95 percent of the wetland area (Niemuth and others, 2013). Probability of occurrence increased curvilinearly with wet area of basin. In a survey of 1,190 wetlands in the PPR of North

Dakota and South Dakota, Igl and others (2017) recorded phalaropes in 97 wetlands that were characterized as having an average of 45 percent open water, 25 percent emergent vegetation, 25 percent wet meadow, and 5 percent shore/mudflat. In South Dakota, the occurrence of Wilson's Phalaropes was positively associated with the area of surface water within a wetland basin and the percentage of grazed shoreline (Weber, 1978). In eastern South Dakota, the probability of occurrence of Wilson's Phalaropes in semipermanent wetlands was positively related to the number of emergent hydrophyte species (for example, willow [*Salix* spp.]) constituting ≥ 10 percent of the vegetated wetland area (Naugle, 1997; Naugle and others, 2001). The probability of occurrence of Wilson's Phalaropes in semipermanent and seasonal wetlands was negatively related to wetlands dominated by thick-stemmed plants (for example, cattail and river bulrush [*Bolboschoenus fluviatilis*]). In a second study in eastern South Dakota, the occurrence of Wilson's Phalaropes was negatively related to wetlands surrounded by trees (Naugle and others, 1999).

Wilson's Phalaropes nest in upland habitats near wetlands, typically <100 m from shoreline (Hohn, 1967; Hatch, 1971; Bomberger, 1984; Colwell and Oring, 1990; Eldridge, 1992). In Alberta, Saskatchewan, and North Dakota, Wilson's Phalaropes nested in grasses of various heights on islands or in wet-meadow zones around lakes and wetlands; in Saskatchewan, brood rearing occurred in patches of foxtail barley (*Hordeum jubatum*) (Bent, 1962; Hohn, 1967; Kagarise, 1979; Colwell, 1987). In another Saskatchewan study, the species nested in heavily grazed uplands with patches of western snowberry (*Symphoricarpos occidentalis*) (Colwell, 1987). In Saskatchewan, nest sites had taller, denser, and more homogeneous vegetation and less bare ground than randomly selected sites (Colwell and Oring, 1990). Nest success (that is, nests that hatched at least one chick) and the number of days that clutches survived were not related to vegetation concealment at nests (Colwell, 1992). In Manitoba, Montana, North Dakota, and South Dakota, the species nested in areas with >50 percent litter and with low, sparse cover, and they avoided areas with 100 percent visual obstruction at ≥ 20 cm or effective vegetation height >46 cm (Kantrud and Higgins, 1992). Dominant vegetation at nests included Kentucky bluegrass (*Poa pratensis*), needlegrass (*Achnatherum* spp. and *Stipa* spp.), wheatgrass (formerly *Agropyron* spp.), sedges, baltic rush, northern reedgrass (*Calamagrostis stricta* spp. *inexpansa*), and inland saltgrass. In North Dakota, Wilson's Phalaropes selected nest sites in short, sparse vegetation in native grasslands (Wiens, 2007). Nest sites had less litter and forb coverage than randomly selected sites and visual obstruction was lower in DNC fields, but higher in grazed pastures, than randomly selected sites. The species appeared to avoid potential nest sites dominated by invasive plant species. In North Dakota and Iowa, Wilson's Phalaropes nested in wetlands associated with river flood plains (Murray, 1983; Koenig, 1984). In Iowa, Wilson's Phalaropes nested on a small mound of vegetation near a wetland (Dinsmore and Schuster, 1997). In Nebraska, the species nested near a saline wetland in a stand of foxtail barley

(Einemann, 1991) and in wet sedge meadows (Faanes and Lingle, 1995). In the Nebraska sandhills, nest sites had shorter vegetation than random sites (Bomberger, 1984). The species nested near wet-meadow zones around wetlands; vegetation height was more important in nest-site selection than was distance to water. Average distance from nests to wetlands ranged from 4.2 to 4.6 m, and nearest wetlands to nests ranged from an average of 32 to 44 hectares (ha) in basin surface area.

Nest-site selection varies seasonally; Wilson's Phalaropes nest in upland vegetation early in the breeding season and wet-meadow vegetation later in the season (Colwell and Oring, 1990). They also exhibit annual variation in nest-site selection, moving to deeper, more permanent wetlands in dry years (Hohn, 1967; Colwell, 1991).

Area Requirements and Landscape Associations

Wilson's Phalaropes may be area sensitive and be sensitive to features of the landscape surrounding suitable habitat. In the northern Great Plains, Wilson's Phalaropes were absent in patches of CRP grassland that were <50 ha (Johnson and Igl, 2001). Highest abundances occurred in large (>8 ha) wetlands (Prescott and others, 1995). Of 97 wetlands in the PPR of North Dakota and South Dakota in which Wilson's Phalaropes were observed, average wetland size was 13 ha (Igl and others, 2017). Landscape composition within 800 m of these wetlands was 56 percent grassland, 20 percent agricultural, 19 percent wetland, and 5 percent other; average number of wetlands within 800 m of these wetlands was 25. At a landscape level, Wilson's Phalarope occurrence in the PPR of eastern Montana, North Dakota, and South Dakota was positively related to the percentage of grassland within 800 m of a survey point consisting of native grassland, forb, or scattered low shrubs; to percentage of area within 800 m consisting of temporary, seasonal, semipermanent, and permanent wetlands; and to wetland perimeter (Niemuth and others, 2012). The number of Wilson's Phalaropes was curvilinearly related to wetland perimeter and negatively related to the number of different water regimes for wetlands (that is, number of wetland types) in the landscape. The use by spring-migrating Wilson's Phalaropes of temporary and seasonal wetlands within agricultural fields in the Drift Prairie of North Dakota was evaluated by Niemuth and others (2006). Wilson's Phalaropes selected wetlands with large perimeters and their presence was positively associated with the percentage of the landscape within 800 m of sampled wetlands that was occupied by semipermanent and permanent wetlands. In South Dakota, the occurrence of Wilson's Phalaropes was positively associated with the area of alfalfa (*Medicago sativa*) hayland within each quarter section (that is, 0.65 ha) surrounding focal wetlands (Weber, 1978). In eastern South Dakota, the probability of occurrence of Wilson's Phalaropes in semipermanent wetlands

was positively related to the proportion of untilled uplands near nesting wetlands (Naugle, 1997).

In North Dakota tallgrass prairies, occurrence of Wilson's Phalarope was positively associated with wetland and grass cover and negatively associated with woodland cover at the 100-m scale (Cunningham and Johnson, 2006). Occurrence was negatively associated with tree cover at the 1,600-m scale. In stock ponds in western South Dakota, abundance of Wilson's Phalaropes was 3.5 times greater in 25.9 square kilometer (km²) landscapes dominated by grasslands than in landscapes dominated by cropland (May and others, 2002). Grassland landscapes contained <5 percent cropland compared to intensively farmed landscapes that contained >75 percent cropland. In semipermanent and seasonal wetlands in South Dakota, Wilson's Phalarope occurrence was positively related to the area of wetland and grassland within 25.9 km² landscapes surrounding surveyed wetlands (Naugle and others, 2001).

Brood Parasitism by Cowbirds and Other Species

The Wilson's Phalarope is an accidental and unsuitable host of the Brown-headed Cowbird (*Molothrus ater*) (Friedmann, 1963; Hatch, 1971). The few records of parasitism are summarized in Shaffer and others (2019). In North Dakota, none of 21 nests were parasitized (M. Winter, WissenLeben e.V., Raisting, Germany, and D.H. Johnson, unpub. data). In Saskatchewan, 1 percent of 386 nests were parasitized by Brown-headed Cowbirds (Colwell and Jehl, 1994). Two records of multiple parasitism have been reported (Friedmann, 1963); Williams and Trowbridge (1939) found two parasitized Wilson's Phalarope nests, each containing four Wilson's Phalarope eggs and two Brown-headed Cowbird eggs.

Breeding-Season Phenology and Site Fidelity

In the central and northern Great Plains (Minnesota, Nebraska, and North Dakota), Wilson's Phalaropes arrive on the breeding grounds from mid-April to early May and depart from mid-August to early September (Roberts, 1932; Howe, 1972; Johnsgard, 1980; Murray, 1983). In Alberta, Manitoba, and Saskatchewan, Wilson's Phalaropes arrive on the breeding grounds from late April to early May and remain until early September (Hohn, 1967; Maher, 1974; Reynolds and others, 1986; Colwell, 1987; Colwell and Oring, 1988a, 1988b). Females arrive on the breeding grounds earlier than males (Reynolds and others, 1986; Colwell, 1987), and commonly depart from breeding areas earlier than males, usually from early June to early July (Hohn, 1967; Howe, 1972; Colwell, 1987; Colwell and Oring, 1988a). Wilson's Phalaropes may

renest after nest failure, and females are capable of laying multiple clutches (Colwell and Jehl, 1994). Polyandry was first documented in the species in Saskatchewan, where a color-banded female laid two clutches with two individual males (Colwell, 1986; Colwell, 1987). Philopatry is uncommon in Wilson's Phalaropes, although males return to breeding areas in successive years more often than females (Colwell, 1987; Colwell and Oring, 1988c). Of 154 adult male phalaropes banded over 4 years in Saskatchewan, 16 percent returned to their previous breeding area in successive years, whereas only 2 percent of 69 banded adult females returned (Colwell, 1987).

Species' Response to Management

Wilson's Phalaropes nest in idle, hayed, or grazed grasslands adjacent to wetlands (Hohn, 1967; Kantrud and Higgins, 1992). In North Dakota, Wilson's Phalaropes nested at higher densities in hayland mowed the previous year than in grazed areas (Kantrud, 1981). Idle grasslands and previously grazed areas provided habitat for nesting, but areas with cattle present during the breeding season were less suitable (Renken, 1983; Renken and Dinsmore, 1987; Kantrud and Higgins, 1992). In Alberta, Wilson's Phalaropes were present in deferred-grazed (grazed after July 15) native pasture (Prescott and others, 1993). Nesting occurred in areas that were moderately grazed in Nebraska (Faanes and Lingle, 1995) and heavily grazed in Saskatchewan (Colwell, 1987). In South Dakota stock ponds, grazing by cattle limited the growth of thick-stemmed emergent vegetation that was not conducive for Wilson's Phalarope nesting (May and others, 2002). Although Wilson's Phalaropes occasionally nested in cropland (small-grain stubble) in North Dakota (Higgins, 1975), native grassland was preferred over cropland and tame grassland in southern Canada and the northern United States (Owens and Myres, 1973; Eldridge, 1992; Kantrud and Higgins, 1992). Johnson and Schwartz (1993) reported that Wilson's Phalaropes were present in low numbers in CRP fields in the northern Great Plains (North Dakota, South Dakota, and eastern Montana). In Saskatchewan aspen parkland, Wilson's Phalaropes were observed in DNC that contained wetlands (Prescott and others, 1993, 1995).

Irrigation activities resulting in high mineral concentrations or that require the control of mosquitos (*Culicidae*) can be detrimental to Wilson's Phalaropes. In Wyoming, irrigation over soils with a high selenium content caused leaching of selenium from the soil to the groundwater. Selenium discharge from wetland basins was related to intensity of irrigation (measured by the area of irrigated land) and the concentration of selenium in the groundwater. High selenium levels in lakes appeared to cause high selenium levels in the eggs (>13 micrograms per gram [$\mu\text{g/g}$]) and livers (>30 $\mu\text{g/g}$) of adult Wilson's Phalaropes (See and others, 1992). One dead adult bird had a liver selenium content of >30 $\mu\text{g/g}$, a level associated with biological risk. Average selenium concentrations of >13 $\mu\text{g/g}$ dry weight were associated with embryo deformities. Of

6 eggs collected over 2 years, selenium concentrations ranged from 5 to 19.9 $\mu\text{g/g}$ dry weight and averaged 11.7 $\mu\text{g/g}$ dry weight. Also in Wyoming, mortality of Wilson's Phalarope was observed after fenthion, a chemical used to control mosquitoes, was aerially applied at a rate of 47 grams per hectare to an irrigated meadow (DeWeese and others, 1983). Fenthion is a cholinesterase inhibitor, and activity of brain cholinesterase was significantly lower for 15 days after application in Wilson's Phalaropes collected from treated areas than in Wilson's Phalaropes from control areas.

Niemuth and others (2013) examined the influence of two wind facilities on Wilson's Phalaropes in North Dakota and South Dakota for 3 years. The species did not appear to avoid wetland basins within 805 m of wind turbines at either facility, although occurrence was slightly and consistently lower at one facility, possibly because that facility was located primarily in cropland and the other facility in grassland. In Montana, Malcolm (1982) reported cases of Wilson's Phalaropes fatally colliding with a power transmission line, which was constructed over a wetland that was intermittently flooded. Highest mortality rates occurred during August and September during the fall migration period.

Management Recommendations from the Literature

Wilson's Phalaropes will benefit from protecting wetlands from drainage (Niemuth and others, 2006). Colwell and Jehl (1994) highlighted the importance of preventing diversion of water from saline lakes and wetlands in western staging areas. Johnson (1996) stressed the value of preserving and restoring wetlands. Several authors have emphasized the importance of protecting wetland complexes containing seasonal and semipermanent wetlands to provide suitable habitat during wet and dry years (Kantrud and Stewart, 1984; Colwell and Oring, 1988b; Niemuth and others, 2006). Wilson's Phalaropes exhibit annual variation in nest-site selection, moving to deeper, more permanent wetlands in dry years (Hohn, 1967; Colwell, 1991).

Colwell and Oring (1988b) recommended preserving wet-meadow areas near deeper wetlands during the breeding season, which may facilitate adults moving their precocial young from nests to wetlands by decreasing overland travel distance. Wilson's Phalaropes nest in upland vegetation early in the breeding season and wet-meadow vegetation later in the season (Colwell and Oring, 1990).

Colwell and Oring (1988b) emphasized that shorebird needs should be considered when creating impoundments for waterfowl, including providing gentle inclines on nesting islands and beaches. Wilson's Phalaropes in Alberta, Saskatchewan, and North Dakota nested on islands or in wet-meadow zones around lakes and wetlands (Bent, 1962; Hohn, 1967; Kagarise, 1979).

Wilson's Phalaropes will benefit from easement programs that discourage the conversion of grasslands to cropland (May and others, 2002), that discourage the planting of trees (Cunningham and Johnson, 2006), and that discourage wetland drainage (Niemuth and others, 2006). Phalaropes also will benefit from planting DNC fields to native grass species and controlling invasive species, where feasible (Wiens, 2007). Naugle and others (1999) recommended controlling or eliminating woody vegetation around wetlands.

Burning, mowing, or grazing can be used to improve nesting habitat (Eldridge, 1992). However, Kantrud and Higgins (1992) emphasized that nesting habitat should not be disturbed (for example, drained, mowed, burned, or heavily grazed) during the breeding season, which generally extends from early May to late July. Prescott and others (1993) recommended deferring livestock grazing until after July 15 in pastures that contain wetlands important to breeding Wilson's Phalaropes (Prescott and others, 1993). Idle grasslands and previously grazed areas provide suitable habitat for nesting, but areas with cattle present during the breeding season are less suitable (Renken, 1983; Renken and Dinsmore, 1987; Kantrud and Higgins, 1992). In Alberta, Wilson's Phalaropes were present in deferred-grazed (grazed after July 15) native pasture (Prescott and others, 1993).

Malcolm (1982) recommended that power lines should not be constructed through or within 1 kilometer of known historical high-water marks of wetlands or dry basins known to hold water intermittently. Similarly, power lines should not be constructed through flight lines or heavily used waterbird migration routes (Malcolm, 1982).

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Table J1. Measured values of vegetation structure and composition in Wilson's Phalarope (*Phalaropus tricolor*) breeding habitat by study. The parenthetical descriptors following authorship and year in the "Study" column indicate that the vegetation measurements were taken in locations or under conditions specified in the descriptor; no descriptor implies that measurements were taken within the general study area.

[cm, centimeter; %, percent; --, no data; DNC, dense nesting cover; WMA, Wildlife Management Area]

Study	State or province	Habitat	Management practice or treatment	Vegetation height (cm)	Vegetation height-density (cm)	Grass cover (%)	Forb cover (%)	Shrub cover (%)	Bare ground cover (%)	Litter cover (%)	Litter depth (cm)
Bomberger, 1984 ^a (nests)	Nebraska	Mixed-grass prairie, shortgrass prairie	--	25.6–31.7	--	99.9	--	--	17.6–22.5	--	0.7
Kantrud and Higgins, 1992 (nests)	Manitoba, Montana, North Dakota, South Dakota	Multiple	Multiple	--	8 ^b , 17 ^c	--	--	--	--	48 ^d	--
Renken, 1983 ^e	North Dakota	Tame grassland (DNC)	Idle, grazed	--	12 ^b	67.9	26.4	7.5	0	99.3	2.3
Svedarsky, 1992 (nests)	Minnesota	Tame grassland (WMA)	--	17 ^f	11 ^b	--	--	--	--	--	--
Wiens, 2007 (nests)	North Dakota	Tame grassland (DNC), pasture	Multiple	15	16 ^b	45	19	--	--	--	1.5
Wiens, 2007 (field)	North Dakota	Tame grassland (DNC), pasture	Multiple	--	18 ^b	53	22	--	--	--	2.0

^aThe sum of the percentages is greater than 100%, based on unclear methods.

^bVisual obstruction reading (Robel and others, 1970).

^cEffective vegetation height.

^dStanding dead vegetation.

^eThe sum of the percentages is greater than 100%, based on the modified point-quadrat technique of Wiens (1969).

^fGrass height.

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