Prepared in cooperation with the Bureau of Land Management and the Great Plains Landscape Conservation Cooperative

Southern Great Plains Rapid Ecoregional Assessment
Volume II. Species and Assemblages

Open-File Report 2018–1109

U.S. Department of the Interior
U.S. Geological Survey

Cover. Swift foxes at Rock River, Wyoming. Photograph by Sandra Taylor, Bureau of Land Management (Creative Commons Attribution 2.0 Generic).

Southern Great Plains Rapid Ecoregional Assessments—
Volume II. Species and Assemblages

By Gordon C. Reese, Natasha B. Carr, and Lucy E. Burris

Prepared in cooperation with the Bureau of Land Management
and the Great Plains Landscape Conservation Cooperative

Open-File Report 2018–1109

U.S. Department of the Interior
U.S. Geological Survey

U.S. Department of the Interior
RYAN K. ZINKE, Secretary

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

U.S. Geological Survey, Reston, Virginia: 2018

For more information on the USGS—the Federal source for science about the Earth, its natural and living resources, natural hazards, and the environment—visit https://www.usgs.gov or call 1–888–ASK–USGS.

For an overview of USGS information products, including maps, imagery, and publications, visit https://store.usgs.gov.

Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Although this information product, for the most part, is in the public domain, it also may contain copyrighted materials as noted in the text. Permission to reproduce copyrighted items must be secured from the copyright owner.

Suggested citation:
Reese, G.C., Carr, N.B., and Burris, L.E., 2018, Southern Great Plains Rapid Ecoregional Assessment—Volume II. Species and assemblages: U.S. Geological Survey Open-File Report 2018–1109, 134 p., https://doi.org/10.3133/ofr20181109.

ISSN 2331-1258 (online)

Acknowledgments

The impetus and oversight for this project was provided by the Southern Great Plains Rapid Ecoregional Assessment advisors including Steve Tryon, Melanie Barnes, Kevin Miller, George Thomas, Patrick Rich, Charles Schmidt, David Herrell, Mark Coca, Carolyn Russell, Dan Baggao, Becky Peters, Robert Pawelek, Kristen Wobbe, and Jason Frels of the Bureau of Land Management and Nicole Athearn and Jonathan Hayes of the Great Plains Landscape Conservation Cooperative. We greatly appreciate reviews of maps and the manuscript provided by George Thomas and Valerie Williams of the Bureau of Land Management; Anne Bartuszevige, Playa Lakes Joint Venture; T. Luke George, Bird Conservancy of the Rockies; Reesa Conrey, Colorado Parks and Wildlife; Michael Wunder, University of Colorado Denver; Stephen Dinsmore, Iowa State University; Mary Bomberger Brown, University of Nebraska; David Augustine, U.S. Department of Agriculture Agricultural Research Service; Paul Hartfield and Clay Nichols, U.S. Fish and Wildlife Service; Paul Cryan, David Eads, Stephen Germaine, Timothy Grabowski, David Haukos, Larry Igl, Catherine Jarnevich, Daniel Manier, Michael O'Donnell, Douglas Ouren, Brian Reichert, Katherine Schoenecker, Susan Skagen, and Marsha Sovoda, U.S. Geological Survey.

The following organizations developed or contributed to the datasets used in the Rapid Ecoregional Assessment: the Bureau of Land Management, Bureau of Meteorology (Australia), Colorado Division of Water Resources, Colorado Natural Heritage Program, Commonwealth Scientific and Industrial Research Organization, Community Earth System Model Contributors (National Science Foundation, Department of Energy, National Center for Atmospheric Research), eBird, Global Biodiversity Information Facility, IHS, Kansas Biological Survey, Kansas Geological Survey, Kansas Natural Heritage Inventory, National Agricultural Statistics Service, Natural Heritage New Mexico, Nebraska Department of Natural Resources, Nebraska Natural Heritage Program, New Mexico Office of the State Engineer, Oklahoma Natural Heritage Inventory, Oklahoma Water Resources Board, Playa Lakes Joint Venture, South Dakota Department of Environment and Natural Resources, South Dakota Natural Heritage Database, Texas Commission on Environmental Quality, Texas Natural Diversity Database, Texas Parks and Wildlife Department, Texas Tech University, Texas Water Development Board, U.S. Army Corps of Engineers, U.S. Census Bureau, U.S. Department of Transportation, U.S. Environmental Protection Agency, U.S. Forest Service, U.S. Geological Survey, World Climate Research Programme's Working Group on Coupled Modelling, Wyoming Natural Diversity Database, Wyoming Water Development Commission, and the Wyoming State Engineer's Office.

Contents

Figures

Tables

Conversion Factors

SI to Inch/Pound

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in)
meter (m)	3.281	foot (ft)
meter (m)	1.094	yard (yd)
millimeter (mm)	0.03937	inch(in)
kilometer (km)	0.6214	mile (mi)
	Area	
square meter (m^2)	10.76	square foot (ft^2)
hectare (ha)	2.471	acre
square kilometer (km^2)	0.3861	square mile (mi^2)
	Volume	
cubic meters (m^3)	0.0008107	acre-feet

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

°F=(1.8×°C)+32

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

°C=(°F-32)/1.8

Scientific Notation

Symbol	Meaning
<	Less than
≤	Less than or equal to
=	Equals
>	Greater than
≥	Greater than or equal to

Abbreviations

ADI aquatic development index

ARS Arkansas River shiner

BLM Bureau of Land Management

EVT Existing Vegetation Type

GAP Gap Analysis Program (U.S. Geological Survey)

GPLCC Great Plains Landscape Conservation Cooperative

LPC lesser prairie-chicken

NHD National Hydrography Dataset (U.S. Geological Survey)

PRISM Parameter-elevation Regressions on Independent Slopes Model

REA Rapid Ecoregional Assessment

SGP	Southern Great Plains
TDI	terrestrial development index
USGS	U.S. Geological Survey
WNS	white-nose syndrome

Species Names

Plants

cheatgrass (*Bromus tectorum*)
crested wheatgrass (*Agropyron cristatum*)
honey mesquite (*Prosopis glandulosa*)
eastern redcedar (*Juniperus virginiana*)
knapweed (*Centaurea* spp.)
Russian olive (*Elaeagnus angustifolia*)
sand sagebrush (*Artemisia filifolia*)
sand shinnery oak (*Quercus havardii*)
tamarisk (*Tamarix* spp.)

Birds

burrowing owl (*Athene cunicularia*)
ferruginous hawk (*Buteo regalis*)
interior least tern (*Sternula antillarum athalassos*)
least tern (*Sternula antillarum*)
lesser prairie-chicken (*Tympanuchus pallidicinctus*)
long-billed curlew (*Numenius americanus*)
mountain plover (*Charadrius montanus*)
piping plover (*Charadrius melodus*)
raven (*Corvus* spp.)
snowy plover (*Charadrius nivosus*)

Mammals

American bison (*Bison bison*)

black-footed ferret (*Mustela nigripes*)

black-tailed prairie dog (*Cynomys ludovicianus*)

coyote (*Canis latrans*)

eastern red bat (*Lasiurus borealis*)

ground squirrels (*Spermophilus* spp.)

hoary bat (*Lasiurus cinereus*)

little brown myotis (*Myotis lucifugus*)

mule deer (*Odocoileus hemionus*)

northern myotis (*Myotis septentrionalis*)

pallid bat (*Antrozous pallidus*)

pocket gophers (Geomyidae)

silver-haired bat (*Lasionycteris noctivagans*)

swift fox (*Vulpes velox*)

tricolored bat (*Perimyotis subflavus*)

Fish

Arkansas River shiner (*Notropis girardi*)

Red River shiner (*Notropis bairdi*)

red shiner (*Cyprinella lutrensis*)

western mosquitofish (*Gambusia affinis*)

Bacteria

sylvatic plague (*Yersinia pestis*)

Contributing Authors

Lucy E. Burris

U.S. Geological Survey, Fort Collins Science Center, Fort Collins, Colo.

Natasha B. Carr

U.S. Geological Survey, Fort Collins Science Center, Fort Collins, Colo.

Gordon C. Reese

U.S. Geological Survey, Fort Collins Science Center, Fort Collins, Colo.

Executive Summary

Rapid Ecoregional Assessments

The overall goal of the Bureau of Land Management (BLM) Rapid Ecoregional Assessments (REAs) is to compile and synthesize regional datasets to facilitate broad-scale evaluation of the effects of change agents on priority species and communities. More specifically, the REAs identify and map the distribution of priority communities and wildlife habitats at broad spatial extents and provide assessments of ecological conditions. The REAs also identify where and to what degree ecological resources are currently at risk from change agents—natural processes or human activities that drive ecosystem change—such as development, fire, invasive species, and climate change. The REAs can help managers identify and prioritize potential areas for conservation or restoration, assess cumulative effects as required by the National Environmental Policy Act, and inform landscape-level planning and management decisions for multiple uses of public lands. Overall, the REAs provide a vehicle for creating stronger, more effective, and more efficient collaboration and cooperation among all parties interested in regional land and resource management and thereby support the BLM landscape approach to resource management.

Rapid Ecoregional Assessment Components

There are several components to the REAs—management questions, conservation elements, and change agents. Management questions, developed by the BLM and other stakeholders, identify the regionally significant information needed to address land-management responsibilities. Conservation elements represent ecological communities and species that are of regional management concern. The emphasis on ecological communities is based on the premise that intact and functioning ecological systems are more resistant and resilient to change agents, including both natural and human stressors. Because it is not feasible to manage or monitor all species individually, the protection of intact ecological communities may serve as a safety net for species not addressed specifically by the REA. Species or species assemblages of management concern not adequately addressed at the community level may be specifically addressed as conservation elements. The REA identifies and assesses the primary factors, or change agents, that currently affect or are likely to affect the condition of communities and species in the future.

The Southern Great Plains Rapid Ecoregional Assessment

The BLM partnered with the Great Plains Landscape Conservation Cooperative (GPLCC) to ensure that the results of the Southern Great Plains REA provide information useful in addressing management issues identified by a diverse set of stakeholders representing both the REA and the GPLCC. The Southern Great Plains (SGP) REA project area includes the full extent of the GPLCC area and four level-III ecoregions: High Plains, Central Great Plains, Southwestern Tablelands, and Nebraska Sand Hills. The project area for this REA is the largest of all completed REAs; it encompasses 961,105 square kilometers (371,085 square miles) and includes portions of Colorado, Kansas, Nebraska, New Mexico, Oklahoma, South Dakota, Texas, and Wyoming.

The Southern Great Plains REA is summarized in a series of three reports. The pre-assessment report summarizes the process used by the REA stakeholders to select management questions, conservation elements, and change agents. It also provides background information for each conservation element selected, including a description of the key ecological attributes and change agents. Volume I of the Southern Great Plains REA report provides background information, overall methods, and data gaps for the REA, as well as summaries for all the ecological communities evaluated for the Southern Great Plains REA. Volume II (this volume) addresses the species and species assemblages evaluated for the Southern Great Plains REA and provides summary maps and graphs of all conservation elements addressed in both volumes.

In volume I, seven major ecological communities were evaluated as conservation elements for the Southern Great Plains REA. Of those seven, four are grassland communities: mixed-grass prairie, shortgrass prairie, sand prairie, and all grassland types combined. The remaining three are aquatic communities: riparian and nonplaya wetlands, playa wetlands and saline lakes, and prairie streams and rivers.

In volume II, a total of 12 species and species assemblages were evaluated for the Southern Great Plains REA: Arkansas River shiner (*Notropis girardi*), ferruginous hawk (*Buteo regalis*), lesser prairie-chicken (*Tympanuchus pallidicinctus*), snowy plover (*Charadrius nivosus*), mountain plover (*Charadrius montanus*), long-billed curlew (*Numenius americanus*), interior least tern (*Sternula antillarum athalassos*), burrowing owl (*Athene cunicularia*), black-tailed prairie dog (*Cynomys ludovicianus*), tree-roosting bat assemblage, swift fox (*Vulpes velox*), and mule deer (*Odocoileus hemionus*). The freshwater mussel assemblage was identified as a priority for the SGP, but data limitations precluded inclusion in the REA.

Assessment Framework

Management questions form the basis of the REA framework. Core management questions relate to the key ecological attributes and change agents associated with each conservation element. Integrated management questions synthesize the results of the primary core management questions into overall landscape-level ranks for each conservation element. The change agents evaluated vary among conservation elements depending on the core management questions and the availability of data. Four change agents were evaluated for the Southern Great Plains REA—fire, invasive species, and climate change were evaluated in volume I, and development (agricultural croplands, urban areas, roads, railroads, and energy and minerals) is addressed for all conservation elements and reported in both volumes. We evaluated development for all species and ecological communities by using either the terrestrial development index or the aquatic development index, which are used to quantify the cumulative landscape-level effects of development. All source and derived datasets used to produce the maps and graphs for REAs are available online at the BLM Geospatial Business Platform (https://gbp-blm-egis.hub.arcgis.com).

Management Implications

REAs summarize information at broad spatial extents and can be used with information at local levels to inform management decisions. For example, REAs can be used as a screening tool to identify potential areas for conservation, restoration, or development projects. Local-level information, including additional surveys and research, can be used to assess conditions not quantified by REAs because of a lack of regional data (such as population sizes of species and occurrence of invasive species). Additionally, REAs can provide assessments of spatially explicit cumulative effects of change agents, especially development. REAs also can augment information from local projects to provide a broader spatial context for evaluating potential effects of proposed actions and alternatives that cannot be determined with local-level information alone. REAs, therefore, contribute to multiscale information necessary for implementing the BLM's landscape approach.

Chapter 1. Introduction and Overview

Introduction

Rapid Ecoregional Assessments

The overall goal of the Bureau of Land Management (BLM) Rapid Ecoregional Assessments (REAs) is to compile and synthesize regional datasets to facilitate broad-scale evaluation of the effects of change agents on priority species and ecological communities. The REAs can help managers identify and prioritize potential areas for conservation or restoration, assess cumulative effects as required by the National Environmental Policy Act, and inform landscape-level planning and management decisions for multiple uses of public lands. They also support the BLM landscape approach to resource management by facilitating collaboration and cooperation among all parties interested in regional land and resource management (Carter and others, 2017). For additional background information on REAs, see the introduction to volume I of the Southern Great Plains (SGP) REA (Reese and others, 2017, chap. 1).

The REA process is guided by a Management Team, Technical Team, and advisors consisting of BLM managers, partner agencies, and technical specialists representing land management within the ecoregion (hereafter referred to as stakeholders) (Assal and others, 2015). An REA entails a two-phase process. In the pre-assessment phase, the lists of priority management questions, conservation elements, and change agents are developed and finalized by the stakeholders. The pre-assessment report documents the process and justification used to identify management questions and conservation elements, and it provides background information on all conservation elements (Assal and others, 2015). The assessment phase includes compilation, synthesis, analysis, and documentation of datasets to address management questions and completion of the ecoregional assessment.

Southern Great Plains Rapid Ecoregional Assessment

Project Area

The region covered by the Southern Great Plains REA (fig. 1–1) includes the maximum area covered by the GPLCC buffered boundary (Manier, 2011), four Level-III ecoregions—High Plains, Central Great Plains, Southwestern Tablelands, and Nebraska Sand Hills (Omernik, 1987)—and an adjacent buffer delineated by fifth-level watersheds intersecting the combined ecoregion boundary (Reese and others, 2017).

Management Questions

The management questions developed by the stakeholders were organized into two general themes: core and integrated. Core management questions were tailored to each ecological community and species to evaluate the potential landscape-level effects of change agents. Integrated management questions synthesize the results of the core management questions to provide an overall evaluation of the landscape-level condition of each conservation element.

Conservation Elements

Ecological Communities

Seven major ecological communities (hereafter referred to as communities) were evaluated as conservation elements for the Southern Great Plains REA. Terrestrial communities evaluated were mixed-grass prairie, shortgrass prairie, and sand prairie. We also evaluated the three grassland communities collectively and included other grassland types present in the project area but not addressed individually: tallgrass, northwest mixed-grass, and cool-season bunchgrass prairies; foothill and saline grasslands; and semidesert grasslands and shrublands. Aquatic communities evaluated were riparian and nonplaya wetlands, playa wetlands and saline lakes, and prairie streams and rivers. See volume 1 of the Southern Great Plains REA (Reese and others, 2017) for more information on the ecological communities.

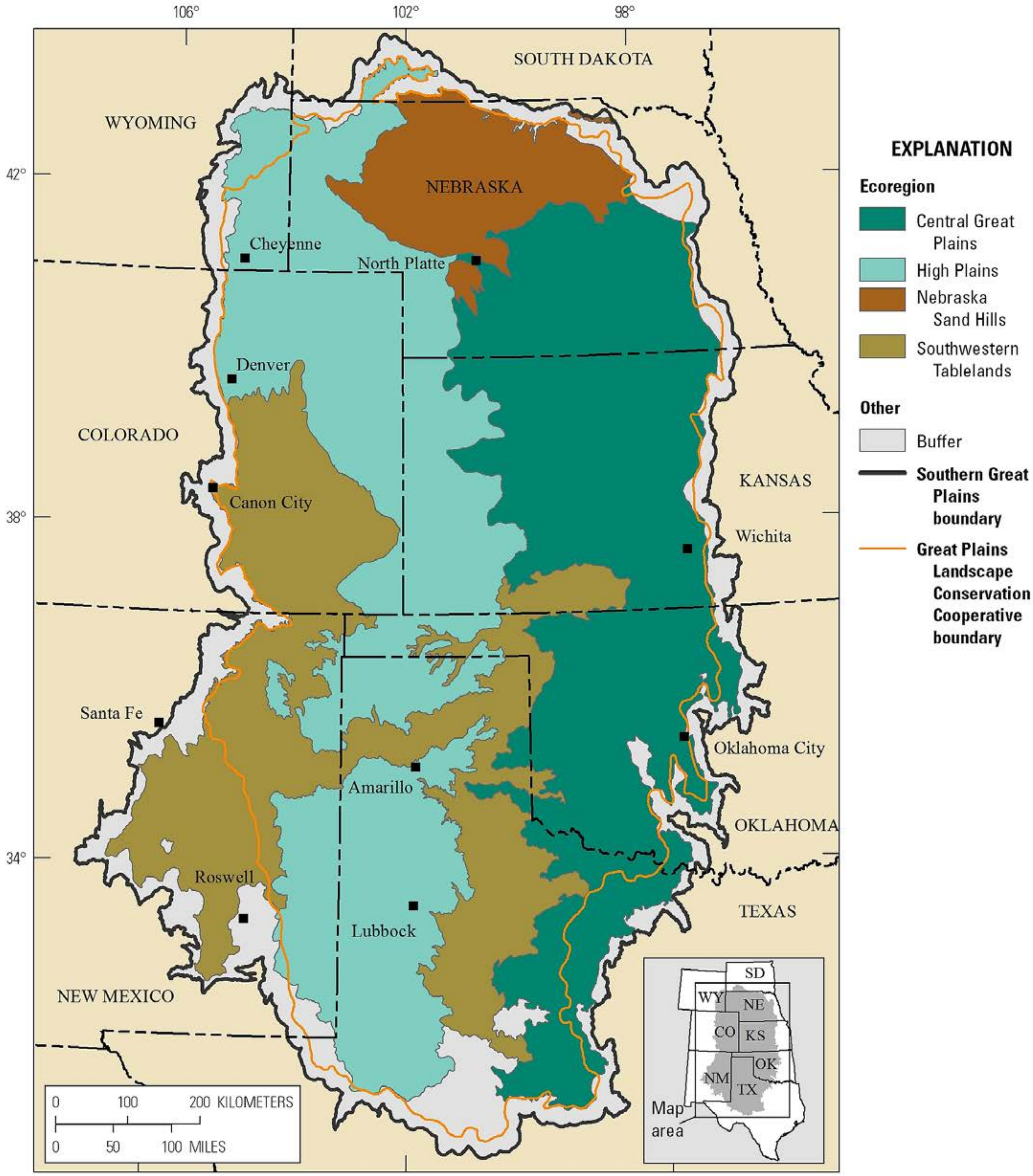


Figure 1–1. Southern Great Plains Rapid Ecoregional Assessment boundary. Level-III ecoregions (Omernik, 1987) and the Great Plains Landscape Conservation Cooperative boundary are shown. (From Reese and others, 2017.)

Table 1–1. Priority species and species assemblages for the Southern Great Plains Rapid Ecoregional Assessment.

System	Taxa	Species	
Aquatic	Invertebrates	Freshwater mussels[1]	
	Fish	Arkansas River shiner	*Notropis girardi*
	Birds	Snowy plover[2]	*Charadrius nivosus*
		Interior least tern[2]	*Sternula antillarum athalassos*
Terrestrial	Birds	Ferruginous hawk	*Buteo regalis*
		Lesser prairie chicken	*Tympanuchus pallidicinctus*
		Mountain plover	*Charadrius montanus*
		Long-billed curlew	*Numenius americanus*
		Burrowing owl	*Athene cunicularia*
	Mammals	Black-tailed prairie dog	*Cynomys ludovicianus*
		Tree-roosting bats	
		Eastern red bat	*Lasiurus borealis*
		Hoary bat	*Lasiurus cinereus*
		Silver-haired bat	*Lasionycteris noctivagans*
		Swift fox	*Vulpes velox*
		Mule deer	*Odocoileus hemionus*

[1]Not addressed in this Rapid Ecoregional Assessment because of data limitations.

[2]Foraging habitat for the snowy plover and interior least tern includes rivers and open water, so these species were evaluated using both aquatic and terrestrial variables.

Species and Assemblages

A preliminary list of priority species and species assemblages was developed during the pre-assessment phase by the stakeholders (Assal and others, 2015). The 13 species and assemblages identified as priorities are listed in table 1–1. The freshwater mussel assemblage was not addressed in the REA because of data limitations.

Change Agents

We evaluated four primary change agents for the REA (development, fire, invasive species, and climate change). We refer to natural drivers of landscape dynamics (such as fire and drought) as key ecological attributes and human influences (such as development, altered fire regimes, invasive species, and anthropogenic climate change) on communities and wildlife habitats as change agents. We initially considered livestock grazing as a change agent, based on input from the stakeholders, but limited data availability precluded a regional assessment (see Reese and others, 2017, chap. 11); the effects of grazing are best addressed through local-level data (Assal and others, 2015). Development was the only change agent evaluated for species and species assemblages. Fire, invasive species, and climate change were evaluated for ecological communities and the SGP overall in volume I (Reese and others, 2017).

Reports and Organization

Pre-Assessment Report

The pre-assessment report (Assal and others, 2015) includes the preliminary management questions, conservation elements, and change agents selected by the REA stakeholders. The report documents the process used to select these REA components for the SGP. Background information is provided on the key ecological attributes and change agents for each conservation element. The background information includes a narrative, an ecological conceptual model that portrays some of the potential primary interactions and feedbacks among change agents, and tables that summarize potential key ecological attributes and change agents. The conceptual models and tables were intended to highlight factors relevant to the REA and are not an exhaustive synthesis of all factors important to a species or community. Not all key ecological attributes and change agents could be addressed because of data and time limitations (see Reese and others, 2017, chap. 11).

Rapid Ecoregional Assessment Reports

The Southern Great Plains REA is summarized in two volumes. Volume I (Reese and others, 2017) provides background information on the REA, methods for all conservation elements, and summaries for all change agents and communities evaluated. Volume II (this volume) addresses the 12 species and species assemblages evaluated and provides an overall summary for all conservation elements for both REA volumes.

Volume II Organization

This chapter provides an overview of the BLM's REA program and the required REA components (additional details are provided in Reese and others, 2017, chap. 1). Chapter 2, "Methods Overview for Species," provides an overview of the assessment framework and describes the methods used to develop baseline habitat distributions for each species and assemblage (additional methods used to assess the core and integrated management questions for each conservation element are addressed in Reese and others, 2017, chap. 2 and appendix A). Chapters 3–14 address the species and assemblages. These chapters have a consistent format and are organized with respect to the management questions. Each chapter includes the following information.

- A brief narrative that highlights ecological information provided in the pre-assessment report (Assal and others, 2015).
- Summary tables for each conservation element (additional details on the indicators are provided in Reese and others, 2017, chap. 2 and appendix A):
    1. The indicators used to evaluate the key ecological attributes.
    2. The indicators used to evaluate change agents.
    3. The ranking factors used for evaluating overall landscape-level rank.
    4. The management questions addressed.
- Maps representing the derived datasets associated with each management question.
- Summary information that highlights a few key findings for each conservation element.

Chapter 15, "REA Synthesis: Species and Communities," summarizes information from volumes I and II. It provides an overview of all communities and species evaluated as a part of the Southern Great Plains REA and identifies the terrestrial and aquatic areas with the least development.

Accessing the Rapid Ecoregional Assessment Datasets

All source and derived datasets for the REAs are served online at the BLM Geospatial Business Platform (https://gbp-blm-egis.hub.arcgis.com).

References Cited

Assal, T.J., Melcher, C.P., and Carr, N.B., eds., 2015, Southern Great Plains Rapid Ecoregional Assessment—Pre-assessment report: U.S. Geological Survey Open-File Report 2015−1003, 284 p., accessed September 2015 at https://doi.org/10.3133/ofr20151003.

Carter, S.K., Carr, N .B., Miller, K.H., and Wood, D.J.A., eds., 2017, Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, 79 p., accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Manier, D., 2011, Great Plains Landscape Conservation Cooperative boundary, buffered 10km: U.S. Geological Survey dataset, accessed April 25, 2016, at https://www.sciencebase.gov/catalog/item/5100263ee4b0ca65fed0d344.

Omernik, J.M., 1987, Ecoregions of the conterminous United States: Annals of the Association of American Geographers, v. 77, p. 118−125, map scale 1:7,500,000.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100.

Chapter 2. Methods Overview for Species

Assessment Framework

Management questions form the foundation of the Southern Great Plains (SGP) Rapid Ecoregional Assessment (REA). This chapter addresses the methods used for addressing management questions for species and assemblages. Additional details on the assessment methods used for all conservation elements are provided in Reese and others (2017, chap. 2 and appendix A). Core management questions relate to the key ecological attributes and change agents for each conservation element, and the integrated management question synthesizes information from core management questions to provide overall landscape-level ranks for conservation elements. The overall landscape-level ranks can be used to identify the largest intact (least developed) areas across the entire distribution of each conservation element, which is one application of REA datasets (Carr and others, 2017). Conservation element chapters each provide a list of all the management questions addressed for that conservation element and results (maps and graphs) organized by management question. The management questions were organized into the following themes.

Core Management Questions

- Where is the conservation element, and what and where are its key ecological attributes?
- What and where are the change agents that potentially affect the conservation element?
- How do the change agents affect the key ecological attributes of the conservation element?

Integrated Management Question

- Where are the areas with the highest overall landscape-level rank?

Change Agents

The management questions, methods, and results addressing the change agents are provided in volume I (Reese and others, 2017, chap. 3). Development was addressed for all conservation elements; fire, invasive species, and climate change were addressed for ecological communities or the entire SGP (Reese and others, 2017), but they were not evaluated for species. An overview of the methods used to address change agents is provided below.

Development

For species that are primarily terrestrial, we evaluated the broad-scale cumulative effects of existing development (agricultural croplands, urban areas, roads, railroads, energy, and minerals) by using the terrestrial development index (TDI). To address core management questions, we used the TDI summarizing the overall footprint area as a percentage of a circular moving window with a radius of 2.5 kilometers (km) (1.55 miles [mi]). TDI scores range from 0 to 100 percent and were divided into seven classes for visualization and analysis purposes. The integrated management questions used the TDI based on a moving window with a radius of 5 km (3.1 mi).

For the three species associated with rivers and open water (Arkansas River shiner [*Notropis girardi*], interior least tern [*Sternula antillarum athalassos*], and snowy plover [*Charadrius nivosus*]), we used the aquatic development index (ADI) to evaluate broad-scale cumulative effects of aquatic development (dams, diversions, and road and railroad stream crossings) as well as the terrestrial development variables (from TDI), which can affect sedimentation rate, flow regime, and water quality. All aquatic and terrestrial development variables were quantified at the local catchment level and by the upstream contributing area for each catchment and were summarized by sixth-level watershed for core management questions and by fifth-level watershed for the integrated management question. Because the ADI addresses both aquatic and terrestrial development for a particular catchment, and because nesting habitat for the snowy plover and the interior least tern is strongly associated with aquatic communities, the ADI addresses both nesting and foraging habitat for the two bird species.

Species differ in their sensitivity to development, and the values of TDI or ADI that correspond to degraded or unsuitable habitat will vary among species. Because of uncertainty in the relationship between TDI or ADI scores and risk from development for a particular species, we retain the entire gradient of development scores in the results. We assume that relatively undeveloped areas represent high landscape-level intactness for all species (Carr and others, 2017) and, consequently, identify these areas for each conservation element.

Fire, Invasive Species, and Climate Change

- *Fire.*—Recent fire occurrence (1984–2014) was evaluated for the entire SGP and summarized by ecological community (Reese and others, 2017, fig. 3–8).
- *Invasive species.*—Available information and models are limited for most invasive species, so we focused on the presence of two woody species in grasslands (honey mesquite [*Prosopis glandulosa*] and eastern redcedar [*Juniperus virginiana*]) (Reese and others, 2017, fig. 4–7). We also evaluated the presence of, and habitat suitability for, two woody species in riparian areas (Russian olive [*Elaeagnus angustifolia*] and tamarisk [*Tamarix* spp.]) (Reese and others, 2017, figs. 8–6, 8–7).
- *Climate change.*—We summarized projected changes in temperature and precipitation for two differing climate change scenarios for the entire SGP (Reese and others, 2017, figs. 3–11, 3–12) and potential effects on grassland communities (Reese and others, 2017, fig. 4–8).

Baseline Habitat for Evaluating Change Agents

To evaluate the current conditions of the SGP, we mapped the baseline habitat for all species and assemblages. For five species, we used published habitat models and maps (table 2–1). For the tree-roosting bat assemblage, a published model was available for only one of the species (hoary bats [*Lasiurus cinereus*]) (Hayes and others, 2015), so we generated models for all three bat species to ensure model consistency among species in this assemblage. We also developed habitat models for remaining six species that lacked published maps or models (table 2–2).

Published Models and Maps

We used published MaxEnt models for two species: the Arkansas River shiner (Worthington and others, 2014) and lesser prairie-chicken (*Tympanuchus pallidicinctus*) (Jarnevich and others, 2016) (table 2–1). MaxEnt is a statistical technique for predicting habitat suitability (also known as species distribution modeling) from occurrence and environmental data (Bellamy and others, 2013). Habitat suitability for the Arkansas River shiner included habitat variables representing climate, land use, geology, stream order, elevation, slope discharge, and river length between barriers in the Arkansas River Basin (Worthington and others, 2014). Occurrence records from nonnative populations in the Pecos and Red River catchments were not included in their model (Worthington and others, 2014). The relative habitat suitability of lek sites for lesser prairie-chickens within their current occupied range was modeled using habitat variables representing land cover, topography, and anthropogenic factors (Jarnevich and others, 2016). The habitat suitability scores for these two species were used to map baseline habitat (see the section “Mapping Baseline Habitat,” p. 9, for additional processing methods).

We used published habitat maps for three species: the interior least tern (U.S. Geological Survey [USGS] Gap Analysis Program, 2011), swift fox (*Vulpes velox*) (Sovada and others, 2009), and mule deer (*Odocoileus hemionus*) (Luce and others, 2005) (table 2–1). We used the summer distribution map for the interior least tern (USGS Gap Analysis Program, 2011) because it corresponded well to breeding colonies mapped by Lott and others (2013). The swift fox distribution was based on a categorical ranking of habitat suitability by a team of subject-matter experts (Sovada and others, 2009). Six categories of mule deer habitat use were mapped by subject-matter experts on a state-by-state basis (Luce and others, 2005). The habitat maps for these three species were used to map baseline habitat (see the section “Mapping Baseline Habitat,” p. 9, for additional processing methods).

Table 2–1. Published models used for mapping baseline habitat for the Southern Great Plains Rapid Ecoregional Assessment.

[USGS, U.S. Geological Survey]

Species	Method	Data source
Arkansas River shiner (*Notropis girardi*)	MaxEnt	Worthington and others (2014)
Lesser prairie-chicken (*Tympanuchus pallidicinctus*)	MaxEnt	Jarnevich and others (2016)
Interior least tern (*Sternula antillarum athalassos*)[1]	Presence-absence by cover type	USGS Gap Analysis Program (2011)
Swift fox (*Vulpes velox*)[2]	Expert opinion	Sovada and others (2009)
Mule deer (*Odocoileus hemionus*)[3]	Expert opinion	Luce and others (2005)

[1]All cover types were included for mapping interior least tern baseline habitat.

[2]Only grassland cover types were included for mapping swift fox baseline habitat.

[3]All habitat-use classes were used for mapping mule deer baseline habitat.

Modeling Habitat Suitability Using MaxEnt

We used MaxEnt (version 3.4.1) to predict the distribution of potential habitat (Phillips and others, 2006) for the remaining nine species (table 2–2). MaxEnt is well suited to using presence-only data to model potential habitat suitability (Peterson and others, 2007). Maps of habitat suitability can be useful for resource managers at broad spatial extents, but they ideally would be used in conjunction with local-level information to account for limitations in the habitat maps resulting from the source data (such as sampling bias, regional variation, or undersampled areas) and model assumptions (such as spatially autocorrelated data) (Bellamy and others, 2013).

We compiled occurrence data from multiple sources (table 2–2). To minimize spatial discrepancies between occurrence data and habitat variables, only occurrences delineated by polygons less than 12.57 square kilometers (km^2) (4.85 square miles [mi^2]) were used, and each polygon was converted to the centroid point. To maximize temporal correspondence between occurrences and habitat variables, we used records after 1996. However, for the bat assemblage, we used records after 1966 because of limited data availability. Occurrences were used regardless of seasonality. To reduce sampling bias, we used home-range estimates from the literature to specify minimum distances between occurrences. Because home range estimates were not available for black-tailed prairie dogs (*Cynomys ludovicianus*) and snowy plovers, we used 42.5 meters (m) (46.5 yards) to ensure that each grid cell would contain no more than one occurrence. Using these constraints, we randomly selected the largest possible number of occurrences for each species (see table 2–2 for minimum linear distances and for the final number of occurrences used to model habitat suitability for each species). For each species, we also randomly selected 10,000 background (pseudoabsence) locations from the SGP project area.

We mapped habitat variables relevant to each species based on the literature. The full list of variables evaluated for all species is provided in table 2–3. To ensure that all variables were in alignment as required by MaxEnt, habitat variables were projected, resampled to 30 m, and clipped to the project boundary as necessary. To summarize the percentage of each cover type (barren, croplands, grasslands, shrublands, or forests), we initially evaluated five circular moving window sizes (radius: 0.27, 0.54, 1, 2.5, or 5 km [0.17, 0.35, 0.62, 1.55, or 3.11 mi, respectively]). For each species, we determined which window size had the greatest predictive power for each cover type by using fivefold cross validation and the jackknife test of variable importance (Phillips, 2009). For each cover type variable, only the top-performing window size was used in the final model. Given the strong association of mountain plovers (*Charadrius montanus*) with prairie dog colonies, we included the black-tailed prairie dog habitat suitability scores as a habitat variable in the mountain plover model. All possible functional relationships (linear, quadratic, product, threshold, and hinge) were permitted in MaxEnt; otherwise, we used the default settings. The percent contribution of each habitat variable in the final model for each species is summarized in table 2–4.

Mapping Baseline Habitat

The final habitat suitability datasets for all species modeled using MaxEnt (tables 2–1 and 2–2) include the full range of habitat suitability scores. To map baseline habitat, we only included areas with habitat suitability scores greater than a threshold corresponding to 10 percent omission error (table 2–5). We used this omission threshold to exclude areas with relatively small suitability values (Radosavljevic and Anderson, 2014). For the bat assemblage, baseline habitat includes areas corresponding to the 10 percent threshold for at least one species. A 10 percent omission error threshold was provided for the lesser prairie-chicken habitat suitability model (Jarnevich and others, 2016). For the Arkansas River shiner threshold, we used a habitat suitability index of 0.5; there were no areas outside of the Canadian River catchment with habitat suitability scores greater than 0.5, and this species has only been recorded at three sites outside of that catchment since 1990 (Worthington and others, 2014). The full range of habitat suitability scores for all species except lesser prairie-chicken are served online at the BLM Geospatial Business Platform (https://gbp-blm-egis.hub.arcgis.com), which can be used to map habitat suitability according to other omission thresholds.

The habitat of species that were not modeled using MaxEnt were processed as follows. For interior least terns, the available data are binary (presence or absence), and mapped breeding habitat was included in the baseline distribution. For swift foxes, we included both high- and medium-quality grassland habitat but excluded habitat classified as agricultural lands (Sovada and others, 2009). All mapped habitat-use categories for mule deer were included (Luce and others, 2005).

For the fish, snowy plover, and interior least tern, there was no additional processing for the baseline habitat maps (figs. 3–1, 6–1, 9–1). For the remaining primarily terrestrial species, habitat coincident with the surface disturbance footprint from development (see Reese and others, 2017, appendix A) was removed from the final baseline habitat map in order to account for habitat conversion from development not adequately quantified by the landcover variables in table 2–4 (figs. 4–1, 5–1, 7–1, 8–1, 10–1, 11–1, 12–1, 13–1, 14–1). Hereafter, we refer to predicted baseline habitat for maps derived from MaxEnt and estimated baseline habitat for maps derived from other methods (table 2–1). Baseline habitat represents areas where a species could occur based on environmental variables, but it does not confirm a species presence or quantify density and abundance.

Table 2–2. Presence-only occurrence data used to model habitat suitability with MaxEnt for the Southern Great Plains Rapid Ecoregional Assessment.

[See table 1–1 for scientific names. –, data were not available; USGS, U.S. Geological Survey]

Data source	Number of occurrences by species								
	Ferruginous hawk	Snowy plover	Mountain plover	Long-billed curlew	Burrowing owl	Black-tailed prairie dog	Tree-roosting bat assemblage		
							Eastern red bat	Hoary bat	Silver-haired bat
Colorado Natural Heritage Program[1]	27	–	26	16	–	80	–	–	–
eBird[2]	6,765	2,393	889	2,399	5,536	–	–	–	–
Global Biodiversity Information Facility[3]	–	–	–	–	–	362	–	–	–
Kansas Natural Heritage Inventory[4]	4	2	2	10	39	–	–	–	–
Natural Heritage New Mexico[5]	9	–	–	28	68	193	–	3	3
Nebraska Natural Heritage Program[6]	98	–	585	143	301	1	–	–	–
Oklahoma Natural Heritage Inventory[7]	–	2	–	1	1	2	–	–	–
South Dakota Natural Heritage Database[8]	–	–	–	1	21	–	–	–	–
Texas Natural Diversity Database[9]	–	2	2	–	5	232	–	–	–
U.S. Forest Service[10]	–	–	–	–	–	199	–	–	–
USGS bat database[11]	–	–	–	–	–	–	213	316	119
Wyoming Natural Diversity Database[12]	104	–	37	63	177	24	2	–	–
Total compiled occurrences	7,007	2,399	1,541	2,661	6,148	1,093	215	319	122
Minimum distance, in meters[13]	1,400	42.5	770	178.5	890	42.5	1,000	1,000	1,000
Number of occurrences used[14]	3,052	383	633	1,362	2,462	1,016	69	74	34

[1]Occurrence data were provided by the Colorado Natural Heritage Program in 2015.

[2]Sullivan and others (2009)

[3]Global Biodiversity Information Facility (2017)

[4]Occurrence data were provided by the Kansas Natural Heritage Inventory in 2015.

[5]Occurrence data were provided by the Natural Heritage New Mexico in 2014.

[6]Occurrence data were provided by the Nebraska Natural Heritage Program in 2015.

[7]Occurrence data were provided by the Oklahoma Natural Heritage Inventory in 2014.

[8]Occurrence data were provided by the South Dakota Natural Heritage Database in 2015.

[9]Occurrence data were provided by the Texas Natural Diversity Database in 2014.

[10]Prairie dog town data were provided by the U.S. Forest Service for Cimarron and Comanche National Grasslands in 2015.

[11]Ellison and others (2003)

[12]Occurrence data were provided by the Wyoming Natural Diversity Database in 2015.

[13]Minimum distance allowed between a random selection of occurrences.

[14]Total number of occurrences used in MaxEnt models.

Table 2–3. Variables used to model habitat suitability with MaxEnt for the Southern Great Plains Rapid Ecoregional Assessment.

[See table 2–2 for the list of species whose habitats were modeled. km, kilometer; cm, centimeter; m, meter; USGS, U.S. Geological Survey; NHDPlus, National Hydrography Dataset Plus; PRISM, Parameter-elevation Regressions on Independent Slopes Model]

Habitat variable description	Abbreviation	Data source
Soil, available water content	AvailWCont	POLARIS; Chaney and others (2016)
Percent of cells classified as barren in 5-km radius	BarPer5k	LANDFIRE Existing Vegetation Types; LANDFIRE (2012)
Black-tailed prairie dog habitat suitability score[1]	BTPDSuit	This volume, see table 2-4
Soil, percent clay: surface to 5 cm	ClayPer5cm	POLARIS; Chaney and others (2016)
Cosine of aspect	CosAspect	USGS National Elevation Dataset; U.S. Geological Survey (2009)
Percent of cells classified as cropland in 270-m radius	CropPer270	LANDFIRE Existing Vegetation Types; LANDFIRE (2012)
Percent of cells classified as cropland in 5-km radius	CropPer5k	LANDFIRE Existing Vegetation Types; LANDFIRE (2012)
Dominant tree type in 270-m radius	DomTree270	LANDFIRE Existing Vegetation Types; LANDFIRE (2012)
Elevation	Elevation	USGS National Elevation Dataset; U.S. Geological Survey (2009)
Percent of cells classified as grassland in 270-m radius	GrsPer270	Reese and others (2016)
Percent of cells classified as grassland in 5-km radius	GrsPer5k	Reese and others (2016)
Euclidean distance to nearest oil/gas development	OilGasDist	IHS, Inc. (2014)
Soil, percent organic matter: surface to 5 cm	OMatPer5cm	POLARIS; Chaney and others (2016)
Euclidean distance to nearest perennial water source	PerenDist	NHDPlus; U.S. Environmental Protection Agency and U.S. Geological Survey (2012)
Euclidean distance to nearest playa	PlayaDist	Reese and others (2017)
Precipitation of the coldest quarter	PrecColdQ	PRISM; PRISM Climate Group (2004)
Precipitation of the hottest quarter	PrecHotQ	PRISM; PRISM Climate Group (2004)
Precipitation seasonality (coefficient of variation)	PrecSeasCV	PRISM; PRISM Climate Group (2004)
Euclidean distance to riparian area	RiparDist	Reese and others (2016), LANDFIRE Existing Vegetation Types; LANDFIRE (2012)
Euclidean distance to saline water source	SalineDist	Reese and others (2017)
Soil, percent sand: surface to 5 cm	SandPer5cm	POLARIS; Chaney and others (2016)
Percent of cells classified as shrubland in 2.5-km radius	ShrbPer2k	LANDFIRE Existing Vegetation Types; LANDFIRE (2012)
Percent of cells classified as shrubland in 5-km radius	ShrbPer5k	LANDFIRE Existing Vegetation Types; LANDFIRE (2012)
Soil, percent silt: surface to 5 cm	SiltPer5cm	POLARIS; Chaney and others (2016)
Slope, in degrees	SlopeDeg	USGS National Elevation Dataset; U.S. Geological Survey (2009)
Soil, pH in water	SoilpH	POLARIS; Chaney and others (2016)
Temperature annual range	TempAnRnge	PRISM; PRISM Climate Group (2004)
Mean temperature of the coldest quarter	TempColdQ	PRISM; PRISM Climate Group (2004)
Mean temperature of the hottest quarter	TempHotQ	PRISM; PRISM Climate Group (2004)
Percent of cells classified as trees in 270-m radius	TreePer270	LANDFIRE Existing Vegetation Types; LANDFIRE (2012)
Percent of cells classified as trees in 1-km radius	TreePer1k	LANDFIRE Existing Vegetation Types; LANDFIRE (2012)
Percent of cells classified as trees in 5-km radius	TreePer5k	LANDFIRE Existing Vegetation Types; LANDFIRE (2012)
Topographic ruggedness index, surrounding 8 cells	TRI	USGS National Elevation Dataset; U.S. Geological Survey (2009)
Euclidean distance to nearest open water source	WaterDist	Reese and others (2016), LANDFIRE Existing Vegetation Types; LANDFIRE (2012)

[1]Black-tailed prairie dog habitat suitability score is derived from the MaxEnt model, which was only used as a habitat variable for mountain plovers (see table 1–1 for scientific names).

Table 2–4. Percent contribution of habitat variables included in MaxEnt habitat suitability models for the Southern Great Plains Rapid Ecoregional Assessment.

[See table 1–1 for scientific names. –, variable was not included in the final model]

Variable abbreviation[1]	Percent contribution of variable								
							Tree roosting bat assemblage		
	Ferruginous hawk	Snowy plover	Mountain plover	Long-billed curlew	Burrowing owl	Black-tailed prairie dog	Eastern red bat	Hoary bat	Silver-haired bat
AvailWCont	–	–	–	5.5	–	3.8	–	–	–
BarPer5k[2]	6.0	2.1	0.5	0	1.8	2.0	–	–	–
BTPDSuit[3]	–	–	9.9	–	–	–	–	–	–
ClayPer5cm	–	–	–	–	2.3	0.9	–	–	–
CosAspect[2]	0	–	–	0.2	0.1	–	–	–	–
CropPer270	9.4	–	–	5.5	9.3	10.1	–	–	–
CropPer5k	–	–	6.2	–	–	–	–	–	–
DomTree270	7.5	–	–	–	–	–	–	–	–
Elevation	38.6	1.4	64.6	26.7	29.1	14.0	–	–	–
GrsPer270	19.9	–	–	12.9	10.0	0.6	–	–	–
GrsPer5k	–	–	1.5	–	–	–	–	–	–
OilGasDist	2.8	–	–	–	–	–	–	–	–
OMatPer5cm[2]	–	–	–	–	–	0	–	–	–
PerenDist	–	0.3	–	–	0.3	–	8.7	18.1	34.8
PlayaDist	–	4.1	–	2.1	–	–	–	–	–
PrecColdQ	–	–	–	5.2	12.9	2.6	–	–	–
PrecHotQ	–	–	–	1.9	7.5	15.1	–	–	–
PrecSeasCV	0.5	–	–	–	–	–	4.4	10.1	11.4
RiparDist	0.5	–	–	0.3	–	–	4.4	2.0	3.2
SalineDist	–	11.8	–	–	–	–	–	–	–
SandPer5cm	–	0.6	–	–	0.1	2.1	–	–	–
ShrbPer2k	–	–	–	–	–	–	11.7	8.5	–
ShrbPer5k	–	–	–	7.3	6.8	8.7	–	–	8.6
SiltPer5cm	–	0.2	–	–	1.4	–	–	–	–
SlopeDeg	5.4	8.4	5.4	7.9	7.6	5.3	–	–	–
SoilpH	–	–	–	–	–	9.4	–	–	–
TempAnRnge	–	–	–	–	–	–	0.4	3.4	2.1
TempColdQ	–	–	–	3.7	3.8	16.9	5.6	10.3	8.1
TempHotQ	2.8	–	–	1.7	0.8	0.5	1.9	7.5	6.1
TreePer270	1.9	–	–	–	–	–	–	–	–
TreePer1k	–	–	–	–	–	–	24.1	27.5	–
TreePer5k	–	–	10.3	1.1	4.0	1.1	–	–	12.8
TRI	0.3	0.5	1.6	2.4	0.1	6.9	16.0	10.2	7.2
WaterDist	4.4	70.6	–	15.6	2.1	–	22.8	2.4	5.7

[1]See table 2–3 for descriptions of habitat variables.

[2]Percent contribution values of 0 indicate the variable had no predictive value in the final model or it was correlated with other variables.

[3]BTPDSuit is the habitat suitability score for black-tailed prairie dog derived from the MaxEnt model. BTPDSuit was used as a habitat variable for mountain plovers only.

Management Questions

The general themes for management questions were used to develop the following specific core and integrated management questions for terrestrial and aquatic species and species assemblages.

Core Management Questions

Where does existing development pose the greatest threat to baseline habitat, and where are the large, relatively undeveloped areas?

To evaluate the cumulative effects of development for each primarily terrestrial species and the bat assemblage, we overlaid the TDI on the baseline habitat (figs. 4–2, 5–2, 7–2, 8–2, 10–2, 11–2, 12–2, 13–2, 14–2). To identify terrestrial areas with the least development, we defined relatively undeveloped areas as having a TDI score ≤2 percent. For the Arkansas River shiner, snowy plover, and interior least tern, we overlaid ADI on the baseline habitat (figs. 3–2, 6–2, 9–2). Relatively undeveloped aquatic areas were defined as having an ADI score ≤20.

How has development fragmented baseline habitat?

We evaluated the fragmenting effects of development for each terrestrial species by comparing patch sizes of baseline and relatively undeveloped habitat (figs. 4–5, 5–5, 7–5, 8–5, 10–5, 11–5, 12–5, 13–5, 14–5). For the Arkansas River shiner, we derived stream-segment length of perennial streams based on the presence of dams. Differences in stream-segment length between the baseline distribution and relatively undeveloped areas were used as an index of fragmentation (fig. 3–4). Fragmentation of habitat for interior least tern and snowy plover was not evaluated because their habitats are naturally patchy and dynamic.

Integrated Management Question for Evaluating Species

Where are the areas with the highest overall landscape-level ranks?

The integrated management question synthesizes the results from core management questions for each species or assemblage (figs. 3–5, 4–6, 5–6, 6–4, 7–6, 8–6, 9–4, 10–6, 11–6, 12–6, 13–6, 14–6). We used the amount and distribution of baseline habitat, which is a key ecological attribute, for summarizing landscape-level area (or density for streams). We used TDI or ADI for summarizing landscape-level development for baseline habitat. For each species, landscape-level area (density) and landscape-level development were ranked, and these ranks were combined into an overall landscape-level rank (for additional details, see Reese and others, 2017, chap. 2). The highest overall landscape-level rank represents locations with the largest area (or density) of baseline habitat and the lowest development levels. The lowest landscape-level rank represents locations with the smallest area (or density) of baseline habitat and the highest development levels. Because rankings are sensitive to the input data and criteria used to develop the ranking thresholds, they are not intended to be standalone maps. However, they are useful for comparing rankings among areas in the SGP when used in conjunction with more detailed geospatial data summarized for core management questions.

Table 2–5. Minimum thresholds in habitat suitability scores used to map baseline habitat for the Southern Great Plains Rapid Ecoregional Assessment. Thresholds correspond to 10 percent omission error in habitat suitability derived from MaxEnt, unless otherwise noted.

Species	Habitat suitability threshold
Fish	
Arkansas River shiner (*Notropis girardi*)[1]	0.500000
Birds	
Ferruginous hawk (*Buteo regalis*)	0.272833
Lesser prairie-chicken (*Tympanuchus pallidicinctus*)[2]	0.255000
Snowy plover (*Charadrius nivosus*)	0.135297
Mountain plover (*Charadrius montanus*)	0.264030
Long-billed curlew (*Numenius americanus*)	0.311720
Burrowing owl (*Athene cunicularia hypugaea*)	0.323906
Mammals	
Black-tailed prairie dog (*Cynomys ludovicianus*)	0.297115
Mammals—Tree-roosting bat assemblage	
Eastern red bat (*Juniperus virginiana*)	0.301635
Hoary bat (*Lasiurus cinereus*)	0.234752
Silver-haired bat (*Lasionycteris noctivagans*)	0.320346

[1]MaxEnt model and threshold, corresponding to most extant populations, from Worthington and others (2014). There were no areas outside of the Canadian River catchment with habitat suitability scores greater than 0.5. The Arkansas River shiner has only been recorded at three sites outside the Canadian River catchment since 1990 (Worthington and others, 2014).

[2]MaxEnt model and threshold, corresponding to 10 percent omission error, from Jarnevich and others (2016).

References Cited

Bellamy, C., Scott, C., and Altringham, J., 2013, Multiscale, presence-only habitat suitability models—Fine-resolution maps for eight bat species: Journal of Applied Ecology, v. 50, p. 892–901.

Carr, N.B., Leinwand I.I.F., and Wood, D.J.A., 2017, A multiscale index of landscape intactness for management of public lands, chap. 5 *of* Carter, S.K., Carr, N.B., Miller, K.H, and Wood, D.J.A., eds., Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, p. 55–74, accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Chaney, N.W., Wood, E.F., McBratney, A.B., Hempel, J.W., Nauman, T., Brungard, C.W., and Odgers, N.P., 2016, POLARIS—A 30-meter probabilistic soil series map of the contiguous United States: Geoderma, v. 274, p. 54–67, accessed March 6, 2017, at http://stream.princeton.edu/POLARIS/.

Ellison, L.E., O'Shea, T.J., Bogan, M.A., Everette, A.L., and Schneider, D.M., 2003, U.S. Geological Survey bat population database for the United States and trust territories (ver. 1.0): U.S. Geological Survey database, accessed February 2014 at https://www.fort.usgs.gov/products/23233.

Global Biodiversity Information Facility, 2017, *Cynomys ludovicianus*: Global Biodiversity Information Facility web page, accessed September 11, 2017, at https://doi.org/10.15468/dl.wxk9i9.

Hayes, M.A., Cryan, P.M., and Wunder, M.B., 2015, Seasonally-dynamic presence-only species distribution models for a cryptic migratory bat impacted by wind energy development: PLoS ONE, v. 10, no. 7, article e0132599, 20 p., accessed May 16, 2016, at https://doi.org/10.1371/journal.pone.0132599.

IHS, Inc., 2014, Production unallocated database: Englewood, Colo., IHS, Inc., accessed January 16, 2015, at https://www.ihs.com/products/oil-gas-tools-enerdeq-browser.html.

Jarnevich, C.S., Holcombe, T.S., Grisham, B.A., Timmer, J., Boal, C.W., Butler, M.J., Pitman, J., Kyle, S.C., Klute, D., Beauprez, G.M., Janus, A., and Van Pelt, W.E., 2016, Assessing range-wide habitat suitability for the lesser prairie-chicken: Avian Conservation and Ecology, v. 11, no. 2, accessed August 23, 2017, at https://doi.org/10.5751/ACE-00807-110102.

LANDFIRE, 2012, Existing Vegetation Type (ver. 1.3.0): U.S. Forest Service and U.S. Department of the Interior, accessed December 29, 2014, at https://landfire.gov/NationalProductDescriptions21.php.

Lott, C.A., Wiley, R.L., Fischer, R.A., Hartfield, P.D., and Scott, J.M., 2013, Interior least tern (*Sternula antillarum*) breeding distribution and ecology—Implications for population-level studies and the evaluation of alternative management strategies on large, regulated rivers: Ecology and Evolution, v. 3, no. 10, p. 3613–3627.

Luce, A., Hurd, W., and Lowry, J.H., 2005, Mule Deer Mapping Project: Utah State University, Remote Sensing/GIS Laboratory data, accessed August 30, 2017, at https://www.gis.usu.edu/projects/mule-deer-mapping-project/.

Peterson, A.T., Papes, M., and Eaton, M., 2007, Transferability and model evaluation in ecological niche modeling—A comparison of GARP and Maxent: Ecography, v. 30, p. 550–560.

Phillips, S., 2009, A brief tutorial on Maxent: Lessons in Conservation, v. 3, p. 109–135.

Phillips, S.J., Anderson, R.P., and Schapire, R.E., 2006, Maximum entropy modeling of species geographic distributions: Ecological Modelling, v. 190, p. 231–259.

PRISM Climate Group, 2004, 30-year normals: Oregon State University, PRISM Climate Group dataset, accessed June 28, 2012, at http://prism.oregonstate.edu/normals/.

Radosavljevic, A., and Anderson, R.P., 2014, Making better MAXENT models of species distributions—Complexity, overfitting and evaluation: Journal of Biogeography, v. 41, p. 629–643.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100.

Reese, G.C., Manier, D.J., Carr, N.B., Callan, R., Leinwand, I.I.F., Assal, T.J., Burris, L., and Ignizio, D.A., 2016, Estimated historical distribution of grassland communities of the Southern Great Plains: U.S. Geological Survey Open-File Report 2016–1184, 13 p., accessed December 12, 2016, at https://doi.org/10.3133/ofr20161184.

Sovada, M.A., Woodward, R.O., and Igl, L.D., 2009, Historical range, current distribution, and conservation status of the swift fox, *Vulpes velox*, in North America: Canadian Field-Naturalist, v. 123, p. 346–367.

Sullivan, B.L., Wood, C.L., Iliff, M.J., Bonney, R.E., Fink, D., and Kelling, S., 2009, eBird—A citizen-based bird observation network in the biological sciences: Biological Conservation, v. 142, p. 2282–2292.

U.S. Environmental Protection Agency and U.S. Geological Survey, 2012, NHDPlus 2.1 [vector digital data, tabular digital data]: U.S. Environmental Protection Agency and U.S. Geological Survey, accessed January 25, 2015, at http://www.horizon-systems.com/NHDPlus/NHDPlusV2_data.php.

U.S. Geological Survey, 2009, National Elevation Dataset: U.S. Geological Survey data, accessed December 2011 at https://nationalmap.gov/elevation.html.

U.S. Geological Survey [USGS] Gap Analysis Program, 2011, National GAP vertebrate species distribution model: U.S. Geological Survey application, accessed October 5, 2015, at http://gapanalysis.usgs.gov/species.

Worthington, T.A., Brewer, S.K., Grabowski, T.B., and Mueller, J., 2014, Backcasting the decline of a vulnerable Great Plains reproductive ecotype—Identifying threats and conservation priorities: Global Change Biology, v. 20, p. 89–102.

Chapter 3. Arkansas River Shiner

Introduction

The Arkansas River shiner (*Notropis girardi*) is a small-bodied (less than 65 millimeters [mm]) member of the minnow family (Cyprinidae). The Arkansas River shiner (hereafter referred to as ARS) is a broadcast spawner (the eggs and milt are released into the water column) that spawns multiple times during the spring and summer; their nonadhesive and semibuoyant eggs are distributed by river currents (Bonner and Wilde, 2000). Broadcast-spawning cyprinids are extremely vulnerable to altered flow regimes, habitat fragmentation, and invasive fishes; consequently, many cyprinids are either of conservation concern or unknown status (Worthington and others, 2014, 2016, 2018). The ARS currently occupies only 20 percent of its original distribution, leading to its listing as threatened under the Endangered Species Act (U.S. Fish and Wildlife Service, 1998, 2005); critical habitat for the ARS was finalized in 2005 (U.S. Fish and Wildlife Service, 2005). The last remaining strongholds for native ARS populations are restricted to streams primarily associated with the Canadian River (New Mexico, Texas, and Oklahoma). Although the species may also be extant in reaches of the Cimarron River in Kansas, the current status of the Cimarron population is unclear (Larson, 1991; Pigg, 1991; Wilde, 2002). A nonnative population in the Pecos River (New Mexico), which is outside of the upper Arkansas River basin, is believed to have been established from released baitfish (Bestgen and others, 1989; Hoagstrom and Brooks, 2005; Osborne and others, 2013).

The ARS is found in main channels of braided, shallow (15–25 cm deep), wide rivers with sandy bottoms and slow currents (25–40 cubic meters per second [m^3/s]) (Polivka, 1999; Bonner and Wilde, 2000). The stream dynamics of their habitat is highly variable, and the species tolerates extreme physiochemical conditions (Wilde, 2002; Worthington and others, 2014, 2018). The ARS primarily forages on benthic invertebrates, but their diet also includes plants, algae, and detritus (Wilde and others, 2001).

To reproduce successfully, the ARS requires relatively long, unfragmented rivers (greater than 200 km) (Moore, 1944; Perkin and Gido, 2011). The probability of ARS presence increases in stream segments greater than 375 km (Worthington and others, 2014), and successful reproduction decreases in streams fragmented by impoundments and low flows (Durham and Wilde, 2006). Models indicate that the magnitude of streamflow (above a minimum threshold) appears to be less important to reproductive success than the presence of flowing water (Durham and Wilde, 2006).

The Arkansas River has been highly degraded and fragmented by agricultural activities, including reservoir and dam construction, groundwater pumping, stream channelization, and pesticide runoff, as well as by sedimentation from nearby terrestrial development and sewage effluent (Limbird, 1993; Perkin and Gido, 2011; Worthington and others, 2016). Fragmentation impedes spawning migrations and passive movement of both eggs and larvae (Bonner, 2000). Changes in the natural flow regime and pumping from the Ogallala aquifer can decrease overall flows and flow variability, which have been correlated with population declines of the ARS (Bonner and Wilde, 2000; Falke and others, 2010; Perkin and Gido, 2011). The ARS is short-lived (typically <2 years), and maturity is reached by 1 year of age (Wilde, 2002); consequently, the ARS may be particularly vulnerable to long-term hydrological alteration.

Several invasive fish species potentially pose threats to the ARS, including the Red River shiner (*Notropis bairdi*), the red shiner (*Cyprinella lutrensis*), and the western mosquitofish (*Gambusia affinis*) (Luttrell and others, 1995; Pigg and others, 1999). The ecologically similar Red River shiner has largely replaced the ARS in the Cimarron River since its introduction in the 1960s (Felley and Cothran, 1981); whether this is due to direct competition or better resistance to the modified conditions is unclear. Red shiners prey on juvenile fishes including ARS (Gido and others, 1999). Western mosquitofish prey on eggs and larvae and may pose a risk to the ARS in several Oklahoma rivers, including the Canadian River (Pigg and others, 1999). Additional background information on the Arkansas River shiner can be found in the SGP pre-assessment report (Roberts, 2015).

Rapid Ecoregional Assessment Components Evaluated for the Arkansas River Shiner

The key ecological attributes and change agents addressed by core management questions for ARS habitat include the amount and distribution, landscape structure, and development (tables 3−1 and 3−2). Fire occurrence and climate change were evaluated for the entire SGP (see Reese and others, 2017, chap. 3). Overall landscape-level ranking variables are summarized in table 3−3. The core and integrated management questions are listed in table 3−4.

Table 3–1. Key ecological attributes and associated indicators used to address core management questions for Arkansas River shiners for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Amount and distribution	Total area	Baseline habitat (year round)[2]
Landscape structure	Index of fragmentation	Stream-segment length for baseline habitat
Landscape dynamics	Fire occurrence	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2] MaxEnt was used to predict baseline habitat (Worthington and others, 2014). See chapter 2, "Methods Overview for Species," for methods and datasets used.

Table 3–2. Anthropogenic change agents and associated indicators used to address core management questions for Arkansas River shiners for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Development	Aquatic development index (ADI)	Percentage of baseline habitat in seven development classes by sixth-level watershed
	Index of fragmentation	Stream-segment length for relatively undeveloped[2] habitat
Climate change	Projected temperature and precipitation	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]Aquatic development index score less than or equal to 20.

Table 3–3. Landscape-level variables used to address the integrated management question for Arkansas River shiners. Ranks for landscape-level density and development were combined into an overall landscape-level rank for the Southern Great Plains Rapid Ecoregional Assessment.

[>, greater than]

Landscape-level variables[1]	Description	Relative rank[2]		
		Lowest	Medium	Highest
Density	Density of Arkansas River shiner habitat (stream length of baseline habitat/ area of fifth-level watershed)	>0–0.01	>0.01–0.03	>0.03
Development	Mean aquatic development index (ADI) score for Arkansas River shiner habitat, summarized by fifth-level watershed	0–20	>20–40	>40

[1]See Reese and others (2017, chap. 2) for methods used to address integrated management questions.

[2]Ranking breakpoints for density of baseline habitat were determined from equal subsets of the data. Ranking breakpoints for the aquatic development index were standardized for all aquatic conservation elements.

Table 3–4. Management questions addressed for Arkansas River shiners for the Southern Great Plains Rapid Ecoregional Assessment.

Core management questions[1]	Results
What is the distribution of baseline habitat for Arkansas River shiners?	Figure 3−1
Where does existing development pose the greatest threat to Arkansas River shiner habitat, and where are the large, relatively undeveloped areas?	Figures 3−2 and 3−3
How has development fragmented Arkansas River shiner habitat?	Figure 3−4
Integrated management question[2]	**Results**
Where is Arkansas River shiner habitat with the highest overall landscape-level rank?	Figure 3−5

[1]See Reese and others (2017, chap. 11) for management questions that could not be addressed.

[2]See table 3−3.

Management Questions and Results

What is the distribution of baseline habitat for Arkansas River shiners (fig. 3–1)?

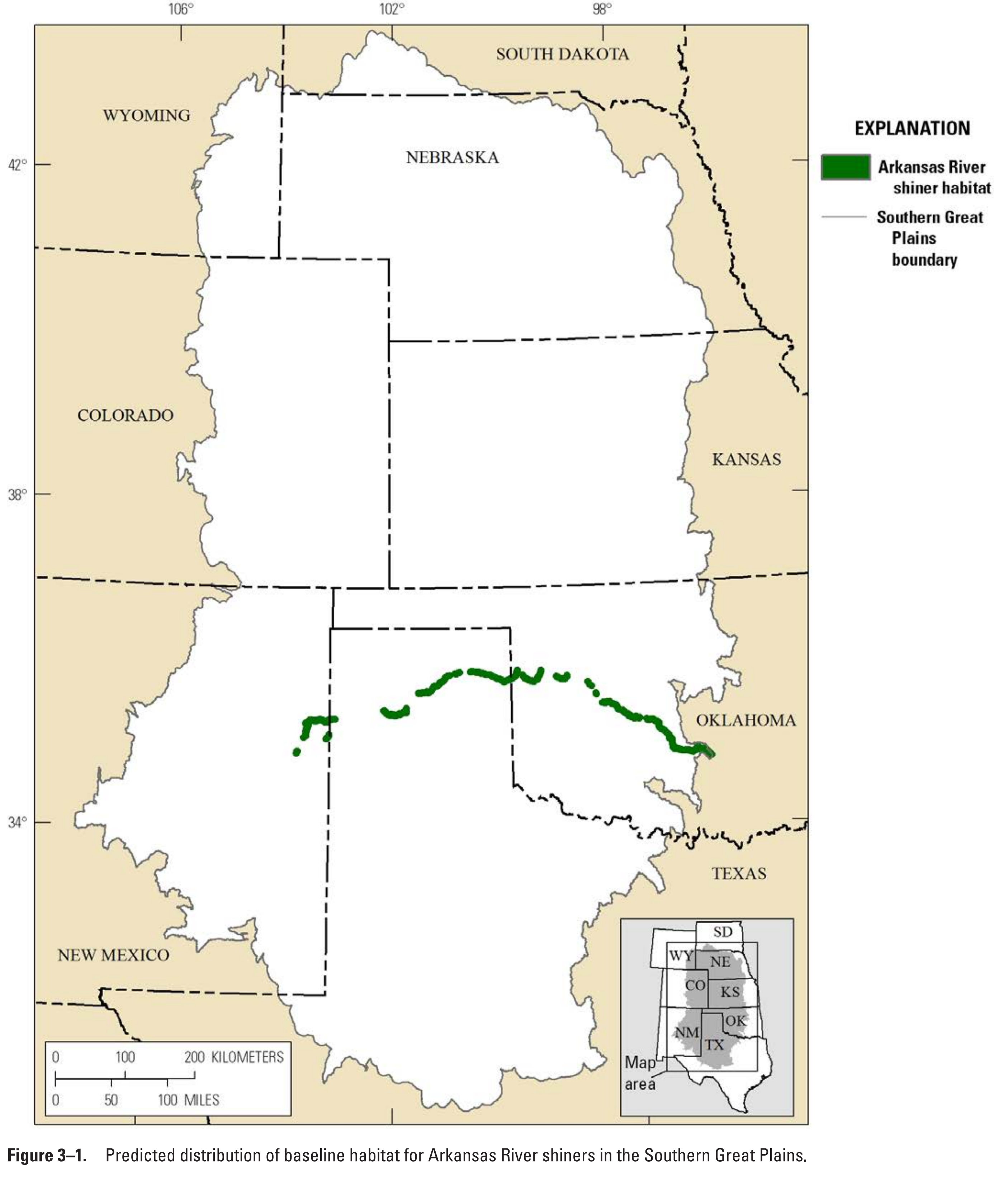


Figure 3–1. Predicted distribution of baseline habitat for Arkansas River shiners in the Southern Great Plains.

Where does existing development pose the greatest threat to Arkansas River shiner habitat, and where are the large, relatively undeveloped areas (figs. 3–2 and 3–3)?

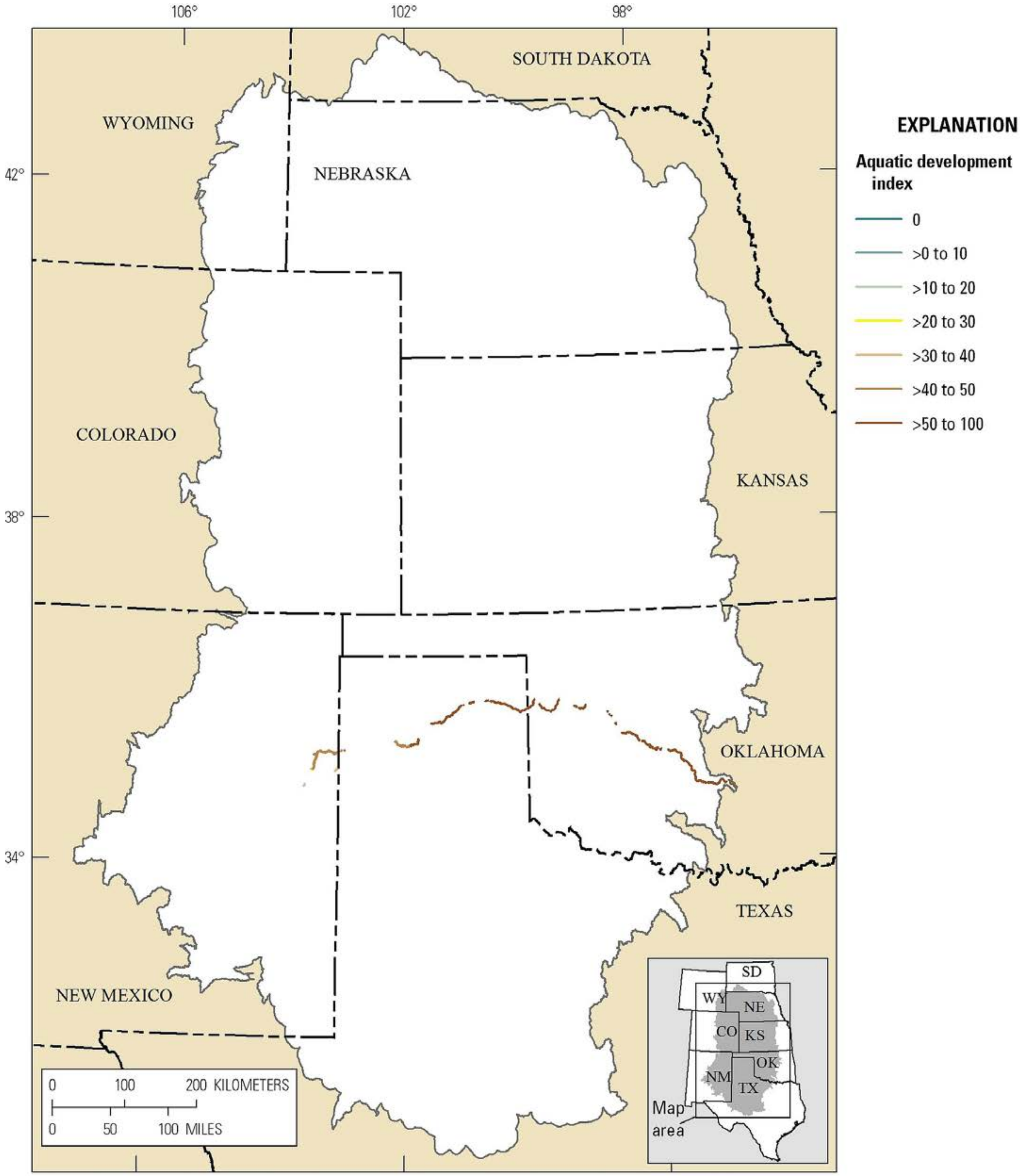


Figure 3–2. Aquatic development index for Arkansas River shiner baseline habitat in the Southern Great Plains. (Less than 1 percent of baseline habitat has aquatic development index scores ≤ 30; see fig. 3–3).

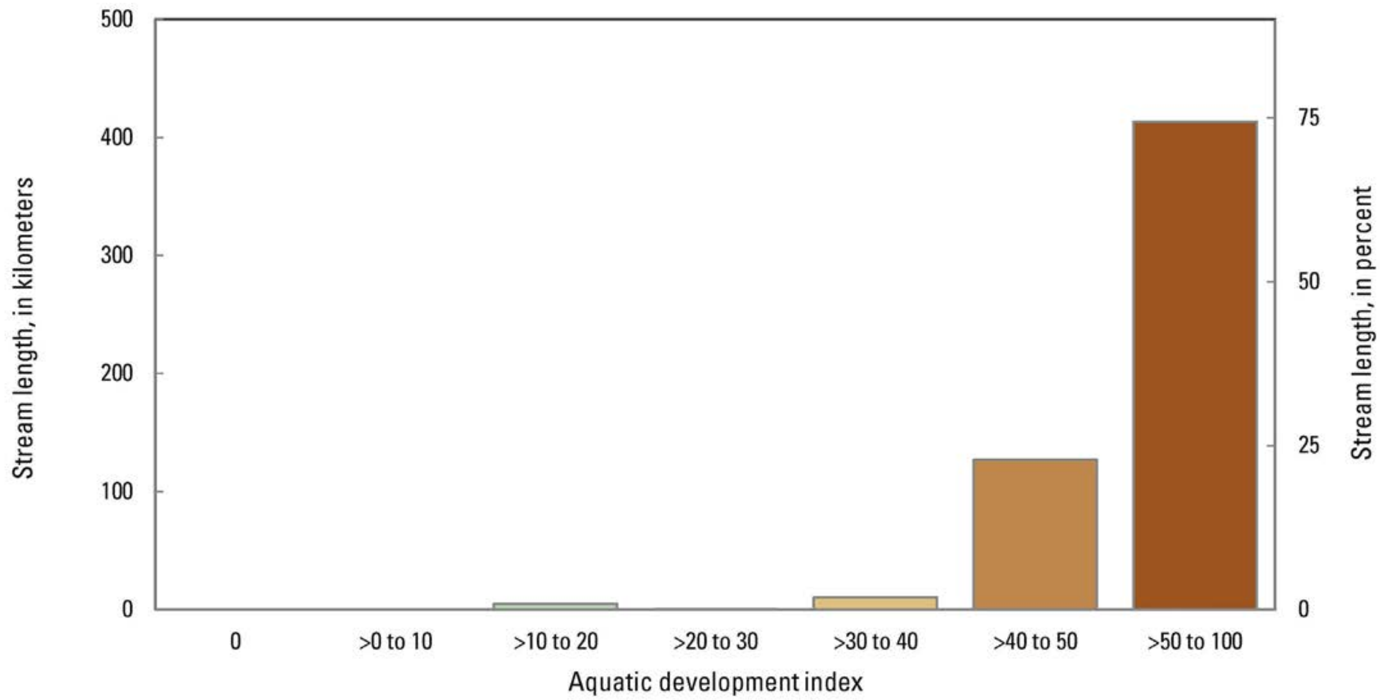


Figure 3–3. Total length of Arkansas River shiner baseline habitat by aquatic development index class in the Southern Great Plains.

How has development fragmented Arkansas River shiner habitat (fig. 3–4)?

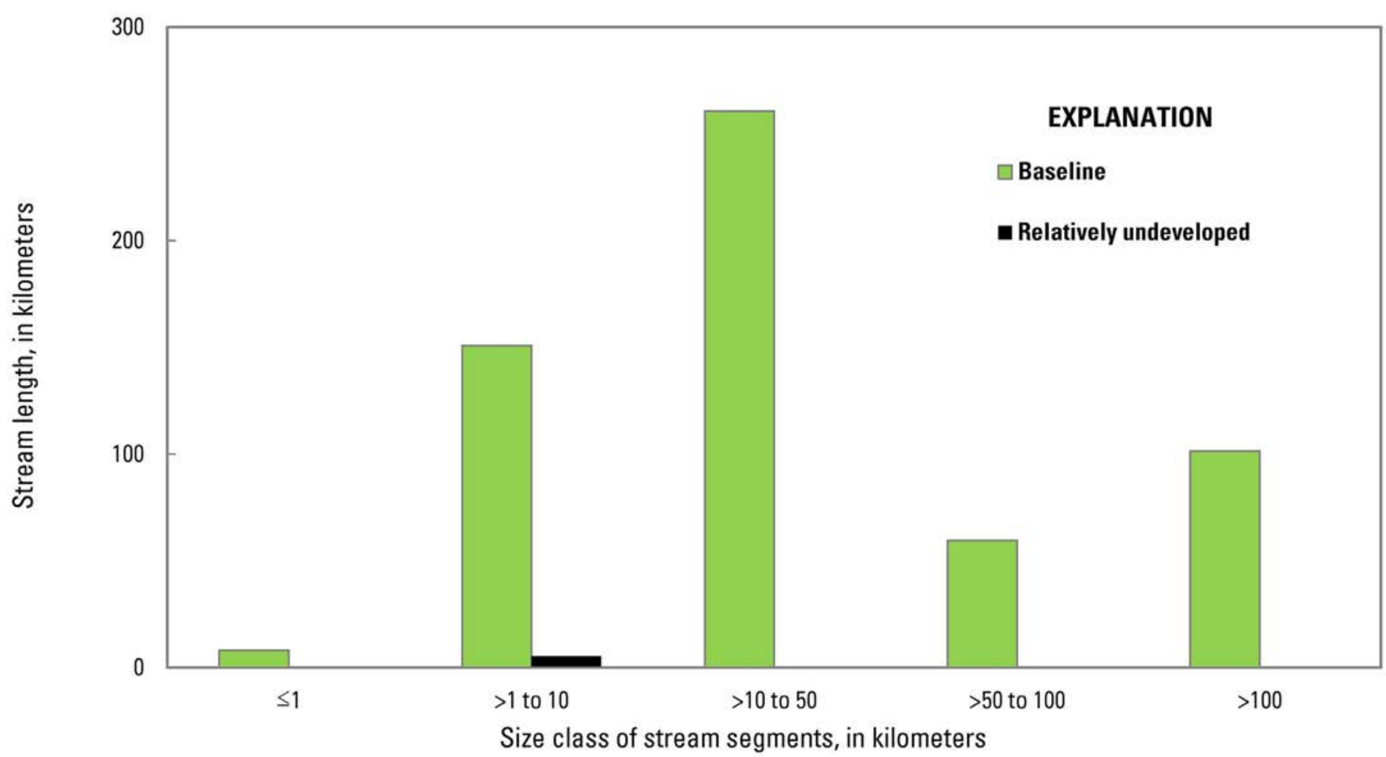


Figure 3–4. Total length of Arkansas River shiner habitat in the Southern Great Plains by stream-segment size class for baseline and relatively undeveloped conditions (aquatic development index score ≤20).

Where is Arkansas River shiner habitat with the highest overall landscape-level rank (fig. 3–5)?

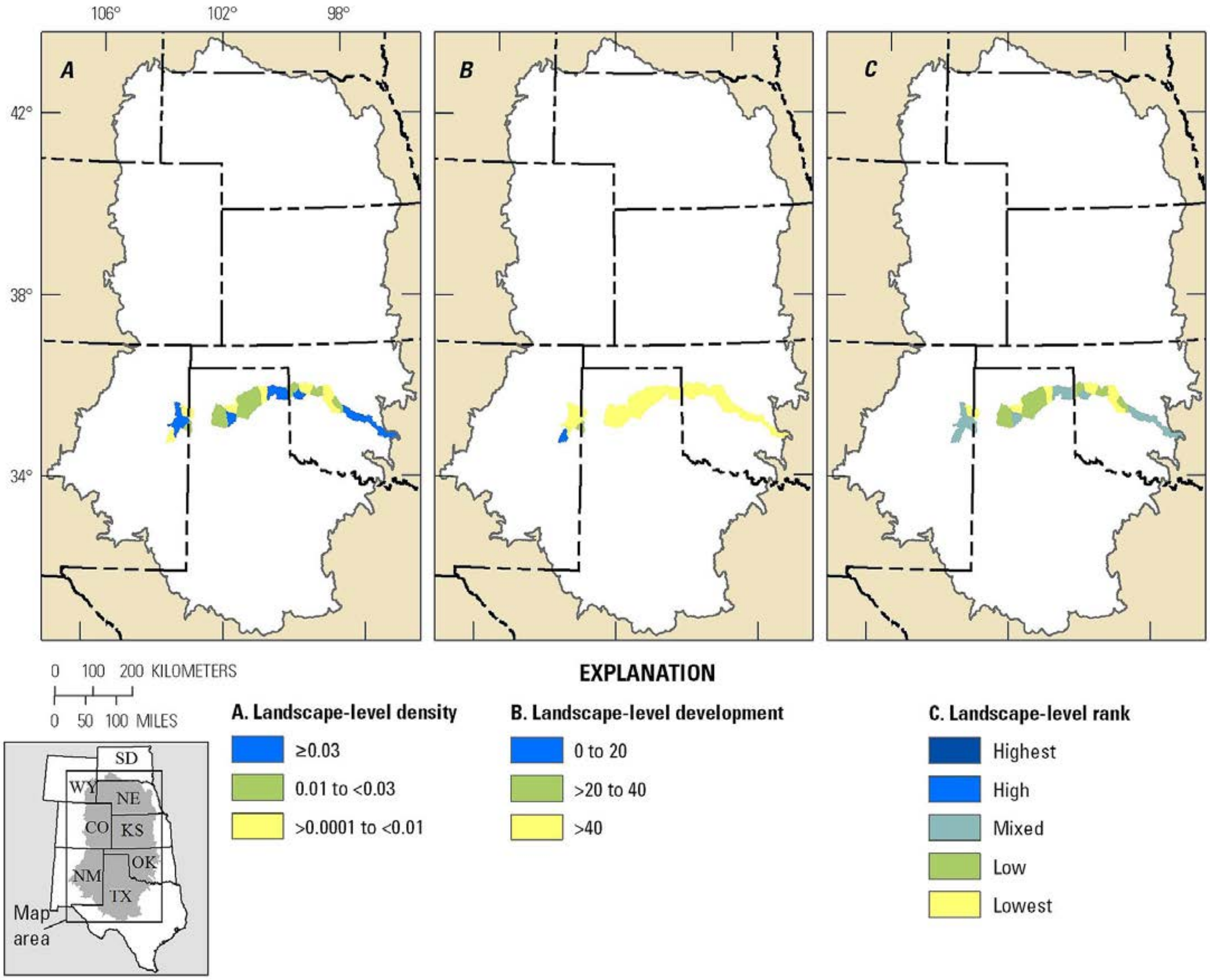


Figure 3–5. Landscape-level summaries for Arkansas River shiner habitat in the Southern Great Plains. Overall landscape-level rank (*C*) is derived from (*A*) landscape-level density and (*B*) landscape-level development, summarized by fifth-level watershed (see table 3–3). Highest overall landscape-level rank corresponds to the largest landscape-level density and the lowest landscape-level development. Lowest overall landscape-level rank corresponds to the smallest landscape-level density and highest landscape-level development. None of the watersheds had overall ranks of high or highest. Landscape-level ranks are not intended as standalone summaries and are best interpreted in conjunction with the geospatial datasets used to address core management questions.

Summary

- Baseline habitat for the Arkansas River shiner (fig. 3–1) was limited to its native range in the Canadian River and includes segments in New Mexico, Texas, and Oklahoma, which are all within the Canadian River catchment. Nonnative populations were not modeled by Worthington and others (2014).
- Less than 1 percent of baseline habitat is relatively undeveloped (ADI score ≤20), and 97 percent has very high development (ADI score >40) (figs. 3–2 and 3–3). The least developed areas are in New Mexico and the Texas panhandle.
- Approximately 45 percent of baseline habitat occurs in segments 10–50 km (6–31 mi) in length. The remaining relatively undeveloped segments are all shorter than 10 km (6 mi) (fig. 3–4). The maximum segment length was 101 km.
- The largest and least developed Arkansas River shiner habitat (overall landscape-level rank of “mixed”) is in Oklahoma, along the Texas-Oklahoma border, and in New Mexico (fig. 3–5*C*). None of the watersheds were assigned a high or very high overall landscape-level rank—all fifth-level watersheds with the highest landscape-level density of baseline habitat (fig. 3–5*A*) also have high levels of development (fig. 3–5*B*). These results indicate high vulnerability of this species in the Southern Great Plains.
- The broad-scale summaries provided by the REA are intended to be used in conjunction with local-level information on habitat conditions (Wood and others, 2017).

References Cited

Bestgen, K.R., Platania, S.P., Brooks, J.E., and Propst, D.L., 1989, Dispersal and life-history traits of *Notropis girardi* (Cypriniformes: Cyprinidae), introduced into the Pecos River, New Mexico: American Midland Naturalist, v. 122, p. 228–235.

Bonner, T.H., 2000, Life history and reproductive ecology of the Arkansas River shiner and peppered chub in the Canadian River, Texas and New Mexico: Lubbock, Tex., Texas Technology University, Ph.D. dissertation, 147 p.

Bonner, T.H., and Wilde, G.R., 2000, Changes in the Canadian River fish assemblage associated with reservoir construction: Journal of Freshwater Ecology, v. 15, p. 189–198.

Durham, B.W., and Wilde, G.R., 2006, Influence of stream discharge on reproductive success of a prairie stream fish assemblage: Transactions of the American Fisheries Society, v. 135, p. 1644–1653.

Falke, J.A., Fausch, K.D., Magelky, R., Aldred, A., Durnford, D.S., Riley, L.K., and Oad, R., 2010, The role of groundwater pumping and drought in shaping ecological futures for stream fishes in a dryland river basin of the western Great Plains, USA: Ecohydrology, v. 4, p. 682–697.

Felley, J.D., and Cothran, E.G., 1981, *Notropis bairdi* (Cyprinidae) in the Cimarron River, Oklahoma: Southwestern Naturalist, v. 25, p. 564.

Gido, K.B., Schaefer, J.F., Work, K., Lienesch, P.W., Marsh-Matthews, E., and Matthews, W.J., 1999, Effects of red shiner (*Cyprinella lutrensis*) on Red River pupfish (*Cyprinodon rubrofluviatilis*): Southwest Naturalist, v. 44, p. 287–295.

Hoagstrom, C.W., and Brooks, J.E., 2005, Distribution and status of Arkansas River shiner *Notropis girardi* and Rio Grande shiner *Notropis jemezanus*, Pecos River, New Mexico: Texas Journal of Science, v. 57, p. 35–58.

Larson, R.D., Jr., 1991, Present status and distribution of the Arkansas River shiner, *Notropis girardi* (Pisces: Cyprinidae), and possible cause for its decline: Stillwater, Okla., Oklahoma State University, M.S. thesis, 76 p.

Limbird, R.L., 1993, The Arkansas River—A changing river, *in* Hesse, L.W., Stalnaker, C.B., Benson, N.G., and Zuboy, J.R., eds., Restoration planning for the rivers of the Mississippi River ecosystem: U.S. Geological Survey Biological Report 19, p. 282–294.

Luttrell, G.R., Underwood, D.M., Fisher, W.L., and Pigg, J., 1995, Distribution of the Red River shiner, *Notropis bairdi*, in the Arkansas River drainage: Proceedings of Oklahoma Academy of Science, v. 75, p. 61–62.

Moore, G.A., 1944, Notes on the early life history of *Notropis girardi*: Copeia, v. 1944, p. 209–214.

Osborne, M.J., Diver, T.A., and Turner, T.F., 2013, Introduced populations as genetic reservoirs for imperiled species—A case study of the Arkansas River shiner (*Notropis girardi*): Conservation Genetics, v. 14, p. 637–647.

Perkin, J.S., and Gido, K.B., 2011, Stream fragmentation thresholds for a reproductive guild of Great Plains fishes: Fisheries, v. 36, p. 371–383.

Pigg, J., 1991, Decreasing distribution and current status of the Arkansas River shiner, *Notropis girardi*, in the rivers of Oklahoma and Kansas: Proceedings of Oklahoma Academy of Science, v. 71, p. 5–15.

Pigg, J., Gibbs, R., and Cunningham, K.K., 1999, Decreasing abundance of the Arkansas River shiner in the South Canadian River, Oklahoma: Proceedings of Oklahoma Academy of Science, v. 79, p. 7–12.

Polivka, K.M., 1999, The microhabitat distribution of the Arkansas River shiner, *Notropis girardi*—A habitat-mosaic approach: Environmental Biology of Fishes, v. 55, p. 265–278.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100.

Roberts, J.J., 2015, Arkansas River shiner, chap. 11 *of* Assal, T.J., Melcher, C.P., and Carr, N.B., eds., 2015, Southern Great Plains Rapid Ecoregional Assessment—Pre-assessment report: U.S. Geological Survey Open-File Report 2015–1003, p. 118–126, accessed July 13, 2017, at https://doi.org/10.3133/ofr20151003.

U.S. Fish and Wildlife Service, 1998, Endangered and threatened wildlife and plants—Final rule to list the Arkansas River Basin population of the Arkansas River shiner (*Notropis girardi*) as threatened [Final rule]: Federal Register, v. 63, no. 225, p. 64772–64799.

U.S. Fish and Wildlife Service, 2005, Endangered and threatened wildlife and plants—Final designation of critical habitat for the Arkansas River Basin population of the Arkansas River shiner (*Notropis girardi*) [Final rule]: Federal Register, v. 70, no. 197, p. 59808–59846.

Wilde, G.R., 2002, Threatened fishes of the world—*Notropis girardi* Hubbs and Ortenburger, 1929 (Cyprinidae): Environmental Biology of Fishes, v. 65, p. 98–98.

Wilde, G.R., Bonner, T.H., and Zwank, P.J., 2001, Diets of the Arkansas River shiner and peppered chub in the Canadian River, New Mexico and Texas: Journal of Freshwater Ecology, v. 16, p. 403–410.

Wood, J.A., Carter, S.K., Litshert, S.E., and Carr, N.B., 2017, Including broad-scale indicators in multiscale natural resource monitoring and assessment programs in the Bureau of Land Management, chap. 3 *of* Carter, S.K., Carr, N.B., Miller, K.H., and Wood, D.J.A., eds., Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, p. 15–37, accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Worthington, T.A., Brewer, S.K., Grabowski, T.B., and Mueller, J., 2014, Backcasting the decline of a vulnerable Great Plains reproductive ecotype—Identifying threats and conservation priorities: Global Change Biology, v. 20, p. 89–102.

Worthington, T.A., Echelle, A.A., Perkin, J.S., Mollenhauer, R., Farless, N., Dryer, J.J., Logue, D., and Brewer, S.K., 2018, The emblematic minnows of the North American Great Plains: A synthesis of threats and conservation opportunities—Fish and Fisheries, v. 9, p. 271–307.

Worthington, T.A., Zhang, T., Logue, D.R., Mittelstet, A.R., and Brewer, S.K., 2016, Landscape and flow metrics affecting the distribution of a federally-threatened fish—Improving management, model fit, and model transferability: Ecological Modelling, v. 342, p. 1–18.

Chapter 4. Ferruginous Hawk

Introduction

The ferruginous hawk (*Buteo regalis*) breeds and winters throughout most of the SGP. Overall, their populations generally have been stable or increasing over the past several decades (Bechard and Schmutz, 1995). Although local declines have been observed in eastern New Mexico, western Texas, and the Oklahoma panhandle (Sauer and others, 2011), such trends may be partly due to the dynamics of prey populations (Bechard and Schmutz, 1995). Within the SGP, the ferruginous hawk is listed as a species of management concern by the U.S. Fish and Wildlife Service (Region 6), a sensitive species by the Bureau of Land Management (BLM) (Travsky and Beauvais, 2005), a species of special concern in Colorado, and a species of conservation concern in Oklahoma.

Ferruginous hawks breed in open grasslands and shrublands (Bechard and Schmutz, 1995). Within the SGP, they are found in shortgrass, mixed-grass, and sand prairies (Bechard and Schmutz, 1995; McConnell and others, 2008). Their nests are typically on elevated ground and taller vegetation. The primary prey of ferruginous hawks in the SPG are burrowing mammals, but their diet also may include amphibians, reptiles, insects, and birds (Olendorff, 1993; Cook and others, 2003; Giovanni and others, 2007; Keeley, 2009). In many areas, prairie dogs (*Cynomys* spp.), which do not hibernate, are their primary prey item in winter (Cully, 1991; Bak and others, 2001), and ferruginous hawk numbers may decline locally following sylvatic plague (*Yersinia pestis*) outbreaks that reduce prairie dog abundance (Cully, 1991; Seery and Matiatos, 2000).

Ferruginous hawks winter in grasslands or along edges of cultivated fields in association with a high abundance of burrowing mammals, especially prairie dogs and pocket gophers (Geomyidae) (Bechard and Schmutz, 1995) and may move long distances, presumably in response to prey availability (Watson, 2003). The importance of ground squirrels and other burrowing mammals as prey is exhibited by the ferruginous hawk's unusual foraging technique of waiting in ambush at the mouth of active burrows (Bechard and Schmutz, 1995).

Ferruginous hawk. Photograph by Rick Bohn, U.S. Fish and Wildlife Service (Creative Commons Attribution 2.0 Generic).

Nesting ferruginous hawks appear to be more sensitive to human disturbance than other buteos (Bechard and others, 1990; Olendorff, 1993; Berry and others, 1998), but their response depends on the type of disturbance and the landscape context (Nordell and others, 2017). During the nonbreeding season, ferruginous hawks appear to tolerate higher levels of human activity than during the breeding season (Plumpton and Andersen, 1998). The effects of oil and gas development on nesting ferruginous hawks is equivocal and may depend on well density and landscape context (Smith and others, 2010; Keough and Conover, 2012; Coates and others, 2014; Keough and others, 2015; Wallace and others, 2016a, b; Wiggins and others, 2017). Nesting success of ferruginous hawks has been shown to decrease in relation to increasing densities of wind turbines (Kolar and Bechard, 2016); they also fly at altitudes that make them vulnerable to mortality from turbine blades (Johnson and others, 2000; Smallwood and Thelander, 2008; Johnson and Erickson, 2011). Ferruginous hawks also may be killed by collision with powerlines and electrocution by perching on utility structures (Olendorff, 1993). Agricultural conversion has led to reduced nest densities and breeding success (Houston and Bechard, 1984; White and Thurow, 1985; Woffinden and Murphy, 1989), but ferruginous hawks may forage along the edges of croplands where prey density can be higher (Schmutz, 1989; Zelenak and Rotella, 1997). Rodent-control programs may cause declines of ferruginous hawk numbers directly through poisoning and indirectly through reductions in prey populations. Additional background information on ferruginous hawks can be found in the SGP pre-assessment report (George, 2015).

Rapid Ecoregional Assessment Components Evaluated for the Ferruginous Hawk

The key ecological attributes and change agents addressed by core management questions for ferruginous hawk habitat include amount and distribution, landscape structure, and development (tables 4–1 and 4–2). Invasive woody species and climate change were evaluated for grassland communities (see Reese and others, 2017, chap. 4). Fire occurrence and potentially altered vegetation (including invasive herbaceous plants) were evaluated for the entire SGP (see Reese and others, 2017, chap. 3). Overall landscape-level ranking variables are summarized in table 4–3. The core and integrated management questions are listed in table 4–4.

Table 4–1. Key ecological attributes and associated indicators used to address core management questions for ferruginous hawks for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Amount and distribution	Total area	Baseline habitat (year round)[2]
Landscape structure	Patch size	Patch sizes for baseline habitat
Landscape dynamics	Fire occurrence	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]MaxEnt was used to predict baseline habitat. See chapter 2, "Methods Overview for Species," for methods and datasets used.

Table 4–2. Anthropogenic change agents and associated indicators used to address core management questions for ferruginous hawks for the Southern Great Plains Rapid Ecoregional Assessment.

[km, kilometer; mi, mile]

Attributes	Variables	Indicators[1]
Development	Terrestrial development index (TDI)	Percentage of baseline habitat in seven development classes based on a 2.5-km (1.55-mi) moving window
	Index of fragmentation	Patch sizes for relatively undeveloped[2] habitat
Invasive species	Potential for woody species expansion	See Reese and others (2017, chap. 4)
Climate change	Potential distribution of grasslands	See Reese and others (2017, chap. 4)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]Terrestrial development index score less than or equal to 2 percent.

Table 4–3. Landscape-level variables used to address the integrated management question for ferruginous hawks. Ranks for landscape-level habitat area and development were combined into an overall landscape-level rank for the Southern Great Plains Rapid Ecoregional Assessment.

[>, greater than; km, kilometer; mi, mile]

Landscape-level variables[1]	Description	Relative rank[2]		
		Lowest	Medium	Highest
Area	Percentage of habitat within a 5-km-radius (3.11-mi) moving window	>0–37.7	>37.7–65.8	>65.8
Development	Mean terrestrial development index (TDI) score for baseline habitat within a 5km-radius (3.11-mi) moving window	0–2	>2–10	>10

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address integrated management questions.

[2]Ranking breakpoints for area of baseline habitat were determined from equal subsets of the data. Ranking breakpoints for terrestrial development index scores were standardized for all terrestrial conservation elements.

Table 4–4. Management questions addressed for ferruginous hawks for the Southern Great Plains Rapid Ecoregional Assessment.

Core management questions[1]	Results
What is the distribution of baseline habitat for ferruginous hawks?	Figure 4−1
Where does existing development pose the greatest threat to ferruginous hawk habitat, and where are the large, relatively undeveloped areas?	Figures 4−2 and 4−3
How has development fragmented ferruginous hawk habitat?	Figures 4−4 and 4−5
Integrated management question[2]	**Results**
Where is ferruginous hawk habitat with the highest overall landscape-level rank?	Figure 4−6

[1]See Reese and others (2017, chap. 11) for management questions that could not be addressed.

[2]See table 4−3.

Management Questions and Results

What is the distribution of baseline habitat for ferruginous hawks (fig. 4–1)?

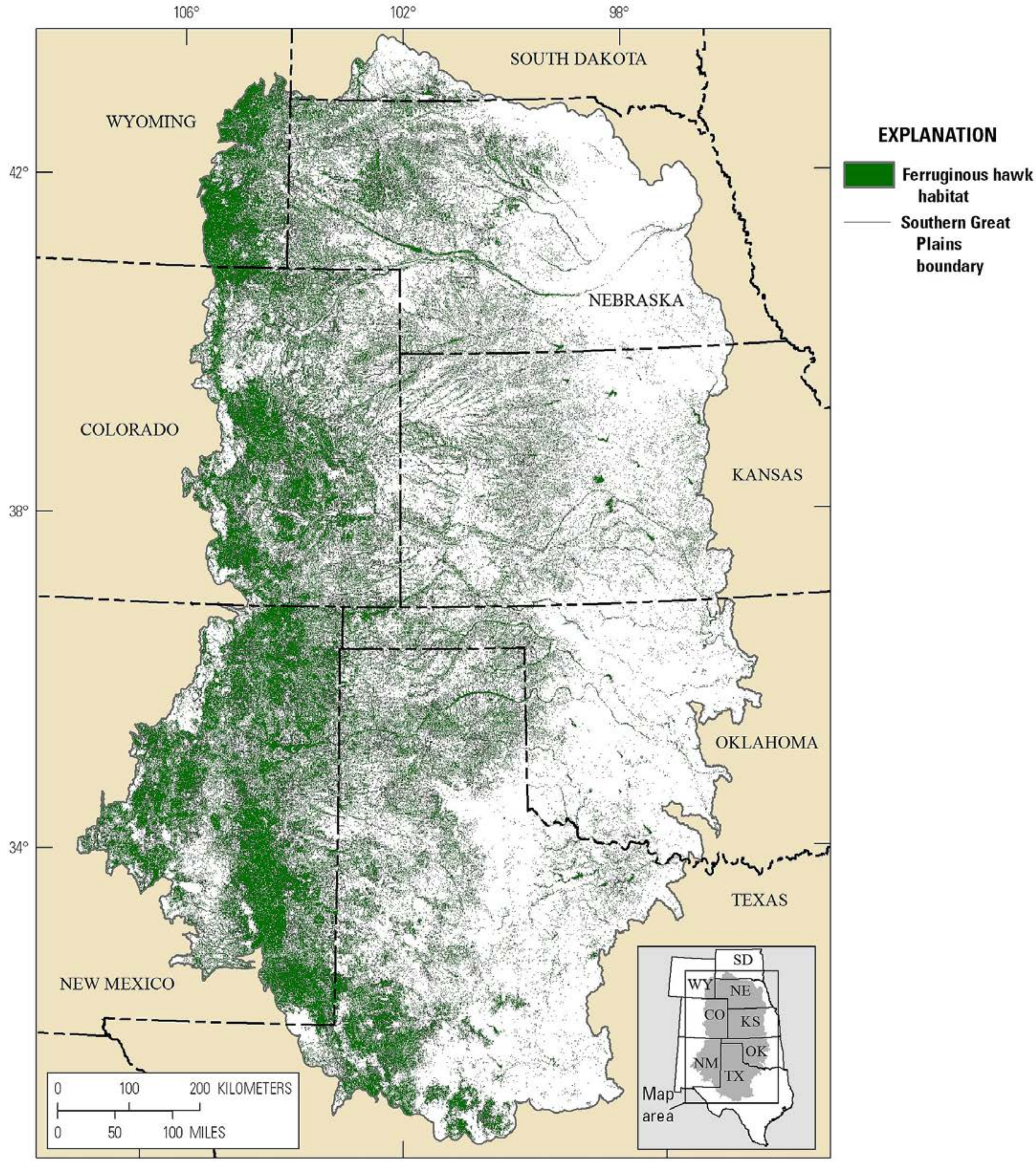


Figure 4–1. Predicted distribution of baseline habitat for ferruginous hawks in the Southern Great Plains.

Where does existing development pose the greatest threat to ferruginous hawk habitat, and where are the large, relatively undeveloped areas (figs. 4–2 and 4–3)?

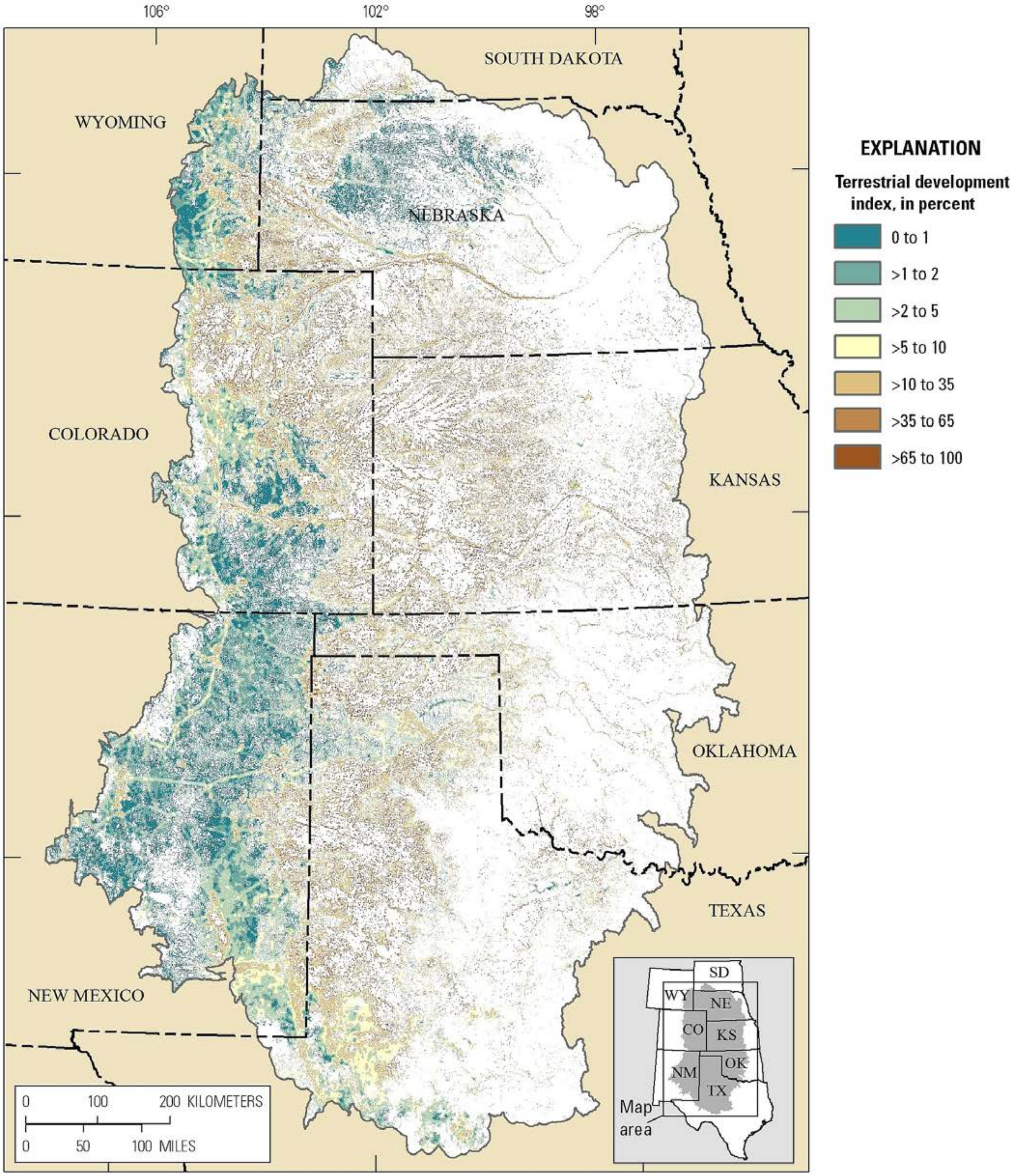


Figure 4–2. Terrestrial development index for ferruginous hawk baseline habitat in the Southern Great Plains.

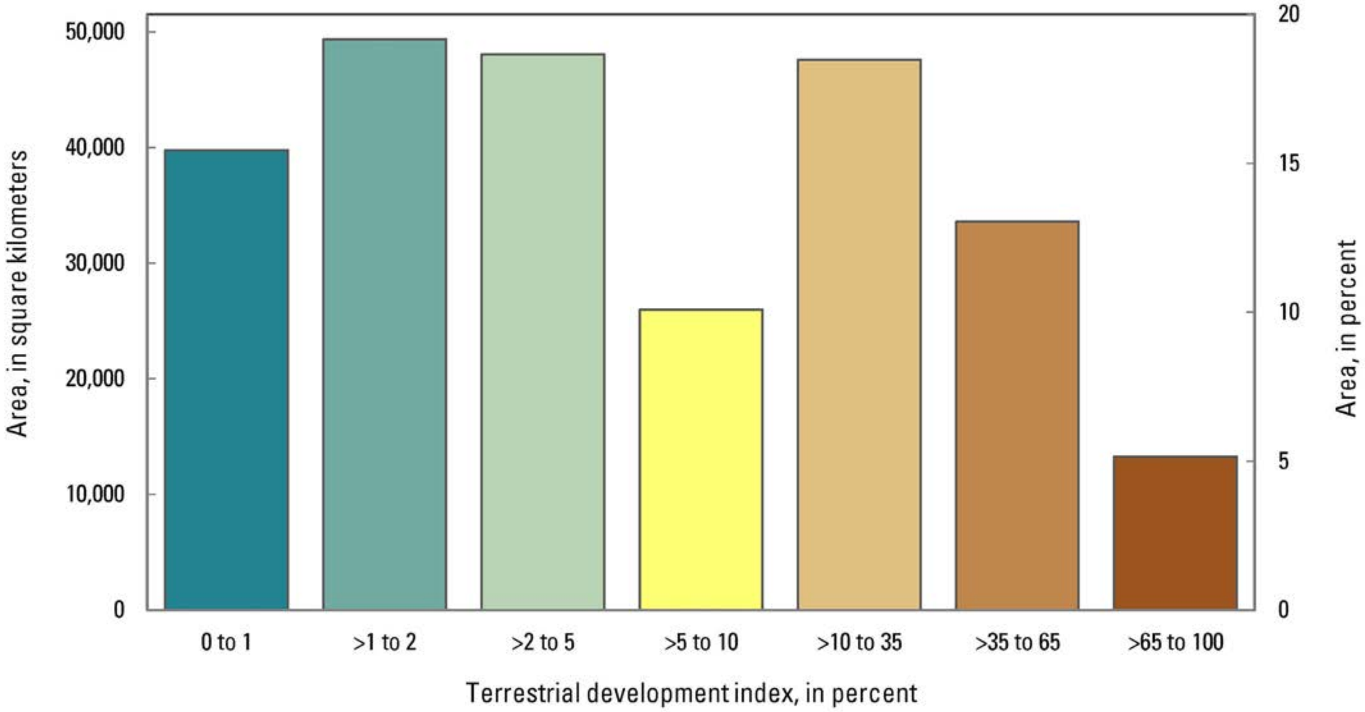


Figure 4–3. Area of ferruginous hawk baseline habitat by terrestrial development index class in the Southern Great Plains.

How has development fragmented ferruginous hawk habitat (figs. 4–4 and 4–5)?

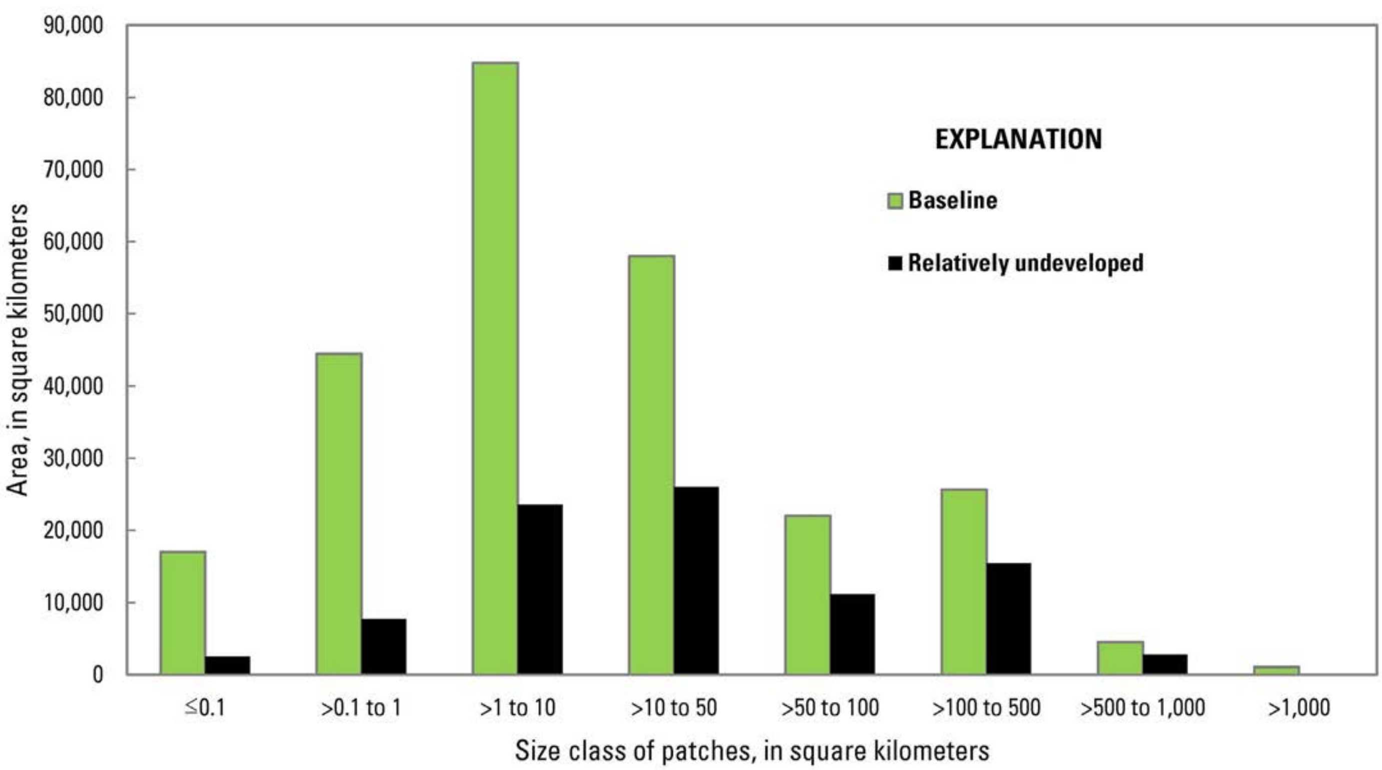


Figure 4–4. Area of ferruginous hawk habitat in the Southern Great Plains by patch size class for baseline and relatively undeveloped conditions (terrestrial development index score ≤2 percent).

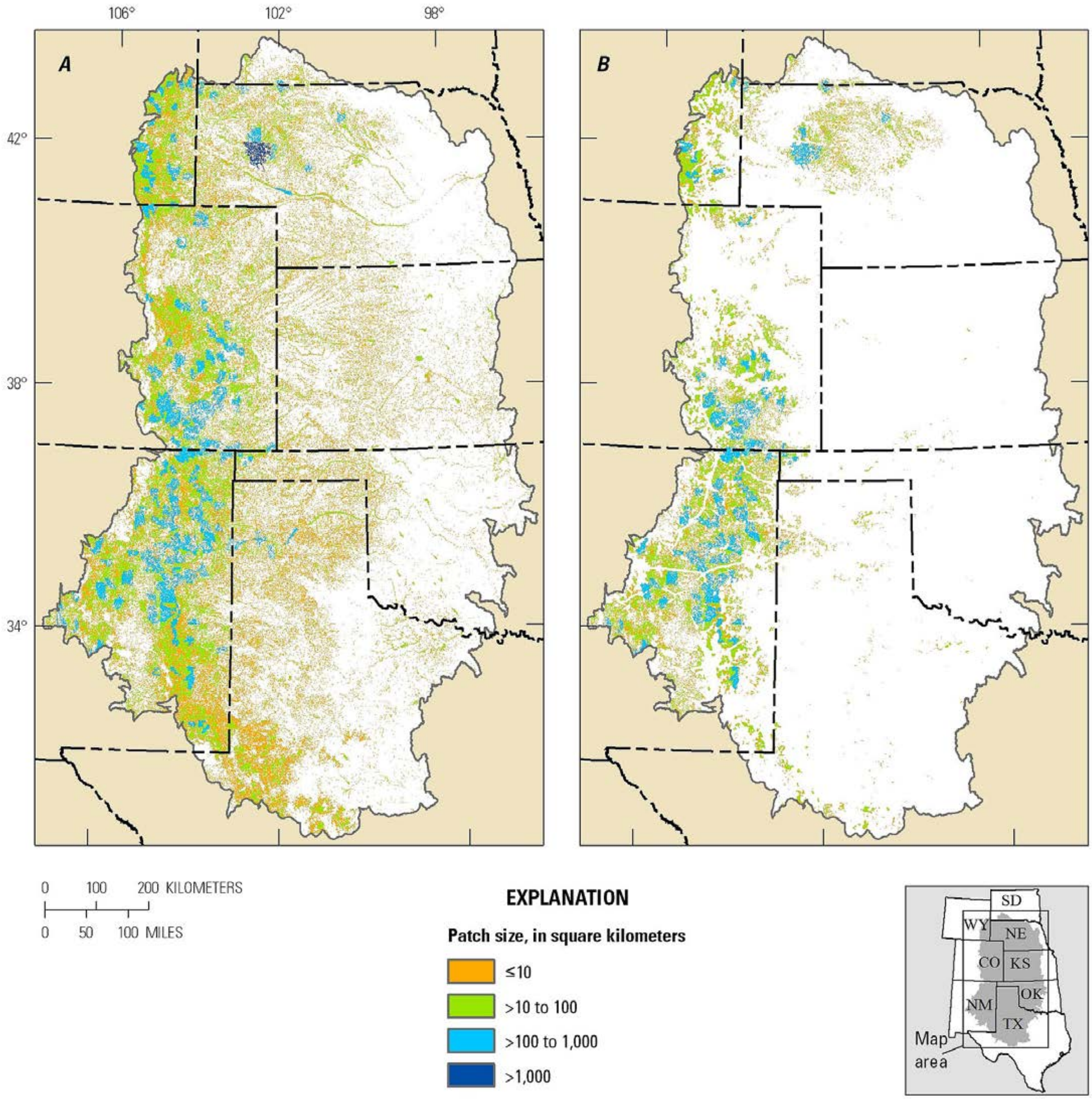


Figure 4–5. Patch size of ferruginous hawk habitat in the Southern Great Plains. *A,* Baseline habitat. *B,* Relatively undeveloped habitat (terrestrial development index score ≤2 percent).

Where is ferruginous hawk habitat with the highest overall landscape-level rank (fig. 4–6)?

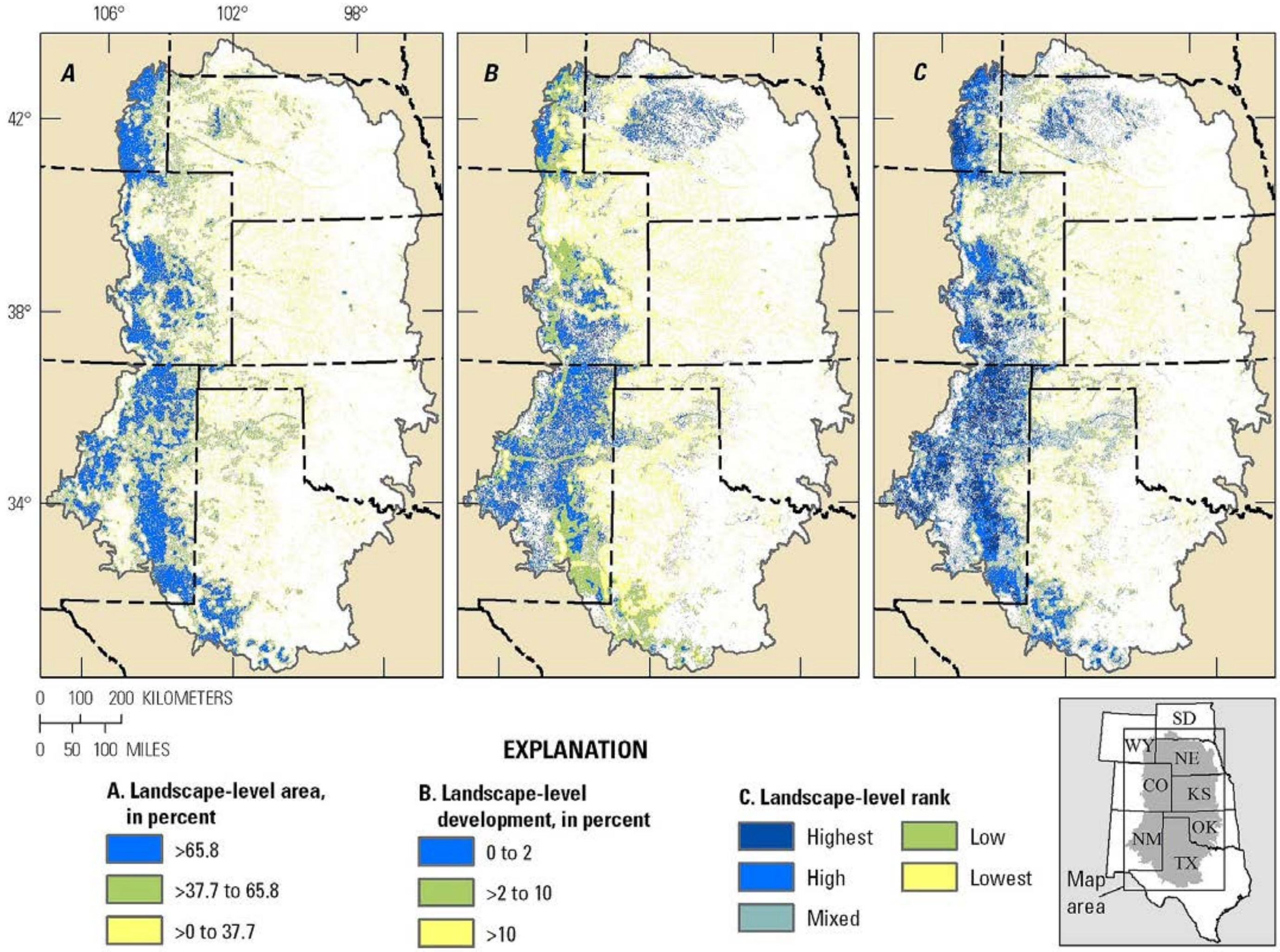


Figure 4–6. Landscape-level summaries for ferruginous hawk habitat in the Southern Great Plains. Overall landscape-level rank (*C*) is derived from (*A*) landscape-level habitat area and (*B*) landscape-level development, summarized by a 5-kilometer-radius (3.11-mile) moving window (see table 4–3). Highest overall landscape-level rank corresponds to the largest landscape-level area and the lowest landscape-level development. Lowest overall landscape-level rank corresponds to the smallest landscape-level area and highest landscape-level development. Landscape-level ranks are not intended as standalone summaries and are best interpreted in conjunction with the geospatial datasets used to address core management questions.

Summary

- Baseline habitat for ferruginous hawks occurs throughout much of the SGP but is concentrated along the western side of the region. There are more than 257,000 km^2 (99,228 mi^2) of baseline habitat in the SGP (fig. 4–1).
- Ferruginous hawk habitat with the lowest development is located predominantly in the western and northern portions of the SGP (fig. 4–2). Nearly 35 percent of their habitat is relatively undeveloped (TDI score ≤2 percent), and 19 percent has low development (TDI scores 2–5 percent) (fig. 4–3). Approximately 18 percent of their habitat, however, has very high development (TDI scores >35 percent).
- Ferruginous hawk habitat is highly fragmented throughout Texas, Oklahoma, Kansas, and southern Nebraska (figs. 4–4 and 4–5*A*). The largest remaining patch of baseline habitat is located in the Sand Hills of Nebraska (fig. 4–5*B*).
- The largest, most intact areas (the highest overall landscape-level rank) are in the western extent of the shortgrass prairie (fig. 4–6*C*) where there has been less conversion to croplands than in other areas of the region (Reese and others, 2017).
- The broad-scale summaries provided by the REA are intended to be used in conjunction with local-level information on habitat conditions (Wood and others, 2017).

References Cited

Bak, J.M., Boykin, K.G., Thompson, B.C., and Daniel, D.L., 2001, Distribution of wintering ferruginous hawks (*Buteo regalis*) in relation to black-tailed prairie dog (*Cynomys ludovicianus*) colonies in southern New Mexico and northern Chihuahua: Journal of Raptor Research, v. 35, p. 124–129.

Bechard, M.J., Knight, R.L., Smith, D.G., and Fitzner, R.E., 1990, Nest sites and habitats of sympatric hawks (*Buteo* spp.) in Washington: Journal of Field Ornithology, v. 61. p. 159–170.

Bechard, M.J., and Schmutz, J.K., 1995, Ferruginous hawk (*Buteo regalis*), *in* Poole, A., ed., The birds of North America: Ithaca, N.Y., Cornell Lab of Ornithology, no. 172, accessed February 2014 at http://bna.birds.cornell.edu/bna/species/172.

Berry, M.E., Bock, C.E., and Haire, S.L., 1998, Abundance of diurnal raptors on open space grasslands in an urbanized landscape: Condor, v. 100, p. 601–608.

Coates, P.S., Howe, K.B., Casazza, M.L., and Delehanty, D.J., 2014, Landscape alterations influence differential habitat use of nesting buteos and ravens within sagebrush ecosystem—Implications for transmission line development: Condor, v. 116, p. 341–356.

Cook, R.R., Cartron, J.L., and Polechla, P.J., Jr., 2003, The importance of prairie dogs to nesting ferruginous hawks in grassland ecosystems: Wildlife Society Bulletin, v. 31, p. 1073–1082.

Cully, J.F., 1991, Response of raptors to a reduction of a Gunnison's prairie dog population by plague: American Midland Naturalist, v. 125, p. 140–149.

George, T.L., 2015, Ferruginous hawk, chap. 13 *of* Assal, T.J., Melcher, C.P., and Carr, N.B., eds., 2015, Southern Great Plains Rapid Ecoregional Assessment—Pre-assessment report: U.S. Geological Survey Open-File Report 2015–1003, p. 104–117, accessed July 13, 2017, at https://doi.org/10.3133/ofr20151003.

Giovanni, M.D., Boal, C.W., and Whitlaw, H.A., 2007, Prey use and provisioning rates of breeding ferruginous and Swainson's hawks on the Southern Great Plains, USA: Wilson Journal of Ornithology, v. 119, p. 558–569.

Houston, C.S., and Bechard, M.J., 1984, Decline of the ferruginous hawk in Saskatchewan: American Birds, v. 38, p. 166–170.

Johnson, G.D., and Erickson, W.P., 2011, Avian, bat and habitat cumulative impacts associated with wind energy development in the Columbia Plateau Ecoregion of eastern Washington and Oregon: Cheyenne, Wyo., Western Eco-Systems Technology, Inc., 40 p., accessed June 1, 2014, at http://www.klickitatcounty.org/planning/filesHtml/200408-EOZ-EIS/cummulative2.pdf.

Johnson, G.D., Young, D.P., Jr., Erickson, W.P., Derby, C.E., Strickland, M.D., and Kern, J.W., 2000, Final report—Wildlife monitoring studies—SeaWest WindPower project, Carbon County, Wyoming, 1995–1999: Cheyenne, Wyo., Western EcoSystems Technology, Inc., 195 p.

Keeley, W.H., 2009, Diet and behavior of ferruginous hawks nesting in two grasslands in New Mexico with differing anthropogenic alteration: Boise, Idaho, Boise State University, M.S. thesis, 294 p.

Keough, H.L., and Conover, M.R., 2012, Breeding-site selection by ferruginous hawks within Utah's Uintah Basin: Journal of Raptor Research, v. 46, p. 378–388.

Keough, H.L., Conover, M.R., and Roberts, A.J., 2015, Factors influencing the reproductive success of ferruginous hawks in the Uintah Basin, Utah: Journal of Raptor Research, v. 49, p. 161–173.

Kolar, P.S., and Bechard, M.J., 2016, Wind energy, nest success, and post-fledging survival of *Buteo* hawks: Journal of Wildlife Management, v. 80, p. 1242–1255.

McConnell, S., O'Connell, T.J., and Leslie, D.M., Jr., 2008, Land cover associations of nesting territories of three sympatric *Buteos* in shortgrass prairie: Wilson Journal of Ornithology, v.120, p. 708–716.

Nordell, C.J., Wellicome, T.I., and Bayne, E.M., 2017, Flight initiation by ferruginous hawks depends on disturbance type, experience, and the anthropogenic landscape: PLoS ONE, v. 12, no. 5, article e0177584, accessed October 20, 2017, at https://doi.org/10.1371/journal.pone.0177584.

Olendorff, R.R., 1993, Status, biology, and management of ferruginous hawks—A review: Boise, Idaho, Bureau of Land Management, Raptor Research and Technical Assistance Center, 84 p.

Plumpton, D.L., and Andersen, D.E., 1998, Anthropogenic effects on winter behavior of ferruginous hawks: Journal of Wildlife Management, v. 62, p. 340–346.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100.

Sauer, J.R., Hines, J.E., Fallon, J.E., Pardieck, K.L., Ziolkowski, D.J., Jr., and Link, W.A., 2011, The North American breeding bird survey, results and analysis, 1966–2010 (version 12.07.2011): Laurel, Md., U.S. Geological Survey, Patuxent Wildlife Research Center, accessed February 2014 at http://www.mbr-pwrc.usgs.gov/bbs/bbs2010.html.

Schmutz, J.K., 1989, Hawk occupancy of disturbed grasslands in relation to models of habitat selection: Condor, v. 91, p. 362–371.

Seery, D.B., and Matiatos, D.J., 2000, Response of wintering buteos to plague epizootics in prairie dogs: Western North American Naturalist, v. 60, p. 420–425.

Smallwood, K.S., and Thelander, C., 2008, Bird mortality in the Altamont Pass Wind Resource Area, California: Journal of Wildlife Management, v. 72, no. 1, p. 215–223.

Smith, J.P., Slater, S.J., and Neal, M.C., 2010, An assessment of the effects of oil and gas field activities on nesting raptors in the Rawlins, Wyoming and Price, Utah field offices of the Bureau of Land Management: U.S. Bureau of Land Management Technical Note 433, 63 p., accessed February 2014 at http://www.blm.gov/pgdata/etc/medialib/blm/wo/blm_library/tech_notes.Par.76658.File.dat/TN_433.pdf.

Travsky, A., and Beauvais, G.P., 2005, Species assessment for the ferruginous hawk (*Buteo regalis*) in Wyoming: Cheyenne, Wyo., U.S. Bureau of Land Management, Wyoming State Office, 39 p., accessed February 2014 at http://www.blm.gov/pgdata/etc/medialib/blm/wy/wildlife/animal-assessmnts.Par.1082.File.dat/FerruginousHawk.pdf.

Wallace, Z.P, Kennedy, P.L., Squires, J.R., Oakleaf, R.J., Olson, L.E., and Dugger, K.M., 2016a, Re-occupancy of breeding territories by ferruginous hawks in Wyoming—Relationships to environmental and anthropogenic factors: PLoS ONE, v. 11, no. 4, article e0152977, accessed October 20, 2017, at http://journals.plos.org/plosone/article?id=10.1371/journal.pone.0152977.

Wallace, Z.P, Kennedy, P.L., Squires, J.R., Olson, L.E., and Oakleaf, R.J., 2016b, Human-made structures, vegetation, and weather influence ferruginous hawk breeding performance: Journal of Wildlife Management, v. 80, p. 78–90.

Watson, J.W., 2003, Migration and winter ranges of ferruginous hawks from Washington: Washington Department of Fish and Wildlife, 50 p.

White, C.M., and Thurow, T.L., 1985, Reproduction of ferruginous hawks exposed to controlled disturbance: Condor, v. 87, p. 14–22.

Wiggins, D.A., Grzybowski, J.A., and Schnell, G.D., 2017, Ferruginous hawk demography in areas differing in energy extraction activity: Journal of Wildlife Management, v. 81, p. 337–341.

Woffinden, N.D., and Murphy, J.R., 1989, Decline of a ferruginous hawk population—A 20-year summary: Journal of Wildlife Management, v. 53, p. 1127–1132.

Wood, J.A., Carter, S.K., Litshert, S.E., and Carr, N.B., 2017, Including broad-scale indicators in multiscale natural resource monitoring and assessment programs in the Bureau of Land Management, chap. 3 *of* Carter, S.K., Carr, N.B., Miller, K.H., and Wood, D.J.A., eds., Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, p. 15–37, accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Zelenak, J.R., and Rotella, J.J., 1997, Nest success and productivity of ferruginous hawks in northern Montana: Canadian Journal of Zoology, v. 75, p. 1035–1041.

Chapter 5. Lesser Prairie-Chicken

Introduction

The distribution of the lesser prairie-chicken (*Tympanuchus pallidicinctus*) is entirely restricted to the SGP. Despite the relatively recent range expansion documented in northern Kansas, it is generally assumed to occupy only 10–15 percent of its estimated historical range (Hagen and Giesen, 2005; Rodgers, 2016). Habitat loss, fragmentation, and degradation resulting from changing land uses have been listed as the primary causes of population declines (Hagen and Giesen, 2005; Rodgers, 2016). The remaining small and fragmented populations are extremely vulnerable to severe drought, which is believed to have caused a 50 percent decline in population size in 2013 (McDonald and others, 2014; Robinson and others, 2016; Ross and others, 2016). In early 2014, the lesser prairie-chicken (hereafter referred to as LPC) was listed as a federally threatened species under the Endangered Species Act, but a court ruling in 2015 vacated this listing decision; the status of the species is again under review (U.S. Fish and Wildlife Service, 2016; Van Pelt, 2016). A range-wide conservation plan provides many conservation measures intended to preclude the need to list the LPC as threatened (Van Pelt and others, 2013)

Habitat for the LPC is characterized by mid-height and tall grasses mixed with dwarf shrublands (particularly sand shinnery oak [*Quercus havardii*] and sand sagebrush [*Artemisia filifolia*]) growing in sandy soils (Woodward and others, 2001; Hagen and Giesen, 2005; Haukos and Zavaleta, 2016). Their habitat is found in close proximity to lek sites, which are minimally vegetated and often somewhat higher in elevation than the surrounding terrain (Hagen and Giesen, 2005; Haukos and Zavaleta, 2016). Males and females exhibit strong lek fidelity and stable home ranges (Hagen and Giesen, 2005).

Landscape dynamics resulting from fire, herbivory, and climate can help to maintain their habitat, but LPCs are vulnerable to extreme episodic events because of their limited distribution (Woodward and others, 2001; Fuhlendorf and others, 2002; Garton and others, 2016; Haukos and Zavaleta, 2016; Ross and others, 2016). Historically, frequent fires helped to control expansion of woody plants and create mosaics of vegetation types (Boyd and Bidwell, 2001), and fire can help maintain existing lek sites (Hagen and others, 2004; Van Pelt and others, 2013). Pervasive fires, however, could decrease cover and reduce production of sand shinnery oak acorns, which can be an important food source during mast years (Fuhlendorf and others, 2002; Hagen and others, 2004; Boyd and Bidwell, 2001). Native herbivores contributed to landscape heterogeneity and dynamics, but their effects on LPC habitat is poorly understood (Patten and others, 2005). Severe or prolonged drought may diminish vegetation cover and food resources critical for brood survival (Jamison and others, 2002; Fields and others, 2006); consequently, drought can reduce nesting success if it occurs during the egg laying or incubation period (Grisham and Boal, 2015; Robinson and others, 2016).

Lesser prairie-chicken. Photograph by the National Resources Conservation Service (Creative Commons Attribution 2.0 Generic).

Agricultural conversion, inappropriate range management, and energy development have contributed to habitat degradation and loss (Hagen and others, 2004; Boal and Haukos, 2016). Lekking males and nesting hens generally avoid cropland edges and disturbances associated with cultivation (Crawford and Bolen, 1976). Small-grain crops can provide foraging habitat, especially in winter or during drought (Applegate and Riley, 1998), but they are not thought to be essential. The birds may sometimes use Conservation Reserve Program lands for lekking, nesting, brood rearing, and winter cover (Giesen, 2000; Fields and others, 2006; Hagen and others, 2016; Rodgers, 2016), particularly where seed mixes include native forbs and grasses (Silvy and others, 2004). Although short-duration grazing may sometimes improve habitat quality, some livestock management practices can degrade habitats (Hagen and others, 2004; Patten and others, 2005; Fuhlendorf and others, 2012; Van Pelt and others, 2013), including the widespread use of herbicides to eliminate sand shinnery oak (to increase rangeland productivity) (Van Pelt and others, 2013; Fritts and others, 2016). LPCs generally avoid areas with active oil, gas, and wind energy development (Pitman and others, 2005; Pruett and others, 2009; Hagen and others, 2011; Jarnevich and Laubhan, 2011; Bartuszevige and Daniels, 2016; Jarnevich and others, 2016). Expansion of woody plant species is also a concern (Hagen and others, 2004; Fuhlendorf and others, 2017). Additional background information on the lesser prairie-chicken can be found in the SGP pre-assessment report (Melcher, 2015).

Rapid Ecoregional Assessment Components Evaluated for Lesser Prairie-Chicken

The key ecological attributes and change agents addressed by core management questions for LPC habitat include the amount and distribution, landscape structure, and development (tables 5–1 and 5–2). Invasive woody species and climate change were evaluated for grassland communities (see Reese and others, 2017, chap. 4). Fire occurrence and potentially altered vegetation (including invasive herbaceous plants) were evaluated for the entire SGP (see Reese and others [2017, chap. 3]). Overall landscape-level ranking variables are summarized in table 5–3. The core and integrated management questions are listed in table 5–4.

Table 5–1. Key ecological attributes and associated indicators used to address core management questions for lesser prairie-chickens for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Amount and distribution	Total area	Baseline habitat (breeding)[2]
Landscape structure	Patch size	Patch sizes for baseline habitat
Landscape dynamics	Fire occurrence	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]MaxEnt was used to predict baseline habitat (Jarnevich and others, 2016). See chapter 2, "Methods Overview for Species," for methods and datasets used.

Table 5–2. Anthropogenic change agents and associated indicators used to address core management questions for lesser prairie-chickens for the Southern Great Plains Rapid Ecoregional Assessment.

[km, kilometer; mi, mile]

Attributes	Variables	Indicators[1]
Development	Terrestrial development index (TDI)	Percentage of baseline habitat in seven development classes based on a 2.5-km (1.55-mi) moving window
	Index of fragmentation	Patch sizes for relatively undeveloped[2] habitat
Invasive species	Potential for woody species expansion	See Reese and others (2017, chap. 4)
Climate change	Potential distribution of grasslands	See Reese and others (2017, chap. 4)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]Terrestrial development index score less than or equal to 2 percent.

Table 5–3. Landscape-level variables used to address the integrated management question for lesser prairie-chickens. Ranks for landscape-level habitat area and development were combined into an overall landscape-level rank for the Southern Great Plains Rapid Ecoregional Assessment.

[>, greater than; km, kilometer; mi, mile]

Landscape-level variables[1]	Description	Relative rank[2]		
		Lowest	Medium	Highest
Area	Percentage of habitat within a 5-km-radius (3.11-mi) moving window	>0–44.7	>44.7–66.0	>66.0
Development	Mean terrestrial development index (TDI) score for habitat within a 5km-radius (3.11-mi) moving window	0–2	>2–10	>10

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address integrated management questions.

[2]Ranking breakpoints for area of baseline habitat were determined from equal subsets of the data. Ranking breakpoints for terrestrial development index scores were standardized for all terrestrial conservation elements.

Table 5–4. Management questions addressed for lesser prairie-chickens for the Southern Great Plains Rapid Ecoregional Assessment.

Core management questions[1]	Results
What is the distribution of baseline habitat for lesser prairie-chickens?	Figure 5−1
Where does existing development pose the greatest threat to lesser prairie-chicken habitat, and where are the large, relatively undeveloped areas?	Figures 5−2 and 5−3
How has development fragmented lesser prairie-chicken habitat?	Figures 5−4 and 5−5
Integrated management question[2]	**Results**
Where is lesser prairie-chicken habitat with the highest overall landscape-level rank?	Figure 5−6

[1]See Reese and others (2017, chap. 11) for management questions that could not be addressed.

[2]See table 5−3.

Management Questions and Results

What is the distribution of baseline habitat for lesser prairie-chickens (fig. 5–1)?

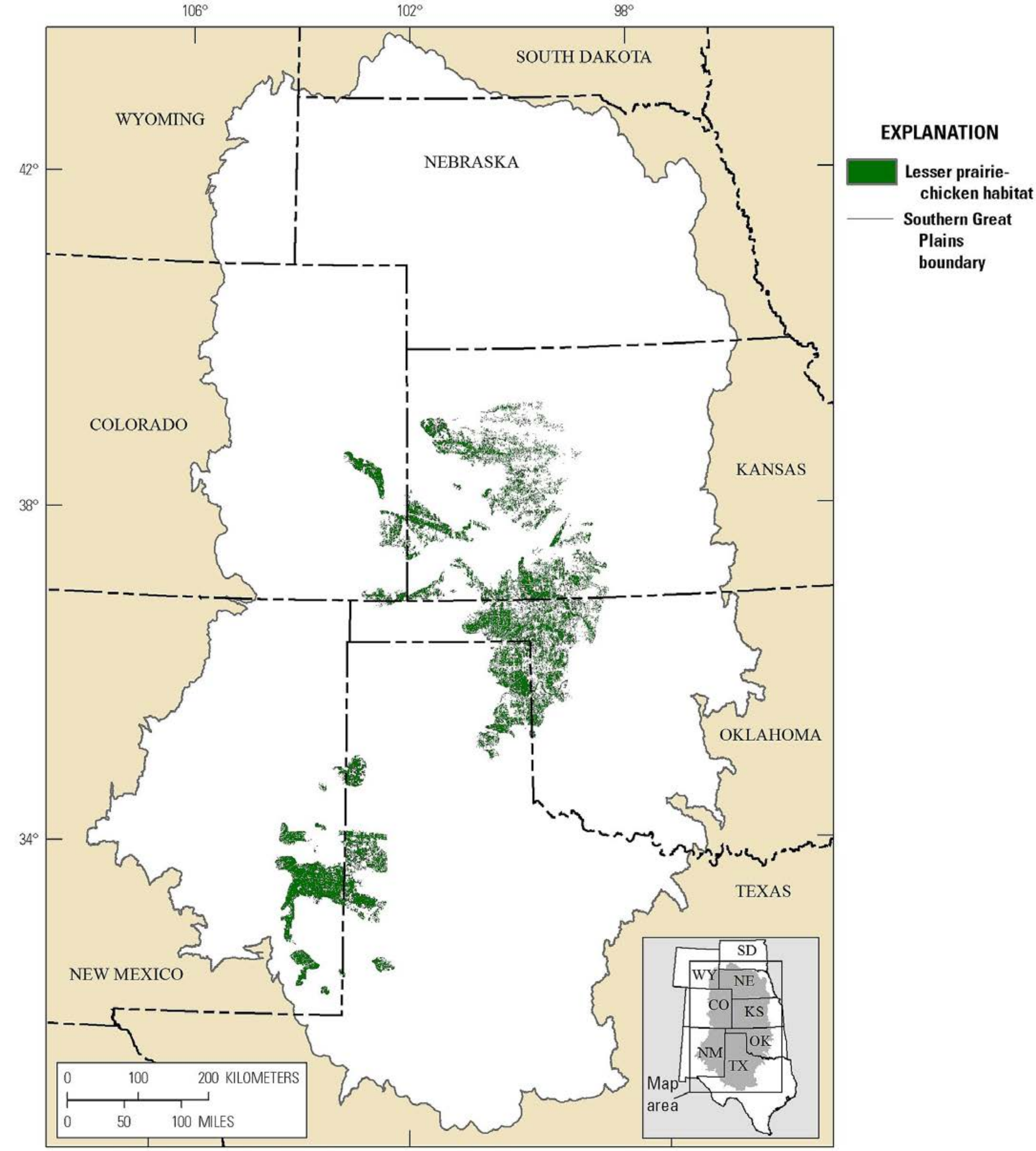


Figure 5–1. Predicted distribution of baseline habitat for lesser prairie-chickens in the Southern Great Plains.

Where does existing development pose the greatest threat to lesser prairie-chicken habitat, and where are the large, relatively undeveloped areas (figs. 5–2 and 5–3)?

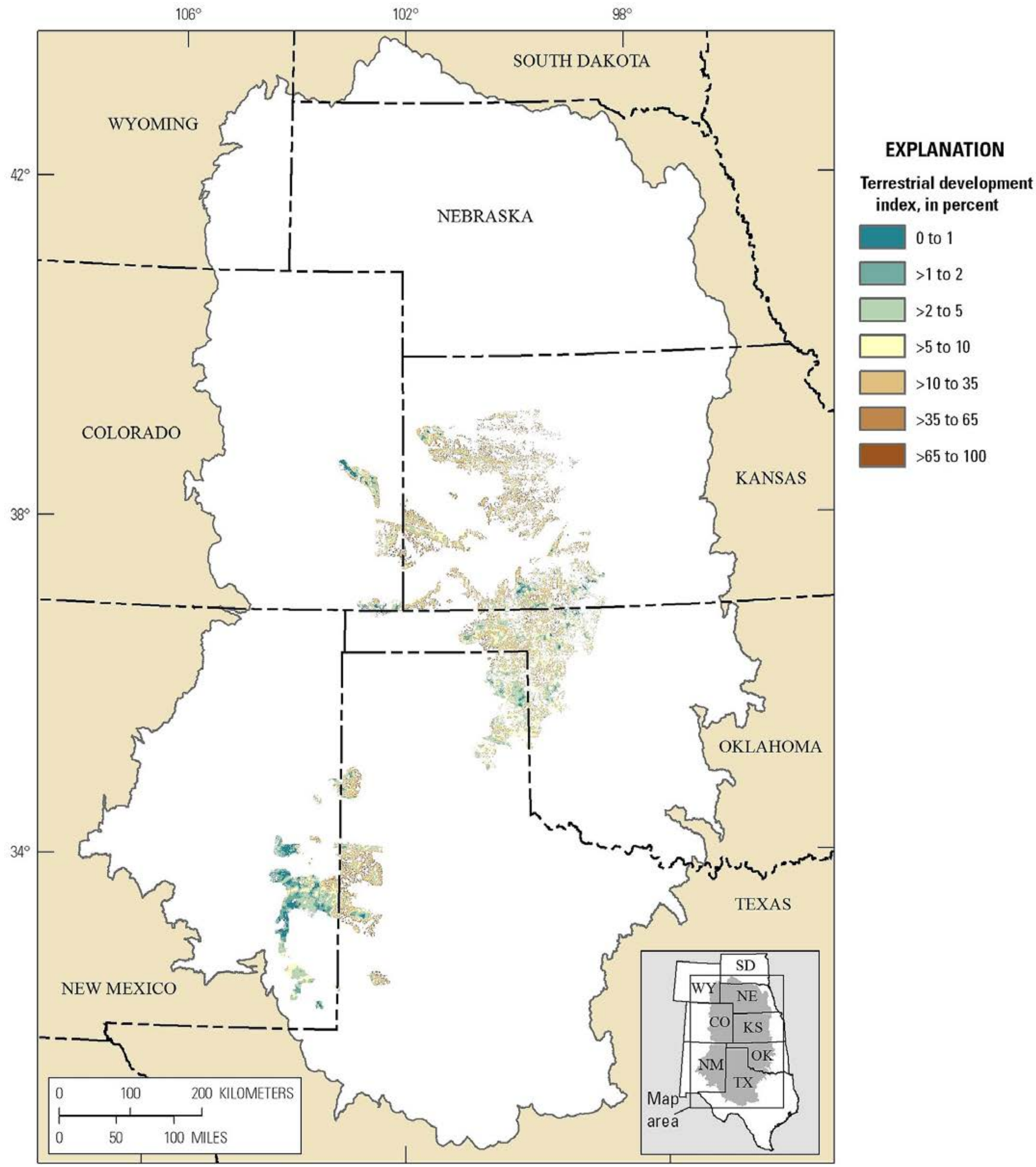


Figure 5–2. Terrestrial development index for lesser prairie-chicken baseline habitat in the Southern Great Plains.

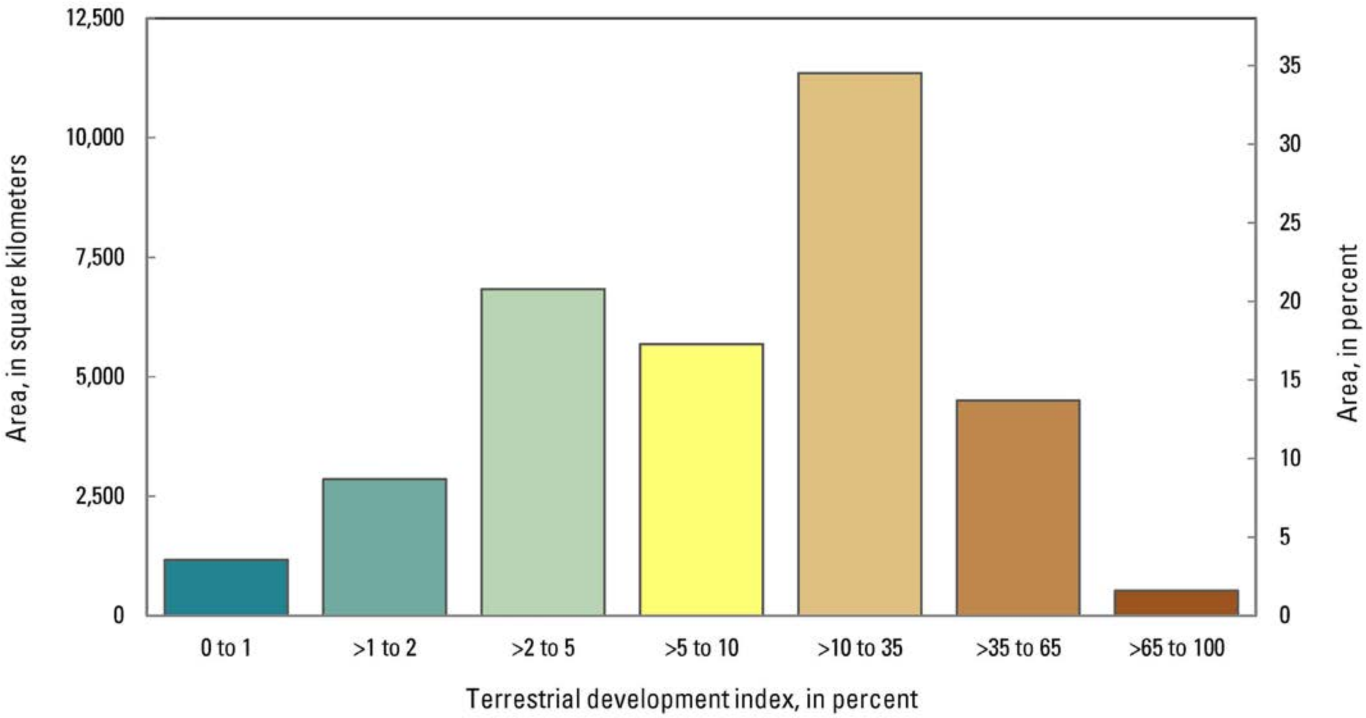


Figure 5–3. Area of lesser prairie-chicken baseline habitat by terrestrial development index class in the Southern Great Plains.

How has development fragmented lesser prairie-chicken habitat (figs. 5–4 and 5–5)?

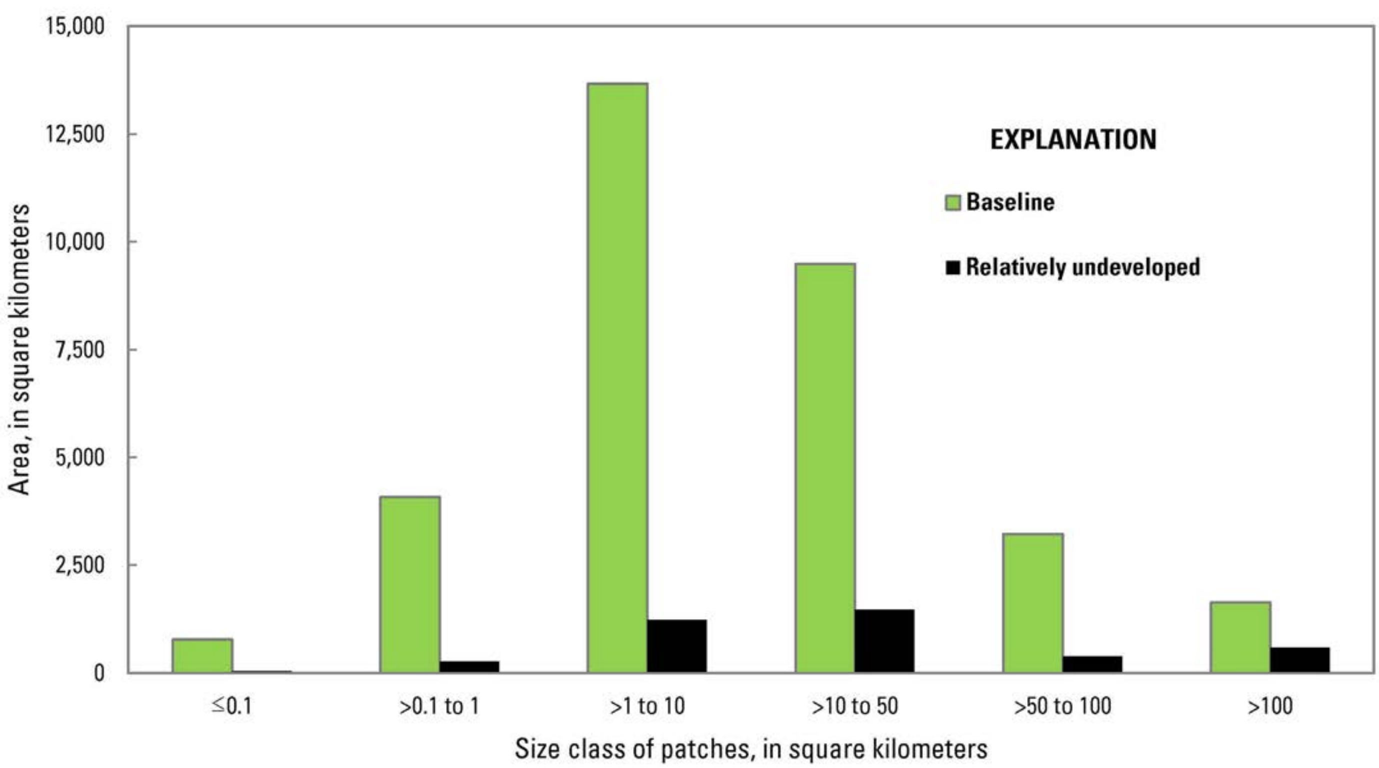


Figure 5–4. Area of lesser prairie-chicken habitat in the Southern Great Plains by patch size class for baseline and relatively undeveloped conditions (terrestrial development index score ≤2 percent).

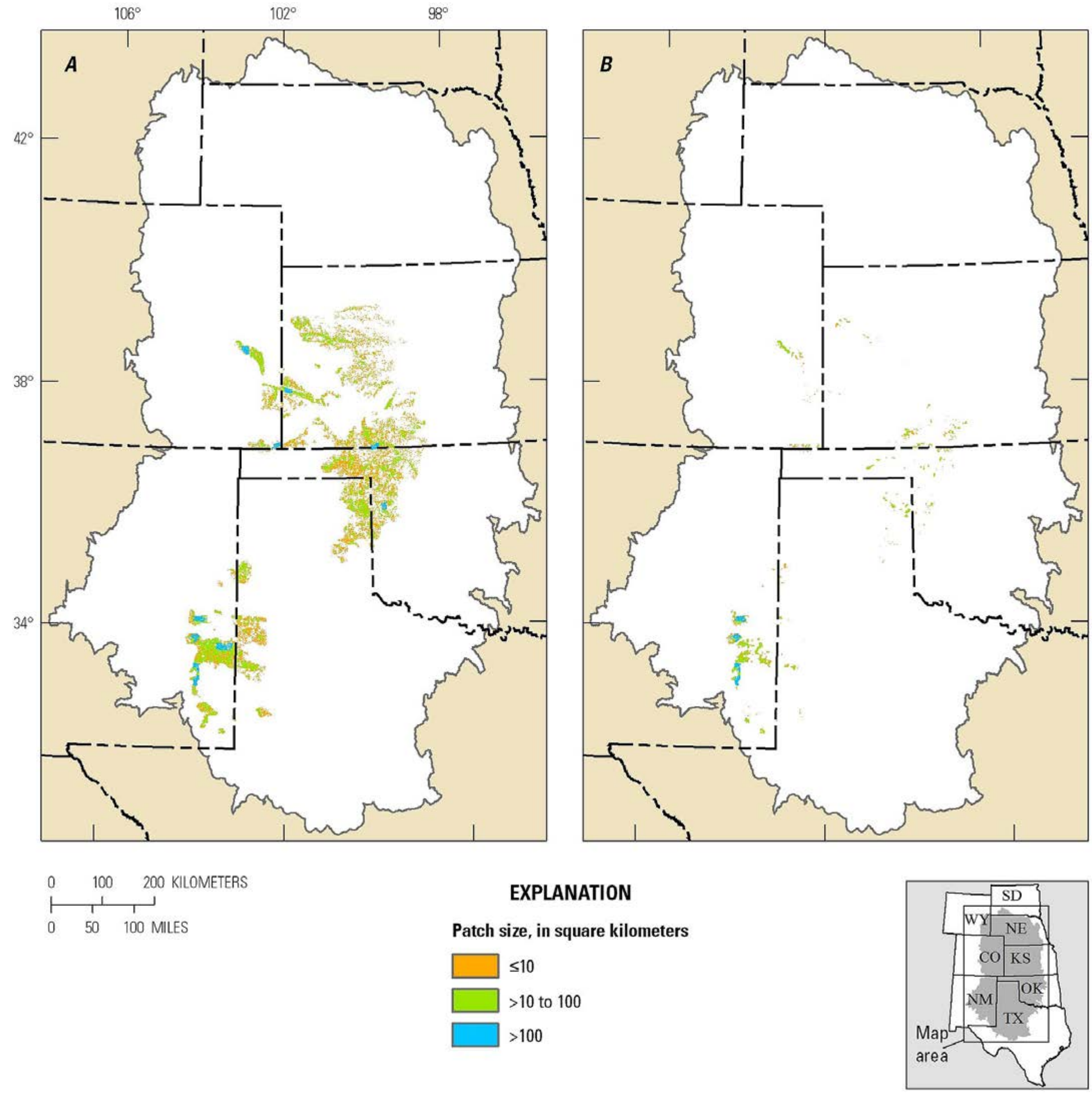


Figure 5–5. Patch size of lesser prairie-chicken habitat in the Southern Great Plains. *A,* Baseline habitat. *B,* Relatively undeveloped habitat (terrestrial development index score ≤2 percent).

Where is lesser prairie-chicken habitat with the highest overall landscape-level rank (fig. 5–6)?

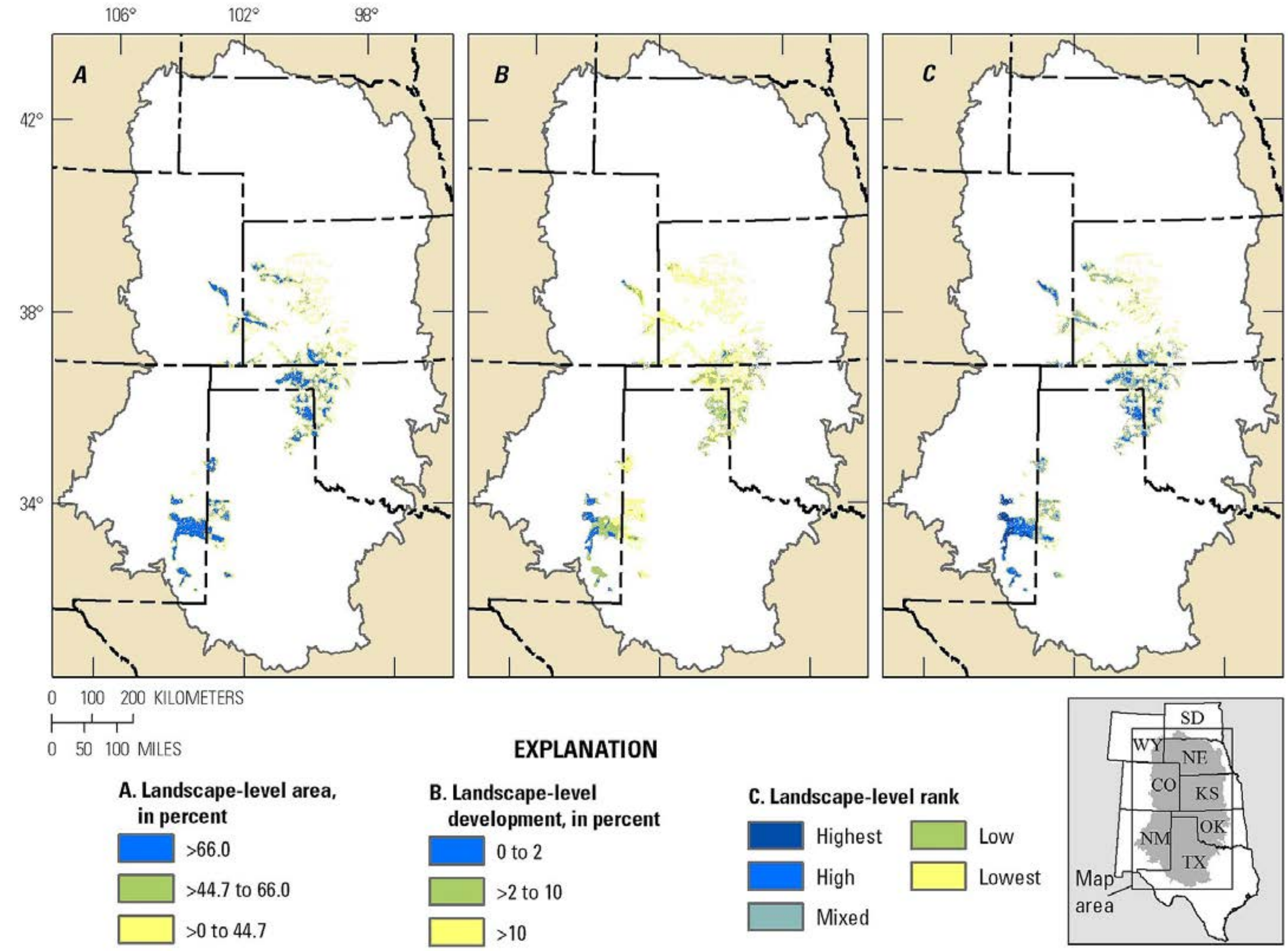


Figure 5–6. Landscape-level summaries for lesser prairie-chicken habitat in the Southern Great Plains. Overall landscape-level rank (*C*) is derived from (*A*) landscape-level habitat area and (*B*) landscape-level development, summarized by a 5-kilometer-radius (3.11-mile) moving window (see table 5–3). Highest overall landscape-level rank corresponds to the largest landscape-level area and the lowest landscape-level development. Lowest overall landscape-level rank corresponds to the smallest landscape-level area and highest landscape-level development. Landscape-level ranks are not intended as standalone summaries and are best interpreted in conjunction with the geospatial datasets used to address core management questions.

Summary

- Baseline habitat for lesser prairie-chicken is largely restricted to southeastern Colorado, western Kansas and Oklahoma, eastern New Mexico, and the panhandle of Texas (fig. 5–1). Baseline habitat totaled approximately 33,000 km^2 (12,741 mi^2).
- Lesser prairie-chicken habitat with the lowest development levels is concentrated in New Mexico. Only 12 percent of their habitat is relatively undeveloped (TDI score ≤2 percent), and 21 percent of habitat has low development (TDI scores 2–5 percent). However, 15 percent of the habitat has very high development levels (TDI score >35 percent) (figs. 5–2 and 5–3).
- The remaining lesser prairie-chicken habitat is highly fragmented (figs. 5–4 and 5–5*A*). More than 85 percent of baseline habitat occurs in isolated patches smaller than 50 km^2 (19.3 mi^2), and only 5 percent is in patches greater than 100 km^2 (38.6 mi^2) (fig. 5–4). The largest patches of relatively undeveloped habitat are in southeastern New Mexico (fig. 5–5*B*).
- The largest, most intact areas (the highest overall landscape-level rank) occur predominantly in southeastern New Mexico, with smaller intact areas scattered throughout the rest of the lesser prairie-chicken distribution (fig. 5–6*C*).
- The broad-scale summaries provided by the REA are intended to be used in conjunction with local-level information on habitat conditions (Wood and others, 2017).

References Cited

Applegate, R.D., and Riley, T.Z., 1998, Lesser prairie-chicken management: Rangelands, v. 20, p. 13−15.

Bartuszevige, A.M., and Daniels, A., 2016, Impacts of energy development, anthropogenic structures, and land use change on lesser prairie-chickens, chap. 11 *of* Haukos, D.A., and Boal, C.W., eds., Ecology and conservation of lesser prairie-chickens: Boca Raton, Fla., CRC Press, Cooper Ornithological Society, Studies in Avian Biology, v. 48, p. 205–220.

Boal, C.W., and Haukos, D.A., 2016, The lesser prairie-chicken—A brief introduction to the grouse of the Southern Great Plains, chap. 1 *of* Haukos, D.A., and Boal, C.W., eds, Ecology and conservation of lesser prairie-chickens: Boca Raton, Fla., CRC Press, Cooper Ornithological Society, Studies in Avian Biology, v. 48, p. 1–11.

Boyd, C.S., and Bidwell, T.G., 2001, Influence of prescribed fire on lesser prairie-chicken habitat in shinnery oak communities in western Oklahoma: Wildlife Society Bulletin, v. 29, no. 3, p. 938–947.

Crawford, J.A., and Bolen, E.G., 1976, Effects of land use on lesser prairie chickens in Texas: Journal of Wildlife Management, v. 40, p. 96−104.

Fields, T.L., White, G.C., Gilgert, W.C., and Rodgers, R.D., 2006, Nest and brood survival of lesser prairie-chickens in west central Kansas: Journal of Wildlife Management, v. 70, no. 4, p. 931–938.

Fritts, S.R., Grisham, B.A., Haukos, D.A., Boarl, C.W., Patten, M.A., Wolfe, D.H., Dixon, C.E., Cox, R.D., and Heck, W.R., 2016, Long-term lesser prairie-chicken nest ecology in response to grassland management: Journal of Wildlife Management, v. 80, p. 527–539.

Fuhlendorf, S.D., Engle, D.M., Elmore, R.D., Limb, R.F., and Bidwell, T.G., 2012, Conservation of pattern and process—Developing an alternative paradigm of rangeland management: Rangeland Ecology and Management, v. 65, p. 579–589.

Fuhlendorf, S.D., Hovick, T.J., Elmore, R.D., Tanner, A.M., Engle, D.M., and Davis, C.A., 2017, A hierarchical perspective to woody plant encroachment for conservation of prairie-chickens: Rangeland Ecology and Management, v. 70, p. 9–14.

Fuhlendorf, S.D., Woodward, A.J.W., Leslie, D.M., Jr., and Shackford, J.S., 2002, Multi-scale effects of habitat loss and fragmentation on lesser prairie-chicken populations of the US Southern Great Plains: Landscape Ecology, v. 17, p. 617–628.

Garton, E.O., Hagen, C.A., Beauprez, G.M., Kyle, S.C., Pitman, J.C., Schoeling, D.D., and Van Pelt, W.E., 2016, Population dynamics of the lesser prairie-chicken, chap. 4 *of* Haukos, D.A., and Boal, C.W., eds, Ecology and conservation of lesser prairie-chickens: Boca Raton, Fla., CRC Press, Cooper Ornithological Society, Studies in Avian Biology, v. 48, p. 49–76.

Giesen, K.M., 2000, Population status and management of lesser prairie-chicken in Colorado: The Prairie Naturalist, v. 32, no. 3, p. 137–148.

Grisham, B.A., and Boal, C.W., 2015, Causes of mortality and temporal patterns in breeding season survival of lesser prairie-chickens in shinnery oak prairies: Wildlife Society Bulletin, v. 39, p. 536–542.

Hagen, C.A., and Giesen, K.M., 2005, Lesser prairie chicken (*Tympanuchus pallidicinctus*), *in* Poole, A., ed., The birds of North America: Ithaca, N.Y., Cornell Lab of Ornithology, no. 364, 20 p., accessed January 2014 at https://birdsna.org/Species-Account/bna/species/lepchi/introduction.

Hagen, C.A., Jamison, B.E., Giesen, K.M, and Riley, T.Z., 2004, Guidelines for managing lesser prairie-chicken populations and their habitats: Wildlife Society Bulletin, v. 32, no. 1, p. 69–82.

Hagen, C.A., Pavlacky, D., Jr., Adachi, K., Hornsby, F.E., Rintz, T.J., and McDonald, L.L., 2016, Multiscale occupancy modeling provides insights into range-wide conservation needs of lesser prairie-chicken (*Tympanuchus pallidicinctus*): Condor, v. 118, p. 597–612.

Hagen, C.A., Pitman, J.C., Loughin, T.M., Sandercock, B.K., Robel, R.J., and Applegate, R.D., 2011, Impacts of anthropogenic features on habitat use by lesser prairie-chickens, *in* Sandercock, B.K., Martin, K., and Segelbacher, G., eds., Ecology, conservation, and management of grouse: Berkeley, Calif., University of California Press, Cooper Ornithological Society, Studies in Avian Biology, no. 39, p. 63–75.

Haukos, D.A., and Zavaleta, J.C., 2016, Habitat, chap. 6 *of* Haukos, D.A., and Boal, C.W., eds, Ecology and conservation of lesser prairie-chickens: Boca Raton, Fla., CRC Press, Cooper Ornithological Society, Studies in Avian Biology, v. 48, p. 99–132.

Jamison, B.E., Robel, R.J., Pontius, J.S., Applegate, R.D., and Pontius, J.S., 2002, Invertebrate biomass—Associations with lesser prairie-chicken habitat use and sand sagebrush density in southwestern Kansas: Wildlife Society Bulletin, v. 30, no. 2, p. 517–526.

Jarnevich, C.S., Holcombe, T.S., Grisham, B.A., Timmer, J., Boal, C.W., Butler, M.J., Pitman, J., Kyle, S.C., Klute, D., Beauprez, G.M., Janus, A., and Van Pelt, W.E., 2016, Assessing range-wide habitat suitability for the lesser prairie-chicken: Avian Conservation and Ecology, v. 11, no. 2, accessed August 23, 2017, at https://doi.org/10.5751/ACE-00807-110102.

Jarnevich, C.S., and Laubhan, M.K., 2011, Balancing energy development and conservation—A method utilizing species distribution models: Environmental Management, v. 47, p. 926–936.

McDonald, L., Beauprez, G., Gardner, G., Griswold, J., Hagen, C., Hornsby, F., Klute, D., Kyle, S., Pitman, J., Rintz, T., Schoeling, D., and Van Pelt, B., 2014, Range-wide population size of the lesser prairie-chicken—2012 and 2013: Wildlife Society Bulletin, v. 38, no. 3, p. 536−546.

Melcher, C.P., 2015, Lesser prairie chicken, chap. 15 *of* Assal, T.J., Melcher, C.P., and Carr, N.B., eds., 2015, Southern Great Plains Rapid Ecoregional Assessment—Pre-assessment report: U.S. Geological Survey Open-File Report 2015–1003, p. 118–126, accessed July 13, 2017, at https://doi.org/10.3133/ofr20151003.

Patten, M.A., Wolfe, D.H., Shochat, E., and Sherrod, S.K., 2005, Effects of microhabitat and microclimate selection on adult survivorship of the lesser prairie-chicken: Journal of Wildlife Management, v. 69, no. 3, p. 1270–1278.

Pitman, J.C., Hagen, C.A., Robel, R.J., Loughin, T.M., and Applegate, R.D., 2005, Location and success of lesser prairie-chicken nests in relation to vegetation and human disturbance: Journal of Wildlife Management, v. 69, no. 3, p. 1259–1269.

Pruett, C.L., Patten, M.A., and Wolfe, D.H., 2009, Avoidance behavior by prairie grouse—Implications for development of wind energy: Conservation Biology, v. 23, no. 5, p. 1253–1259.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100.

Robinson, S.G., Haukos, D.A., Sullins, D.S., and Plumb, R.T., 2016, Use of free water by nesting lesser prairie-chickens: Southwestern Naturalist, v. 61, p. 187–193.

Rodgers, R.D., 2016, A history of lesser prairie-chickens, chap. 2 *of* Haukos, D.A., and Boal, C.W., eds, Ecology and conservation of lesser prairie-chickens: Boca Raton, Fla., CRC Press, Cooper Ornithological Society, Studies in Avian Biology, v. 48, p. 15–38.

Ross, B.E., Haukos, D.A., Hagen, C.A., and Pitman, J.C., 2016, Landscape composition creates a threshold influencing lesser prairie-chicken population resilience to extreme drought: Global Ecology and Conservation, v. 6, p. 179–188.

Silvy, N.J., Peterson, M.J., and Lopez, R.R., 2004, The cause of the decline of pinnated grouse—The Texas example: Wildlife Society Bulletin, v. 32, no. 1, p. 16–21.

U.S. Fish and Wildlife Service, 2016, Endangered and threatened wildlife and plants—90-day findings on three petitions [including the lesser prairie-chicken] [Notice of petition findings and initiation of status reviews]: Federal Register, v. 81, no. 230, p. 86315–86318.

Van Pelt, W.E., 2016, Legal status of the lesser prairie-chicken, chap. 3 *of* Haukos, D.A., and Boal, C.W., eds, Ecology and conservation of lesser prairie-chickens: Boca Raton, Fla., CRC Press, Cooper Ornithological Society, Studies in Avian Biology, v. 48, p. 39–46.

Van Pelt, W.E., Kyle, S., Pitman, J., Klute, D., Beauprez, G., Schoeling, D., Janus, A., and Haufler, J., 2013, The lesser prairie-chicken range-wide conservation plan: Cheyenne, Wyo., Western Association of Fish and Wildlife Agencies, 367 p., accessed March 2014 at http://www.wafwa.org/documents/2013LPCRWPfinalfor4drule12092013.pdf.

Wood, J.A., Carter, S.K., Litshert, S.E., and Carr, N.B., 2017, Including broad-scale indicators in multiscale natural resource monitoring and assessment programs in the Bureau of Land Management, chap. 3 *of* Carter, S.K., Carr, N.B., Miller, K.H., and Wood, D.J.A., eds., Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, p. 15–37, accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Woodward, A.J.W., Fuhlendorf, S.D., and Leslie, D.M., Jr., 2001, Influence of landscape composition and change on lesser prairie-chicken (*Tympanuchus pallidicinctus*): American Midland Naturalist, v. 145, no. 2, p. 261–274.

Chapter 6. Snowy Plover

Introduction

The snowy plover (*Charadrius nivosus*) is a small shorebird that breeds in scattered locations in North America. Despite its widespread breeding distribution, snowy plovers are relatively uncommon because of their specialized breeding habitat. The breeding population in continental North America was recently estimated at 20,000–30,000 individuals, but population trends are unclear and may vary geographically (Busby, 2002; Morrison and others, 2006; Andres and others, 2012; Thomas and others, 2012). The SGP includes the second largest breeding colony of snowy plovers, with more than 5,000 individuals estimated at the Salt Plains National Wildlife Refuge in Oklahoma in 2007 (Thomas and others, 2012). Other large breeding populations in the SGP occur at Quivira National Wildlife Refuge in Kansas and Cargill Salt Flat in Oklahoma. The snowy plover is listed as a priority species for the Great Plains Landscape Conservation Cooperative, a sensitive species by the Bureau of Land Management in Colorado, a Species of Greatest Conservation Need in New Mexico, and a threatened species in Kansas.

In the SGP, snowy plovers breed along saline lakes, sand bars on large rivers, and large ephemeral wetlands (Mabee and Estelle, 2000; Busby, 2002; Conway and others, 2005). The plovers will also use large, unvegetated areas exposed by the drawdown of large reservoirs in early spring (Mabee and Estelle, 2000). Annual variation in precipitation can affect suitability of nesting habitat. For example, high levels of precipitation may flood ephemeral wetlands or delay the drawdown of reservoirs, which may decrease habitat availability (Busby, 2002). In contrast, low levels of precipitation that lead to dewatering in wetlands and rivers may lead to abandonment of previously used nesting areas.

Agricultural development has led to the loss and degradation of snowy plover habitat. Conversion of wetlands to agriculture can decrease potential breeding habitat for snowy plovers (Davis, 1964; Zuvanich and McHenry, 1964), and potential habitat can be eliminated by sediment infilling from upland agriculture (Burris and Skagen, 2013). Extensive groundwater pumping in the SGP has caused Ogallala aquifer levels to drop, reducing discharge to streams and springs in the region (Cross and others, 1985; Reeves and Temple, 1986; Busby, 2002; McGuire and others, 2003). Decreased water availability can alter temporal hydrodynamics and increase lake salinity (Brune, 2002), which may make them unsuitable for migrating and nesting shorebirds, including snowy plovers (Busby, 2002; Conway and others, 2005; Andrei and others, 2008). In addition, enhancing waterfowl habitat in wetlands generally increases vegetation cover, thereby degrading or eliminating snowy plover breeding habitat (Busby, 2002).

Nesting success of snowy plovers can be negatively affected by human disturbance, predation, and competition with invasive species (Grover and Knopf, 1982; Conway and others, 2005; Saalfeld and others, 2011). For example, predation by increasing populations of ravens (*Corvus* spp.) in the southern High Plains of Texas was believed to be the cause of a 31 percent decline in nesting success over a 10-year period (Saalfeld and others, 2011). The expansion of invasive species such as tamarisk (*Tamarix* spp.) along rivers and wetlands may reduce potential plover breeding habitat (Busby, 2002; Page and others, 2009). Additional background information on snowy plovers can be found in the SGP pre-assessment report (George, 2015).

Rapid Ecoregional Assessment Components Evaluated for the Snowy Plover

The key ecological attributes and change agents addressed by core management questions for snowy plover habitat include the amount and distribution (table 6–1) and development (table 6–2). Invasive woody species were evaluated for riparian and wetland communities (see Reese and others, 2017, chap. 8). Fire occurrence and climate change were evaluated for the entire SGP (see Reese and others, 2017, chap. 3). Overall landscape-level ranking variables are summarized in table 6–3. The core and integrated management questions are listed in table 6–4.

Snowy plover. Photograph by Rinus Baak, U.S. Fish and Wildlife Service (Creative Commons Attribution 2.0 Generic).

Table 6–1. Key ecological attributes and associated indicators used to address core management questions for snowy plovers for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Amount and distribution	Total area	Baseline habitat (breeding)[2]
Landscape dynamics	Fire occurrence	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]MaxEnt was used to predict baseline habitat. See chapter 2, "Methods Overview for Species," for methods and datasets used.

Table 6–2. Anthropogenic change agents and associated indicators used to address core management questions for snowy plovers for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Development	Aquatic development index (ADI)	Percentage of baseline habitat in seven development classes by sixth-level watershed
Invasive species	Potential for woody species expansion	See Reese and others (2017, chap. 8)
Climate change	Projected temperature and precipitation	See Reese and others (2017, chap. 4)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

Table 6–3. Landscape-level variables used to address the integrated management question for snowy plovers. Ranks for landscape-level habitat area and development were combined into an overall landscape-level rank for the Southern Great Plains Rapid Ecoregional Assessment.

[>, greater than]

Landscape-level variables[1]	Description	Relative rank[2]		
		Lowest	Medium	Highest
Area	Area of snowy plover habitat as a percentage of fifth-level watershed area	>0–1.78	>1.78–9.42	>9.42
Development	Mean aquatic development index (ADI) score for snowy plover habitat, summarized by fifth-level watershed	0–20	>20–40	>40

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address integrated management questions.

[2]Ranking breakpoints for area of baseline habitat were determined from equal subsets of the data. Ranking breakpoints for the aquatic development index were standardized for all aquatic conservation elements.

Table 6–4. Management questions addressed for snowy plovers for the Southern Great Plains Rapid Ecoregional Assessment.

Core management questions[1]	Results
What is the distribution of baseline habitat for snowy plovers?	Figure 6−1
Where does existing development pose the greatest threat to snowy plover habitat, and where are the large, relatively undeveloped areas?	Figures 6−2 and 6−3
Integrated management question[2]	**Results**
Where is snowy plover habitat with the highest overall landscape-level rank?	Figure 6−4

[1]See Reese and others (2017, chap. 11) for management questions that could not be addressed.

[2]See table 6−3.

Management Questions and Results

What is the distribution of baseline habitat for snowy plovers (figs. 6–1)?

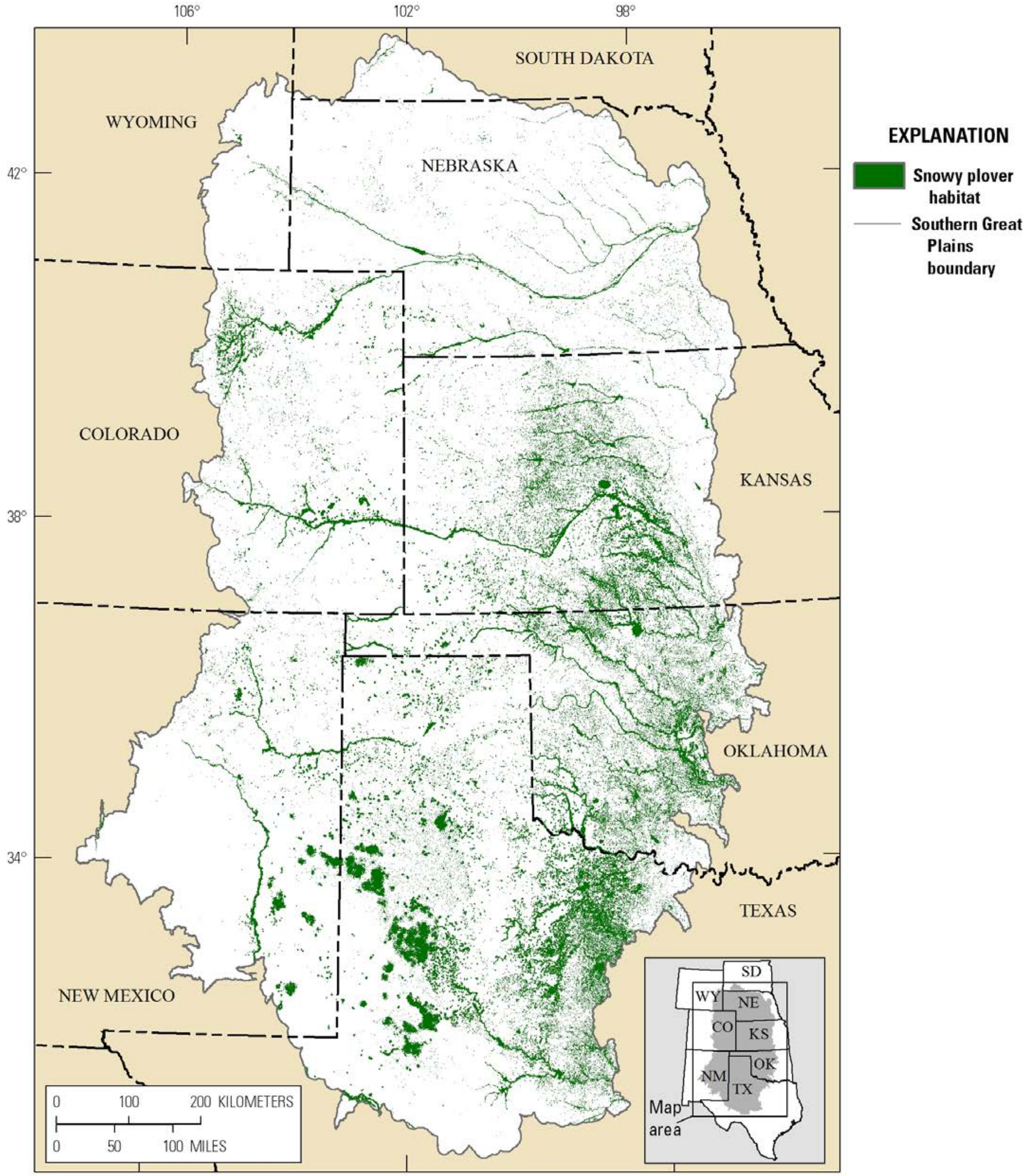


Figure 6–1. Predicted distribution of baseline habitat for snowy plovers in the Southern Great Plains.

Where does existing development pose the greatest threat to snowy plover habitat, and where are the large, relatively undeveloped areas (figs. 6–2 and 6–3)?

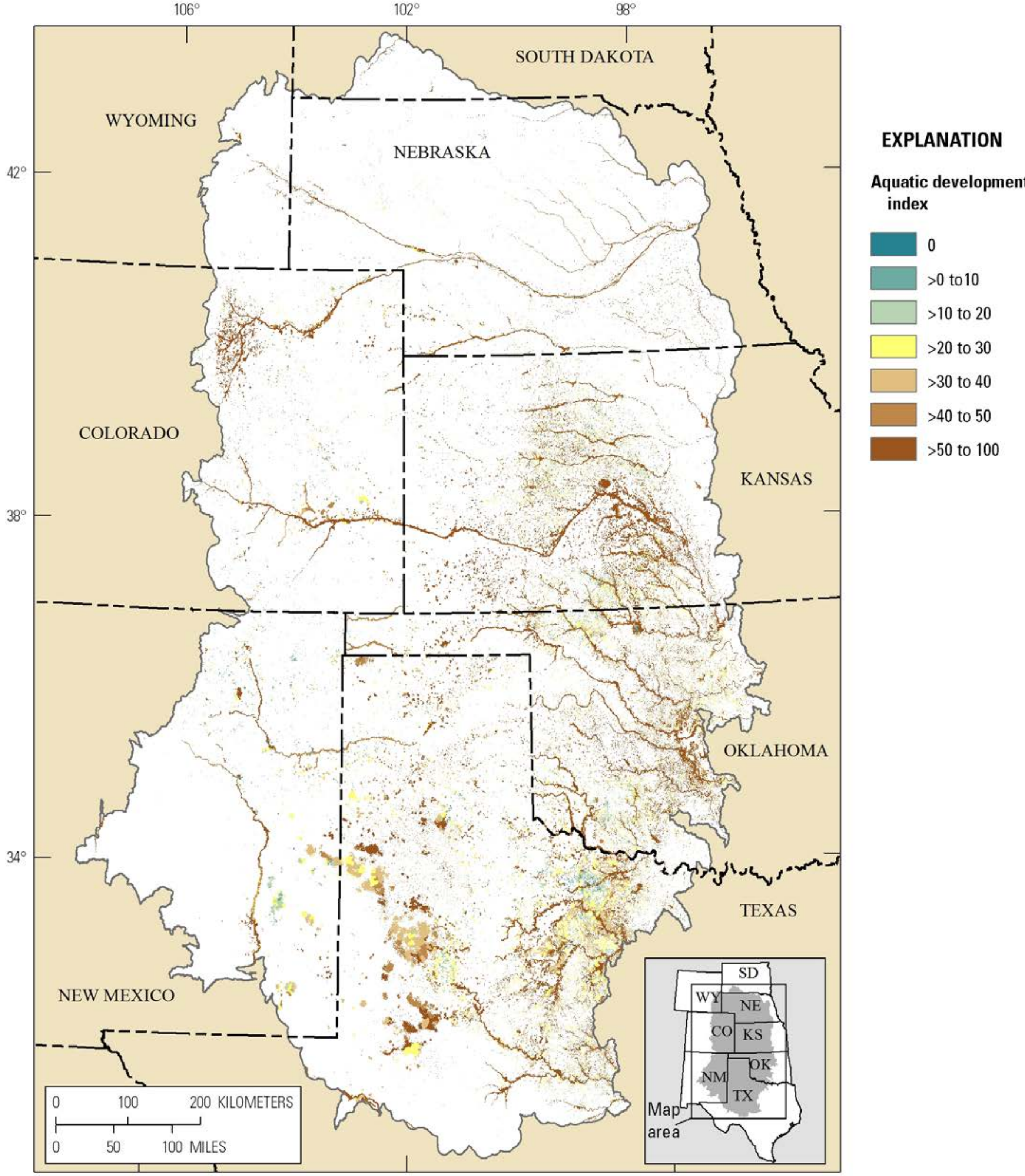


Figure 6–2. Aquatic development index for snowy plover baseline habitat in the Southern Great Plains.

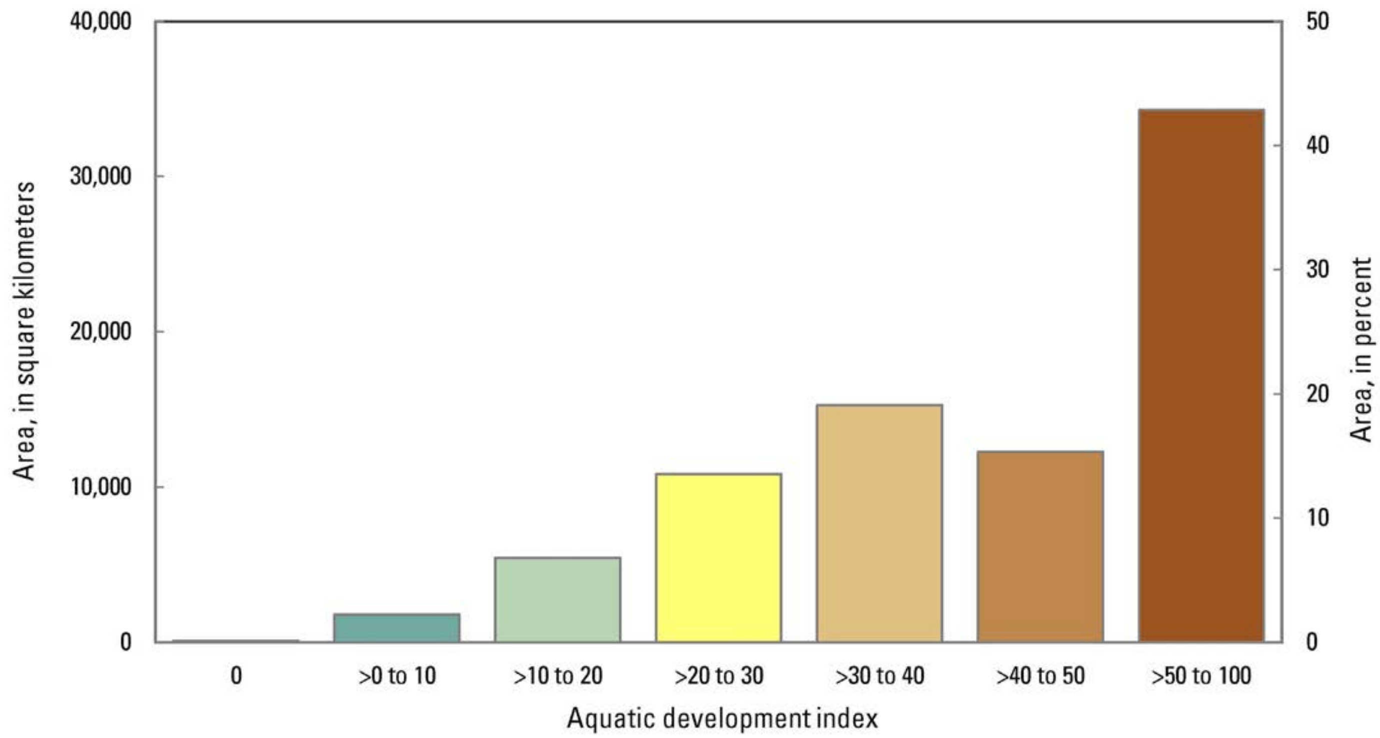


Figure 6–3. Area of snowy plover baseline habitat by aquatic development index class in the Southern Great Plains.

Where is snowy plover habitat with the highest overall landscape-level rank (fig. 6–4)?

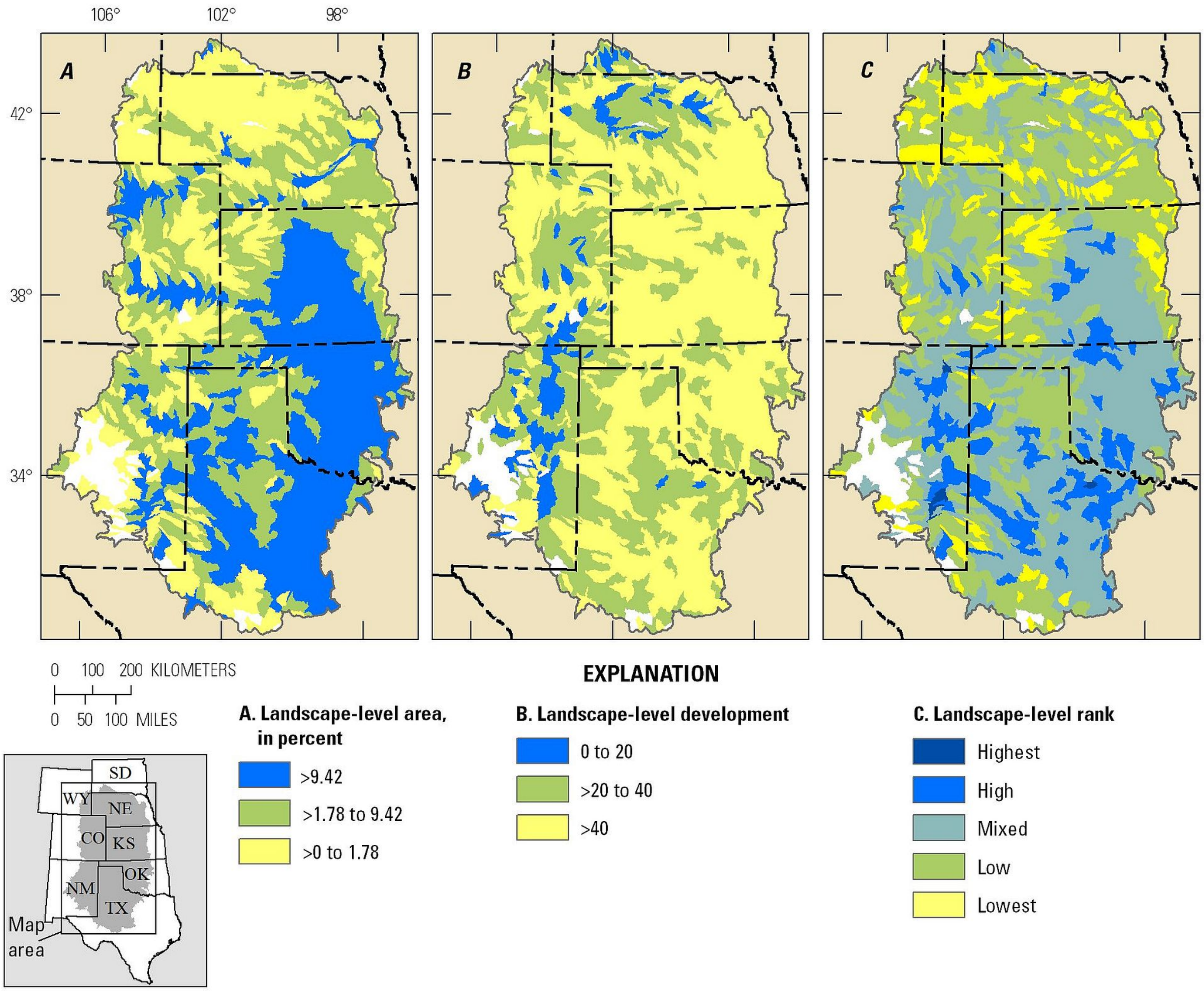


Figure 6–4. Landscape-level summaries for snowy plover habitat in the Southern Great Plains. Overall landscape-level rank (*C*) is derived from (*A*) landscape-level habitat area and (*B*) landscape-level development, summarized by fifth-level watershed (see table 6–3). Highest overall landscape-level rank corresponds to the largest landscape-level area and the lowest landscape-level development. Lowest overall landscape-level rank corresponds to the smallest landscape-level area and highest landscape-level development. Landscape-level ranks are not intended as standalone summaries and are best interpreted in conjunction with the geospatial datasets used to address core management questions.

Summary

- Baseline habitat for snowy plovers is widely dispersed among rivers and open water across the SGP, primarily in Kansas, Oklahoma, and Texas (fig. 6–1). There are approximately 91,000 km^2 (35,135 mi^2) of baseline habitat in the SGP.
- Aquatic development is prevalent throughout most snowy plover habitat (fig. 6–2). Approximately 9 percent of habitat is relatively undeveloped (ADI score ≤20), and 14 percent has low development levels (ADI scores 20–30). However, 58 percent has very high levels of development (ADI score >40) (fig. 6–3).
- The largest, most intact areas (high or highest overall landscape-level rank) are in Texas, Oklahoma, and northeastern New Mexico (fig. 6–4*C*).
- The broad-scale summaries provided by the REA are intended to be used in conjunction with local-level information on habitat conditions (Wood and others, 2017).

References Cited

Andrei, A.E., Smith, L.M., Haukos, D.A., and Surles, J.G., 2008, Habitat use by migrant shorebirds in saline lakes of the Southern Great Plains: Journal of Wildlife Management, v. 72, no. 1, p. 246–253.

Andres, B.A., Smith, P.A., Morrison, R.I.G., Gratto-Trevor, C.L., Brown, S.C., and Friis, C.A., 2012, Population estimates of North American shorebirds, 2012: Wader Study Group Bulletin, v. 119, no. 3, p. 178–194.

Brune, G.M., 2002, Springs of Texas—Volume 1: College Station, Tex., Texas A&M University Press, 566 p.

Burris, L., and Skagen, S.K., 2013, Modeling sediment accumulation in North American playa wetlands in response to climate change, 1940–2100: Climatic Change, v. 117, no. 1–2, p. 69–83.

Busby, W.H., 2002, Kansas recovery plan for the snowy plover (*Charadrius alexandrinus*): Lawrence, Kans., Kansas Biological Survey, 44 p., accessed March 2014 at http://kdwpt.state.ks.us/Services/Threatened-and-Endangered-Wildlife/Recovery-Plan.

Conway, W.C., Smith, L.M., and Ray, J.D., 2005, Shorebird breeding biology in wetlands of the Playa Lakes, Texas, USA: Waterbirds, v. 28, p. 129–138.

Cross, F.B., Moss, R.E., and Collins, J.T., 1985, Assessment of dewatering impacts on stream fisheries in the Arkansas and Cimarron Rivers: Pratt, Kans., Kansas Fish and Game Commission, 161 p.

Davis, J.A., 1964, A survey of migratory shorebirds and water conditions of the Finney Basin near Garden City, Finney County, Kansas: Fort Hayes, Kans., Fort Hays Kansas State College, M.S. thesis, 64 p.

George, T.L., 2015, Snowy plover, chap. 18 *of* Assal, T.J., Melcher, C.P., and Carr, N.B., eds., 2015, Southern Great Plains Rapid Ecoregional Assessment—Pre-assessment report: U.S. Geological Survey Open-File Report 2015–1003, p. 118–126, accessed July 13, 2017, at https://doi.org/10.3133/ofr20151003.

Grover, P.B., and Knopf, F.L., 1982, Habitat requirements and breeding success of Charadriiform birds nesting at Salt Plains National Wildlife Refuge, Oklahoma: Journal of Field Ornithology, v. 53, p. 139–148.

Mabee, T.J., and Estelle, V.B., 2000, Nest fate and vegetation characteristics for snowy plover and killdeer in Colorado, USA: Wader Study Group Bulletin, v. 93, p. 67–72.

McGuire, V.L., Johnson, M.R., Schieffer, R.L., Stanton, J.S., Sebree, S.K., and Verstraeten, I.M., 2003, Water in storage and approaches to ground-water management, High Plains aquifer, 2000: U.S. Geological Survey Circular 1243, 51 p., accessed February 2014 at http://pubs.usgs.gov/circ/2003/circ1243/pdf/C1243.pdf.

Morrison, R.I.G., McCaffery, B.J., Gill, R.E., Skagen, S.K., Jones, S.L., Page, G.W., Gratto-Trevor, C.L., and Andres, B.A., 2006, Population estimates of North American shorebirds, 2006: Wader Study Group Bulletin, v. 111, p. 66–84.

Page, G.W., Warriner, J.S., Warriner, J.C., Paton, P.W., and Stenzel, L.E., 2009, Snowy plover (*Charadrius nivosus*), *in* Poole, A., ed., The birds of North America: Ithaca, N.Y., Cornell Lab of Ornithology, no. 154, accessed February 2014 at http://bna.birds.cornell.edu/bna/species/154.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100.

Reeves, C.C., Jr., and Temple, J.M., 1986, Permian salt dissolution, alkaline lake basins, and nuclear-waste storage, Southern High Plains, Texas and New Mexico: Geology, v. 14, p. 939–942.

Saalfeld, S.T., Conway, W.C., Haukos, D.A., and Johnson, W.P., 2011, Nest success of snowy plovers (*Charadrius nivosus*) in the southern High Plains of Texas: Waterbirds, v. 34, p. 389–399.

Thomas, S.M., Lyons, J.E., Andres, B.A., Elliot-Smith, E., Palacios, E., Cavitt, J.F., Royle, J.A., Fellows, S.D., Maty, K., Howe, W.H., Mellink, E., Melvin, S., and Zimmerman, T., 2012, Population size of snowy plovers breeding in North America: Waterbirds, v. 35, p. 1–14.

Wood, J.A., Carter, S.K., Litshert, S.E., and Carr, N.B., 2017, Including broad-scale indicators in multiscale natural resource monitoring and assessment programs in the Bureau of Land Management, chap. 3 *of* Carter, S.K., Carr, N.B., Miller, K.H., and Wood, D.J.A., eds., Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, p. 15–37, accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Zuvanich, J.R., and McHenry, M.G., 1964, Comparison of water birds observed in Kansas in 1955 and 1963: Transactions of the Kansas Academy of Science, v. 67, p. 169–183.

Chapter 7. Mountain Plover

Introduction

The mountain plover (*Charadrius montanus*) is a migratory shorebird that breeds on disturbed areas with short vegetation, such as shortgrass prairie, mixed-grass prairie, sagebrush steppe, and agricultural fields, in the western Great Plains. The majority of the population winters in California (Wunder and Knopf, 2003), but the use of migratory stopover sites and the distribution of wintering plovers elsewhere is poorly understood (Knopf and Wunder, 2006; Pierce and others, 2017). The population was recently estimated at approximately 20,000 (U.S. Fish and Wildlife Service, 2011). The mountain plover was proposed for listing as a threatened species under the Endangered Species Act, but it ultimately was not listed (see U.S. Fish and Wildlife Service, 2011). It remains, however, a species of conservation concern at the State level throughout its range (Andres and Stone, 2009).

Mountain plovers breed in sparsely vegetated areas and, in the SGP, are strongly associated with native grazers such as the American bison (*Bison bison*) and black-tailed prairie dog (*Cynomys ludovicianus*) that help to maintain such conditions (Knopf and Wunder, 2006; Dinsmore and others, 2010; Augustine and Derner, 2012). Western portions of the SGP may be used as migratory stopover sites for birds breeding further north (Pierce and others, 2017). The plovers are also associated with recently burned grasslands on both wintering and breeding grounds (Knopf and Wunder, 2006). Plover densities were higher on prairie dog colonies and recently burned rangeland compared to unburned rangeland (Augustine, 2011; Augustine and Derner, 2012, 2015; Augustine and Skagen, 2014). Plovers also can be negatively affected by prairie dog colony die-offs following plague epizootics (Augustine and others, 2008).

Habitat loss and degradation—including planting taller nonnative grasses for livestock, conversion of shortgrass prairie to croplands, and loss of wintering habitat because of development—are major threats to mountain plovers (Knopf, 1994; Knopf and Wunder, 2006). Nonnative grasses, such as crested wheatgrass (*Agropyron cristatum*), often increase vegetation cover and height, which reduce habitat quality for mountain plovers (Knopf and Wunder, 2006; Andres and Stone, 2009). Although conversion of shortgrass prairie to croplands can decrease habitat availability, plovers will sometimes nest in agricultural fields, especially fallow fields (Knopf and Rupert, 1999; Shackford and others, 1999). Nests in croplands are at risk from mechanical agricultural practices, but nest predation can be lower on croplands compared to native rangeland; as a result, nest survival can be similar in both native and nonnative habitat types (Dreitz and Knopf, 2007). Urban and agricultural development in California have decreased the availability of wintering habitat for mountain plovers (Wunder and Knopf, 2003).

Fire suppression and the loss or decrease of native grazers, such as bison and prairie dogs, have reduced the availability of native plover breeding habitat. Compared to native grazers, livestock grazing practices can lead to greater homogeneity across shortgrass prairie habitat (Derner and others, 2009). Indeed, plover nesting density tends to be lower on rangeland than on prairie dog colonies or agricultural fields (Dreitz and others, 2005; Tipton and others, 2009). In eastern Colorado, mountain plovers rarely occupy rangeland that lacks prairie dogs or recent fire (Augustine, 2011). Fire suppression, however, has nearly eliminated the influence of fire in shortgrass prairie (Samson and others, 2004).

The effects of energy development on mountain plovers are poorly understood, but recent studies have suggested that oil and gas development may have limited effects on mountain plovers (U.S. Fish and Wildlife Service, 2011). Mountain plovers are relatively tolerant of vehicles associated with agriculture or oil and gas development (Knopf and Wunder, 2006). Although they may be displaced during active development, they may benefit from bare ground created by development (U.S. Fish and Wildlife Service, 2011). Furthermore, collisions with wind turbines and utility lines are not a major concern because breeding birds usually fly low to the ground (Andres and Stone, 2009). Additional background information on mountain plovers can be found in the SGP pre-assessment report (Woolley, 2015).

Rapid Ecoregional Assessment Components Evaluated for the Mountain Plover

The key ecological attributes and change agents addressed by core management questions for mountain plover habitat include amount and distribution, landscape structure, and development (tables 7–1 and 7–2). Invasive woody species and climate change were evaluated for grassland communities (see Reese and others, 2017, chap. 4). Fire occurrence and potentially altered vegetation (including invasive herbaceous plants) were evaluated for the entire SGP (see Reese and others, 2017, chap. 3). Overall landscape-level ranking variables are summarized in table 7–3. The core and integrated management questions are listed in table 7–4.

Mountain plover. Photograph by Ron Knight (Creative Commons Attribution 2.0 Generic).

Table 7–1. Key ecological attributes and associated indicators used to address core management questions for mountain plovers for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Amount and distribution	Total area	Baseline habitat (breeding)[2]
Landscape structure	Patch size	Patch sizes for baseline habitat
Landscape dynamics	Fire occurrence	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]MaxEnt was used to predict baseline habitat. See chapter 2, "Methods Overview for Species," for methods and datasets used.

Table 7–2. Anthropogenic change agents and associated indicators used to address core management questions for mountain plovers for the Southern Great Plains Rapid Ecoregional Assessment.

[km, kilometer; mi, mile]

Attributes	Variables	Indicators[1]
Development	Terrestrial development index (TDI)	Percentage of baseline habitat in seven development classes based on a 2.5-km (1.55-mi) moving window
	Index of fragmentation	Patch sizes for relatively undeveloped[2] habitat
Invasive species	Potential for woody species expansion	See Reese and others (2017, chap. 4)
Climate change	Potential distribution of grasslands	See Reese and others (2017, chap. 4)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]Terrestrial development index score less than or equal to 2 percent.

Table 7–3. Landscape-level variables used to address the integrated management question for mountain plovers. Ranks for landscape-level habitat area and development were combined into an overall landscape-level rank for the Southern Great Plains Rapid Ecoregional Assessment.

[>, greater than; km, kilometer; mi, mile]

Landscape-level variables[1]	Description	Relative rank[2]		
		Lowest	Medium	Highest
Area	Percentage of baseline habitat within a 5-km-radius (3.11-mi) moving window	>0–53.1	>53.1–76.6	>76.6
Development	Mean terrestrial development index (TDI) score for baseline habitat within a 5km-radius (3.11-mi) moving window	0–2	>2–10	>10

[1]See Reese and others (2017, chap. 2) for methods and datasets used to address integrated management questions.

[2]Ranking breakpoints for area of baseline habitat were determined from equal subsets of the data. Ranking breakpoints for terrestrial development index scores were standardized for all terrestrial conservation elements.

Table 7–4. Management questions addressed for mountain plovers for the Southern Great Plains Rapid Ecoregional Assessment.

Core management questions[1]	Results
What is the distribution of baseline habitat for mountain plovers?	Figure 7−1
Where does existing development pose the greatest threat to mountain plover habitat, and where are the large, relatively undeveloped areas?	Figures 7−2 and 7−3
How has development fragmented mountain plover habitat?	Figures 7−4 and 7−5
Integrated management question[2]	**Results**
Where is mountain plover habitat with the highest overall landscape-level rank?	Figure 7−6

[1]See Reese and others (2017, chap. 11) for management questions that could not be addressed.

[2]See table 7−3.

Management Questions and Results

What is the distribution of baseline habitat for mountain plovers (fig. 7–1)?

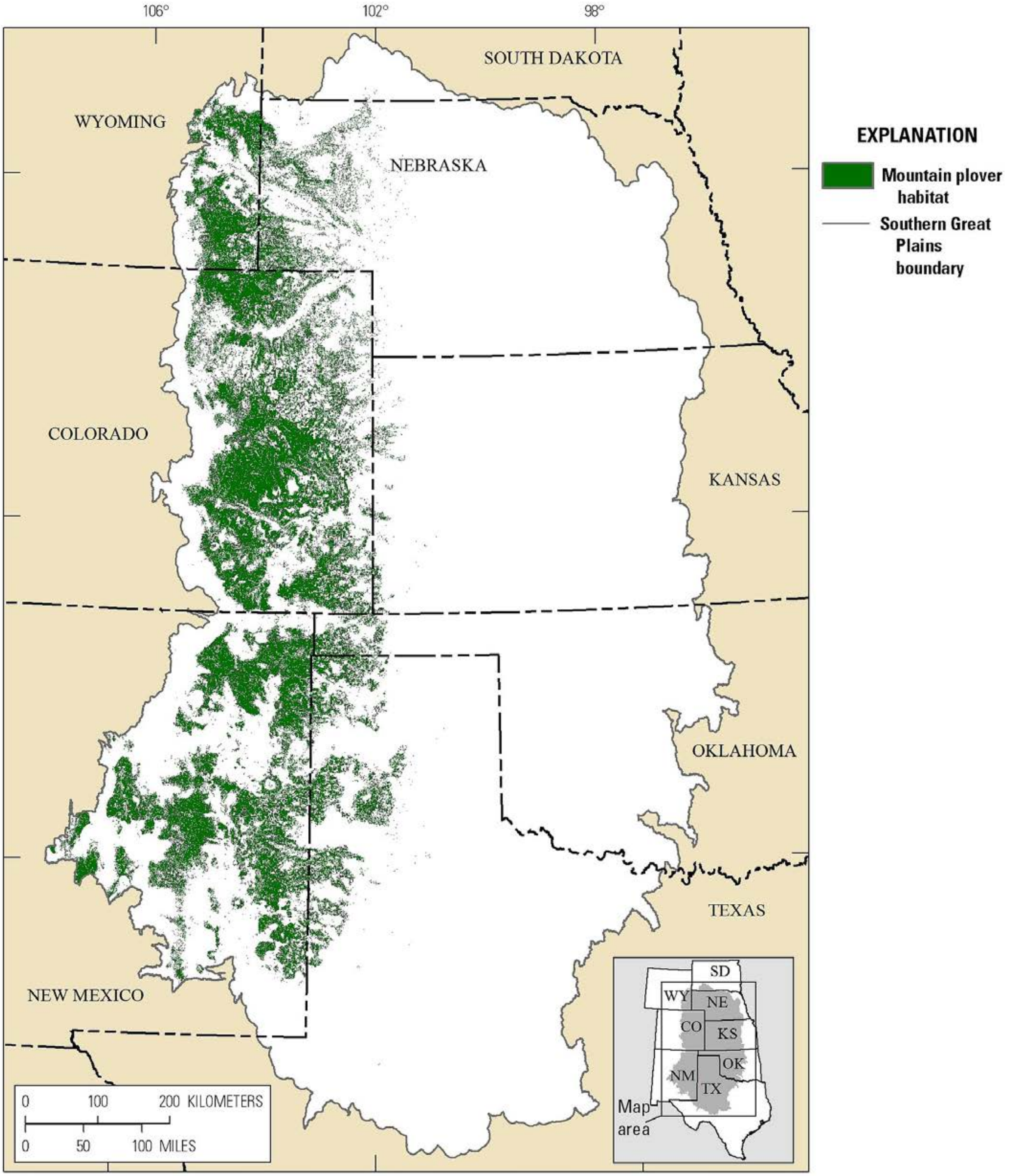


Figure 7–1. Predicted distribution of baseline habitat for mountain plovers in the Southern Great Plains.

Where does existing development pose the greatest threat to mountain plover habitat, and where are the large, relatively undeveloped areas (figs. 7–2 and 7–3)?

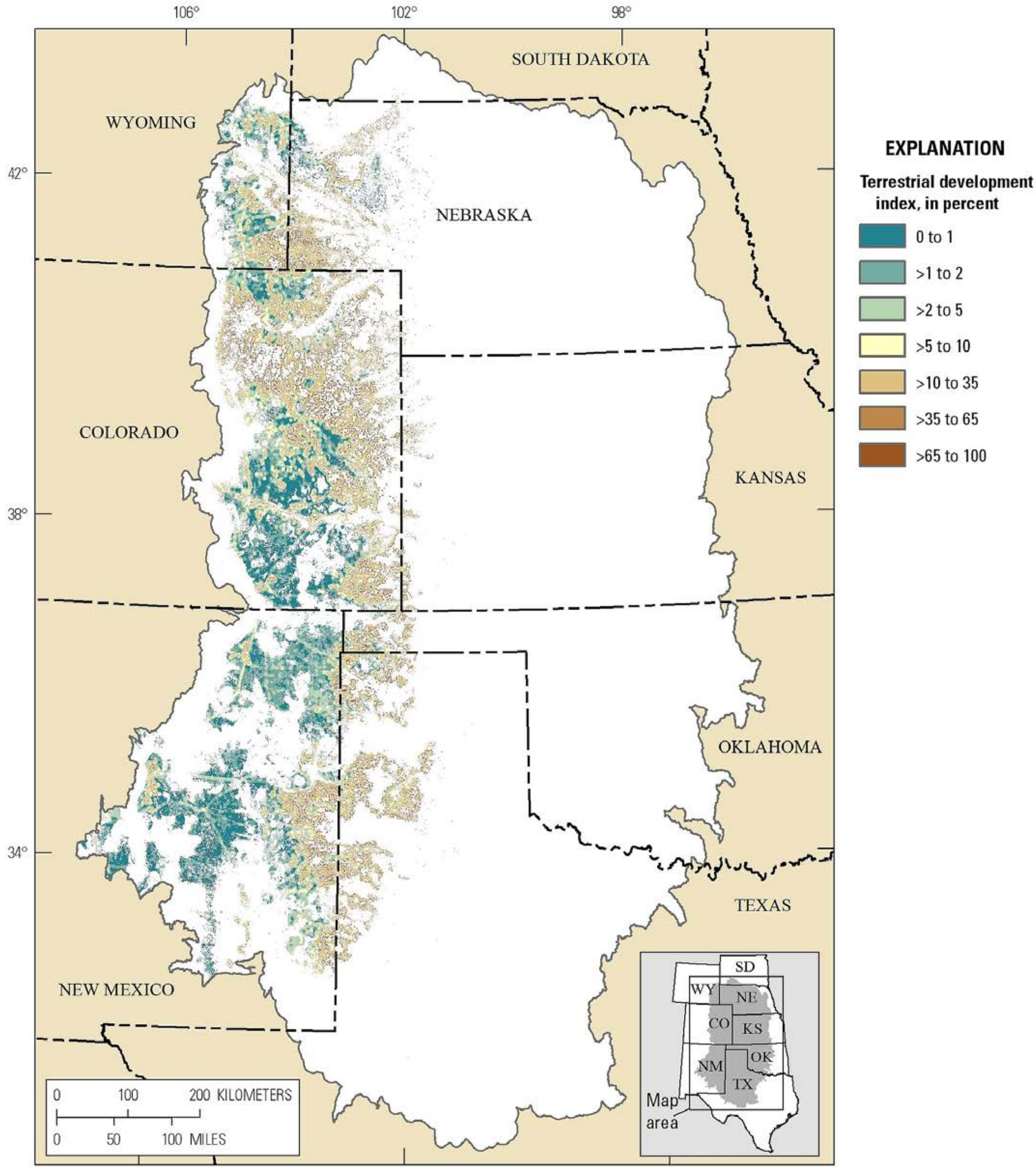


Figure 7–2. Terrestrial development index for mountain plover baseline habitat in the Southern Great Plains.

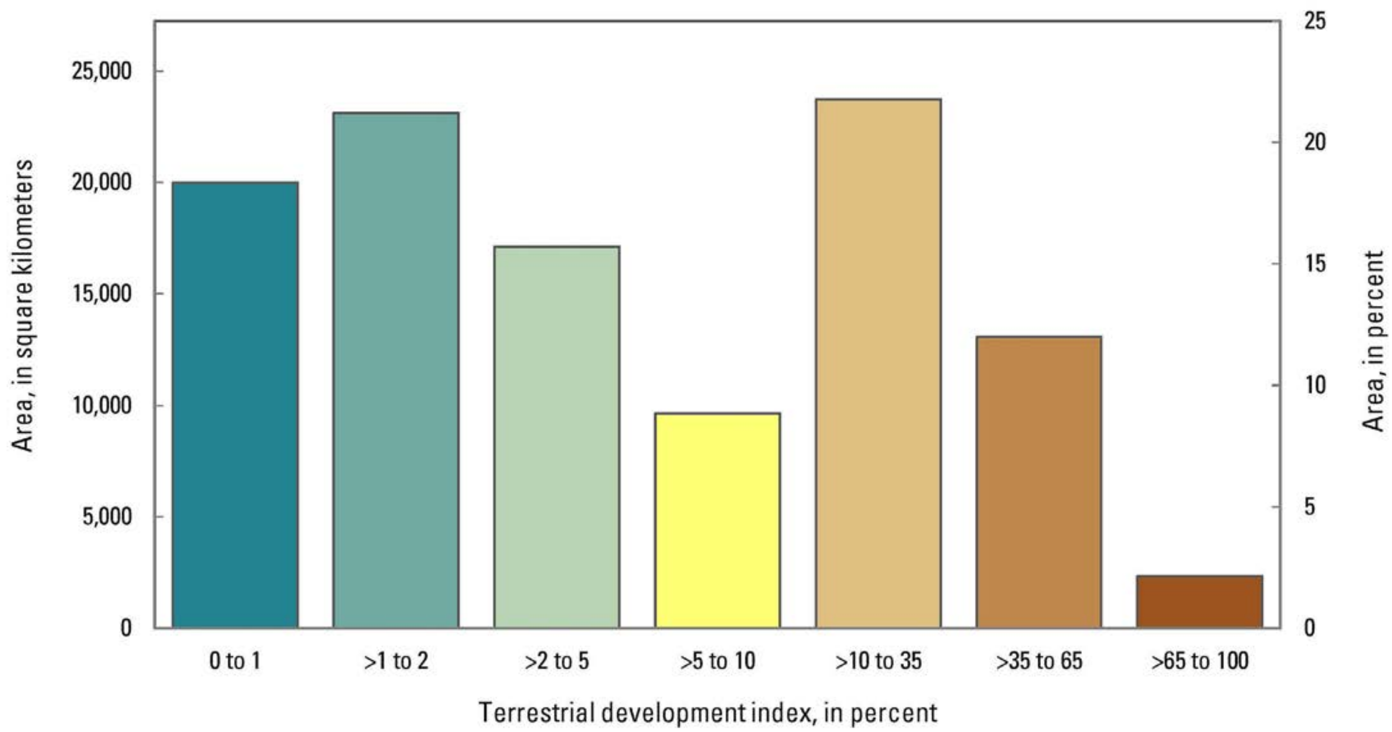


Figure 7–3. Area of mountain plover baseline habitat by terrestrial development index class in the Southern Great Plains.

How has development fragmented mountain plover habitat (figs. 7–4 and 7–5)?

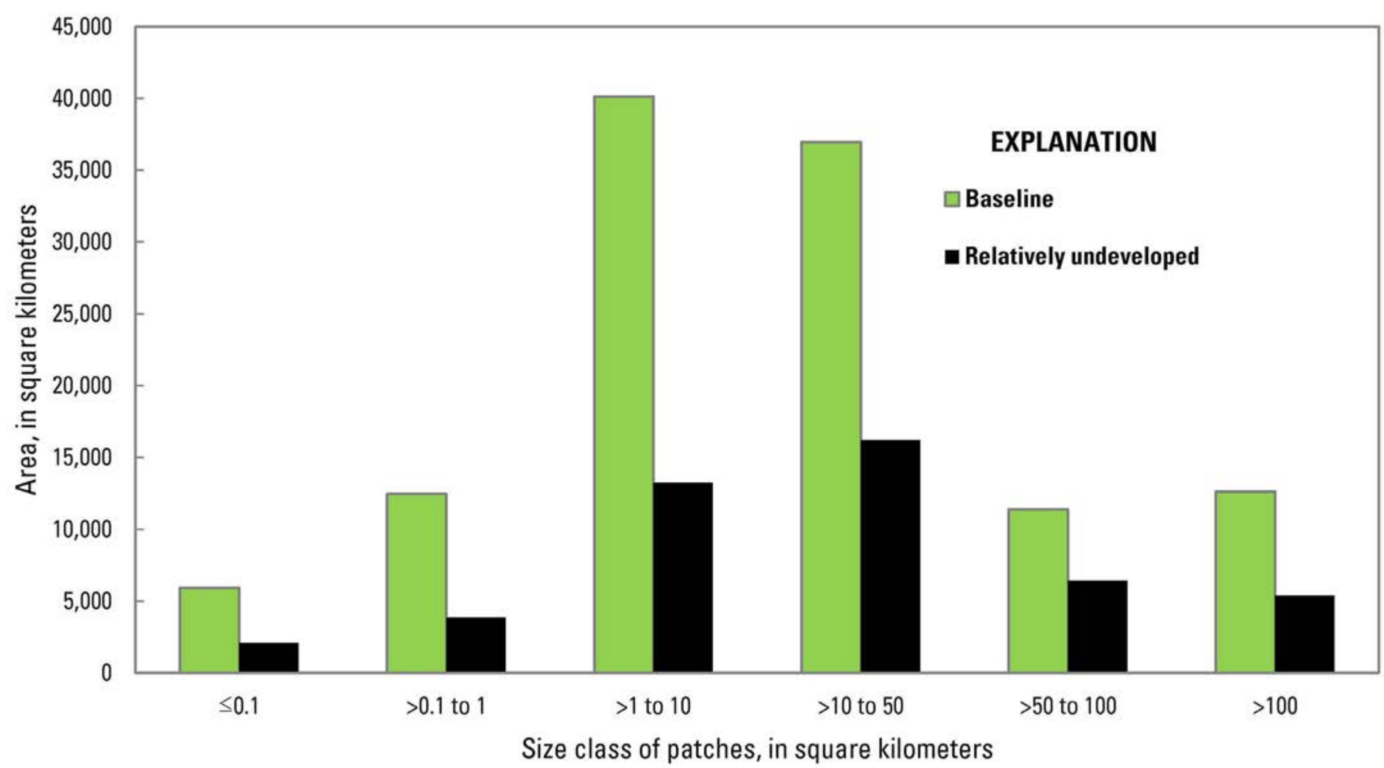


Figure 7–4. Area of mountain plover habitat in the Southern Great Plains by patch size class for baseline and relatively undeveloped conditions (terrestrial development index score ≤2 percent).

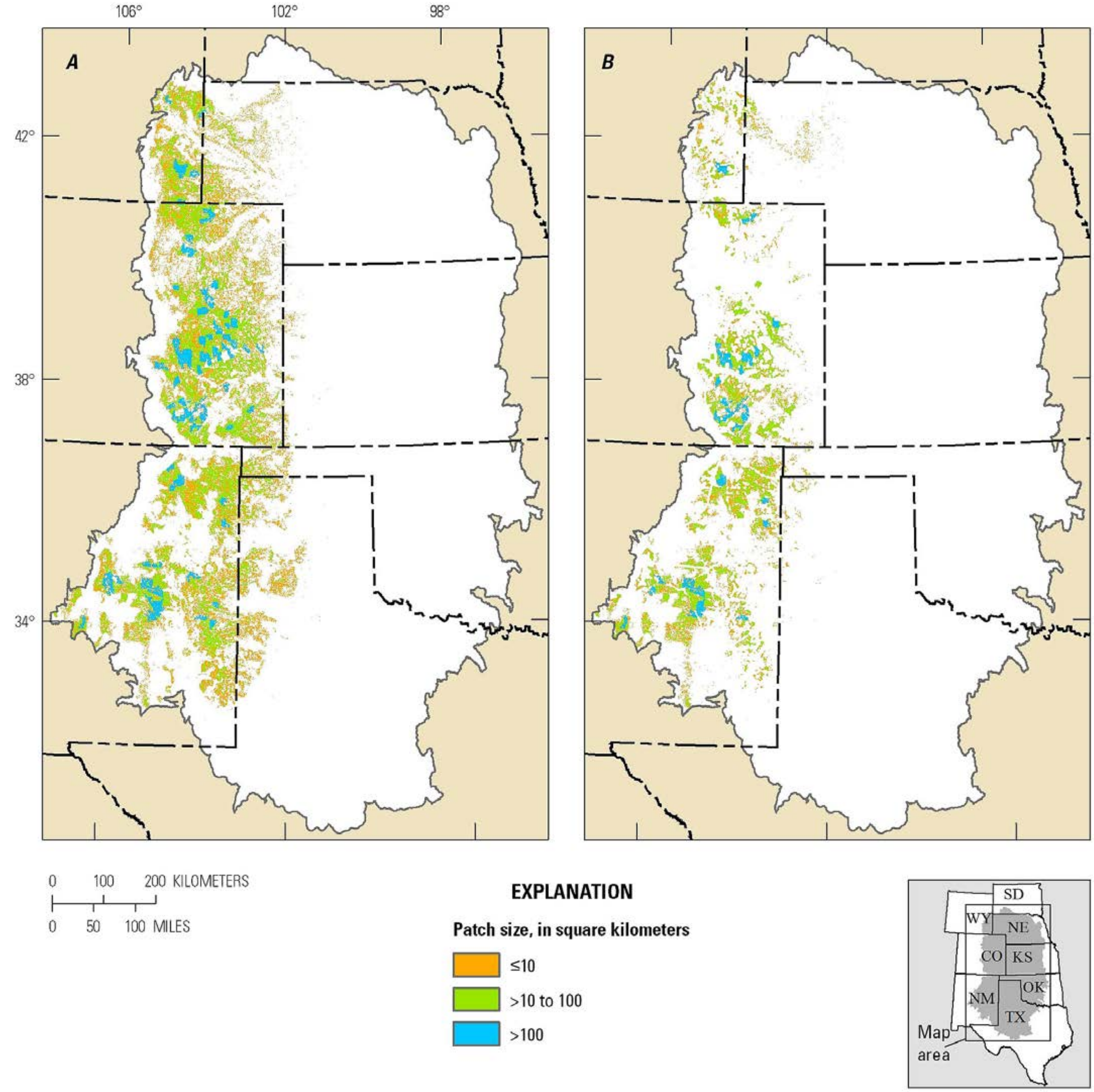


Figure 7–5. Patch size of mountain plover habitat in the Southern Great Plains. *A,* Baseline habitat. *B,* Relatively undeveloped habitat (terrestrial development index score ≤2 percent).

Where is mountain plover habitat with the highest overall landscape-level rank (fig. 7–6)?

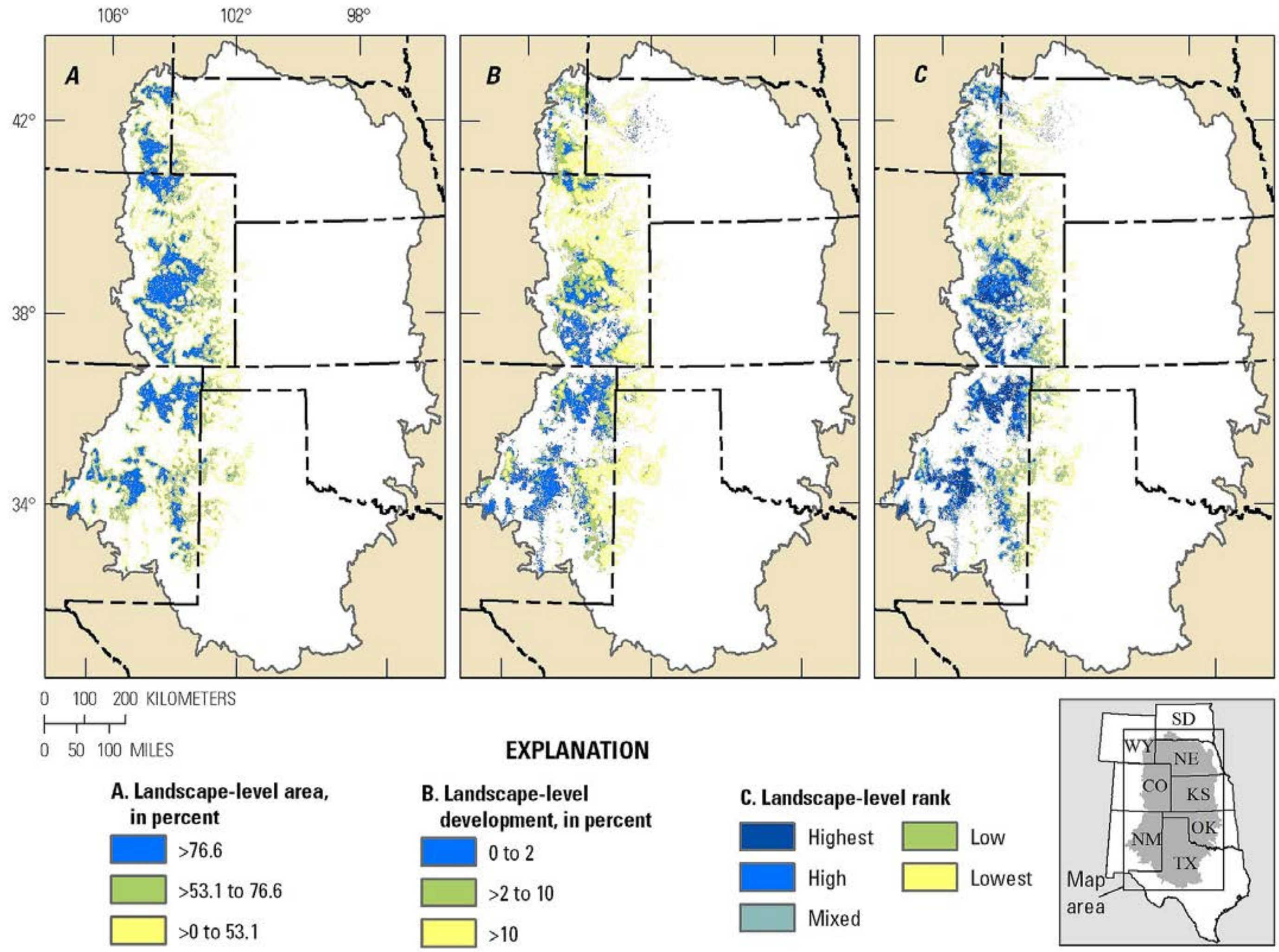


Figure 7–6. Landscape-level summaries for mountain plover habitat in the Southern Great Plains. Overall landscape-level rank (*C*) is derived from (*A*) landscape-level area and (*B*) landscape-level development, summarized by a 5-kilometer-radius (3.11-mile) moving window (see table 7–3). Highest overall landscape-level rank corresponds to the largest landscape-level area and the lowest landscape-level development. Lowest overall landscape-level rank corresponds to the smallest landscape-level area and highest landscape-level development. Landscape-level ranks are not intended as standalone summaries and are best interpreted in conjunction with the geospatial datasets used to address core management questions.

Summary

- Baseline habitat for mountain plovers is within western portions of the SGP (fig. 7–1). There are approximately 119,000 km^2 (45,946 mi^2) of baseline habitat in the SGP.
- Mountain plover habitat with the lowest development levels is concentrated in southeastern Colorado and eastern New Mexico (fig. 7–2). Approximately 40 percent of its habitat is relatively undeveloped (TDI score ≤2 percent), and 16 percent has low development levels (TDI scores 2–5 percent). However, 14 percent has very high levels of development (TDI score >35 percent) (fig. 7–3).
- Mountain plover habitat is highly fragmented (figs. 7–4 and 7–5*A*). Eighty percent of baseline habitat occurs in patches smaller than 50 km^2 (19.3 mi^2) (fig. 7–4). The largest remaining patches of relatively undeveloped habitat are primarily in southeastern Colorado and southeastern New Mexico (fig. 7–5*B*).
- The largest, most intact areas (the highest overall landscape-level rank) are in the western extent of the shortgrass prairie (fig. 7–6*C*), where there has been less conversion to croplands than in other areas of the region (Reese and others, 2017).
- The broad-scale summaries provided by the REA are intended to be used in conjunction with local-level information on habitat conditions (Wood and others, 2017).

References Cited

Andres, B.A., and Stone, K.L., 2009, Conservation plan for the mountain plover (*Charadrius montanus*) (version 1.0): Manomet, Mass., Manomet Center for Conservation Sciences, 48 p., accessed June 29, 2014, at https://www.fws.gov/cno/conservation/mountainplover_consplan_09-05-28a.pdf.

Augustine, D.J., 2011, Habitat selection by mountain plovers in shortgrass steppe: Journal of Wildlife Management, v. 75, no. 2, p. 297–304.

Augustine, D.J., and Derner, J.D., 2012, Disturbance regimes and mountain plover habitat in shortgrass steppe—Large herbivore grazing does not substitute for prairie dog grazing or fire: Journal of Wildlife Management, v. 76, no. 4, p. 721–728.

Augustine, D.J., and Derner, J.D., 2015, Patch-burn grazing management, vegetation heterogeneity, and avian response in a semi-arid grassland: Journal of Wildlife Management, v. 79, p. 927–936.

Augustine, D.J., Dinsmore, S.J., Wunder, M.B., Dreitz, V.J., and Knopf, F.L., 2008, Response of mountain plovers to plague-driven dynamics of black-tailed prairie dog colonies: Landscape Ecology, v. 23, no. 6, p. 689–697.

Augustine, D.J., and Skagen, S.K., 2014, Mountain plover nest survival in relation to prairie dog and fire dynamics in shortgrass steppe: Journal of Wildlife Management, v. 78, p. 595–602.

Derner, J.D., Lauenroth, W.K., Stapp, P., and Augustine, D.J., 2009, Livestock as ecosystem engineers for grassland bird habitat in the western Great Plains of North America: Rangeland Ecology and Management, v. 62, no. 2, p. 111–118.

Dinsmore, S.J., Wunder, M.B., Dreitz, V.J., and Knopf, F.L., 2010, An assessment of factors affecting population growth of the mountain plover: Avian Conservation and Ecology, v. 5, no. 1, article 5, accessed January 18, 2018, at http://www.ace-eco.org/vol5/iss1/art5/.

Dreitz, V.J., and Knopf, F.L., 2007, Mountain plovers and the politics of research on private lands: BioScience, v. 57, no. 8, p. 681–687.

Dreitz, V.J., Wunder, M.B., and Knopf, F.L., 2005, Movements and home ranges of mountain plovers raising broods in three Colorado landscapes: Wilson Bulletin, v. 117, no. 2, p. 128–132.

Knopf, F.L., 1994, Avian assemblages on altered grasslands, *in* Jehl, J.R., and Johnson, N.K., eds., A century of avifaunal change in western North America: Lawrence, Kans., Allen Press, Cooper Ornithological Society, Studies in Avian Biology, no. 15, p. 247–257.

Knopf, F.L., and Rupert, J.R., 1999, Use of cultivated fields by breeding mountain plovers in Colorado, *in* Jehl, J.R., and Johnson, N.K., eds., A century of avifaunal change in western North America: Lawrence, Kans., Allen Press, Cooper Ornithological Society, Studies in Avian Biology, no. 15, p. 81–86.

Knopf, F.L., and Wunder, M.B., 2006, Mountain plover (*Charadrius montanus*), *in* Poole, A., eds., The birds of North America: Ithaca, N.Y., Cornell Lab of Ornithology, no. 211, accessed March 2014 at http://bna.birds.cornell.edu/bna/species/211.

Pierce, A.K., Dinsmore, S.J., Jorgensen, D., and Wunder, M.B., 2017, Migration routes and timing of mountain plovers revealed by geolocators: Journal of Field Ornithology, v. 88, p. 30–38.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100.

Samson, F.B., Knopf, F.L., and Ostlie, W.R., 2004, Great Plains ecosystems—Past, present, and future: Wildlife Society Bulletin, v. 32, no. 1, p. 6–15.

Shackford, J.S., Leslie, D.M., and Harden, W.D., 1999, Range-wide use of cultivated fields by mountain plovers during the breeding season: Journal of Field Ornithology, v. 70, no. 1, p. 114–120.

Tipton, H.C., Doherty, P.D., Jr., and Dreitz, V.J., 2009, Abundance and density of mountain plover (*Charadrius montanus*) and burrowing owl (*Athene cunicularia*) in eastern Colorado: Auk, v. 126, no. 3, p. 493–499.

U.S. Fish and Wildlife Service, 2011, Endangered and threatened wildlife and plants—Withdrawal of the proposed rule to list the mountain plover as threatened [Proposed rule, withdrawal]: Federal Register, v. 76, no. 92, p. 27756–27799.

Wood, J.A., Carter, S.K., Litshert, S.E., and Carr, N.B., 2017, Including broad-scale indicators in multiscale natural resource monitoring and assessment programs in the Bureau of Land Management, chap. 3 *of* Carter, S.K., Carr, N.B., Miller, K.H., and Wood, D.J.A., eds., Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, p. 15–37, accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Woolley, C., 2015, Mountain plover, chap. 17 *of* Assal, T.J., Melcher, C.P., and Carr, N.B., eds., 2015, Southern Great Plains Rapid Ecoregional Assessment—Pre-assessment report: U.S. Geological Survey Open-File Report 2015–1003, p. 118–126, accessed July 13, 2017, at https://doi.org/10.3133/ofr20151003.

Wunder, M.B., and Knopf, F.L., 2003, The Imperial Valley of California is critical to wintering mountain plovers: Journal of Field Ornithology, v. 74, no. 1, p. 74–80.

Chapter 8. Long-Billed Curlew

Introduction

Long-billed curlews (*Numenius americanus*) breed throughout the shortgrass and mixed-grass prairies of the Great Plains. Their winter range, which is largely located outside of the SGP, includes the Gulf Coast of Texas, southern California, and much of Mexico (Dugger and Dugger, 2002). Long-billed curlews experienced population declines as a consequence of market hunting in the late 1800s and conversion of native prairies by agricultural activities (Jones and others, 2008). The population size is currently estimated to be approximately 140,000 (Andres and others, 2012). Population trend estimates for the long-billed curlew from 1966 to 2012 indicate that populations are generally stable but may be declining in some areas (Dugger and Dugger, 2002; Sauer and others, 2014).

The curlew feeds on burrowing invertebrates and terrestrial insects, arachnids, and small vertebrates (Goater and Bush, 1986; Redmond and Jenni, 1986; Dugger and Dugger, 2002). During the breeding season, curlews forage and nest primarily in native grasslands dominated by short- and medium-height grasses. Curlews will also use pastures, haylands, and agricultural fields but avoid shrublands and woodlands (Pampush and Anthony, 1993; Dugger and Dugger, 2002; Dechant and others, 2002; Saalfeld and others, 2010). Habitat models by Saalfeld and others (2010), however, suggest that emergent wetlands in the vicinity of nests may be an important habitat feature. Nonbreeding habitat includes shortgrass prairie, sparsely vegetated playas and shallow wetlands, and fallow or harvested agricultural fields. The curlews can exhibit strong fidelity to breeding, migratory stopover, and wintering sites (Page and others, 2014).

Long-billed curlew. Photograph by Mike Baird (Creative Commons Attribution 2.0 Generic).

Wildfire, drought, and herbivory help maintain short-statured grasslands preferred by nesting curlews (Belovsky and others, 2000). They use a broader range of vegetation heights for feeding, including prairie dog colonies, and rear their broods in areas with taller vegetation (Dugger and Dugger, 2002; Derner and others, 2009). Therefore, landscapes that have substantial heterogeneity in grassland structure may increase suitability for breeding curlews throughout their nesting cycle. Pasture and agricultural fields that resemble grassland structure preferred by curlews may be also be used for nesting and brood rearing.

The potential effects of energy development, including oil, natural gas, and wind, have not been specifically addressed for the curlews. However, many species of birds avoid infrastructure related to energy development, which can destroy or degrade habitats, and are vulnerable to mortality from collisions with wind turbine blades. Agricultural fields are used by curlews during breeding, but mechanical disturbance can damage active nests (Cochran and Anderson, 1987; Dechant and others, 2002). Nesting curlews are often positively associated with grazing, which can help maintain preferred grassland structure (Derner and others, 2009). Depending on the timing and frequency, fires can decrease grass cover and increase habitat suitability for curlews (Dugger and Dugger, 2002); fire suppression that leads to encroachment of woody vegetation can reduce the overall amount of available curlew habitat. Exotic crested wheatgrass (*Agropyron cristatum*) and knapweed (*Centaurea* spp.) can reduce habitat quality for nesting curlews, whereas the shorter, sparser invasive cheatgrass (*Bromus tectorum*) appears to provide suitable nesting substrate (Dugger and Dugger, 2002). Additional background information on long-billed curlews can be found in the SGP pre-assessment report (Skagen, 2015).

Rapid Ecoregional Assessment Components Evaluated for the Long-Billed Curlew

The key ecological attributes and change agents addressed by core management questions for long-billed curlew habitat include amount and distribution, landscape structure, and development (tables 8–1 and 8–2). Invasive woody species and climate change were evaluated for grassland communities (see Reese and others, 2017, chap. 4). Fire occurrence and potentially altered vegetation (including invasive herbaceous plants) were evaluated for the entire SGP (see Reese and others, 2017, chap. 3). Overall landscape-level ranking variables are summarized in table 8–3. The core and integrated management questions are listed in table 8–4.

Table 8–1. Key ecological attributes and associated indicators used to address core management questions for long-billed curlews for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Amount and distribution	Total area	Baseline habitat (year round)[2]
Landscape structure	Patch size	Patch sizes for baseline habitat
Landscape dynamics	Fire occurrence	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]MaxEnt was used to predict baseline habitat. See chapter 2, "Methods Overview for Species," for methods and datasets used.

Table 8–2. Anthropogenic change agents and associated indicators used to address core management questions for long-billed curlews for the Southern Great Plains Rapid Ecoregional Assessment.

[km, kilometer; mi, mile]

Attributes	Variables	Indicators[1]
Development	Terrestrial development index (TDI)	Percentage of baseline habitat in seven development classes based on a 2.5-km (1.55-mi) moving window
	Index of fragmentation	Patch sizes for relatively undeveloped[2] habitat
Invasive species	Potential for woody species expansion	See Reese and others (2017, chap. 4)
Climate change	Potential distribution of grasslands	See Reese and others (2017, chap. 4)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]Terrestrial development index score less than or equal to 2 percent.

Table 8–3. Landscape-level variables used to address the integrated management question for long-billed curlews. Ranks for landscape-level habitat area and development were combined into an overall landscape-level rank for the Southern Great Plains Rapid Ecoregional Assessment.

[>, greater than; km, kilometer; mi, mile]

Landscape-level variables[1]	Description	Relative rank[2]		
		Lowest	Medium	Highest
Area	Percentage of baseline habitat within a 5-km-radius (3.11-mi) moving window	>0–29.5	>29.5–46.2	>46.2
Development	Mean terrestrial development index (TDI) score for baseline habitat within a 5km-radius (3.11-mi) moving window	0–2	>2–10	>10

[1]See Reese and others (2017, chap. 2) for methods and datasets used to address integrated management questions.

[2]Ranking breakpoints for area of baseline habitat were determined from equal subsets of the data. Ranking breakpoints for terrestrial development index scores were standardized for all terrestrial conservation elements.

Table 8–4. Management questions addressed for long-billed curlews for the Southern Great Plains Rapid Ecoregional Assessment.

Core management questions[1]	Results
What is the distribution of baseline habitat for long-billed curlews?	Figures 8−1
Where does existing development pose the greatest threat to long-billed curlew habitat, and where are the large, relatively undeveloped areas?	Figures 8−2 and 8−3
How has development fragmented long-billed curlew habitat?	Figures 8−4 and 8−5
Integrated management question[2]	**Results**
Where is long-billed curlew habitat with the highest overall landscape-level rank?	Figure 8−6

[1]See Reese and others (2017, chap. 11) for management questions that could not be addressed.

[2]See table 8−3.

Management Questions and Results

What is the distribution of baseline habitat for long-billed curlews (fig. 8–1)?

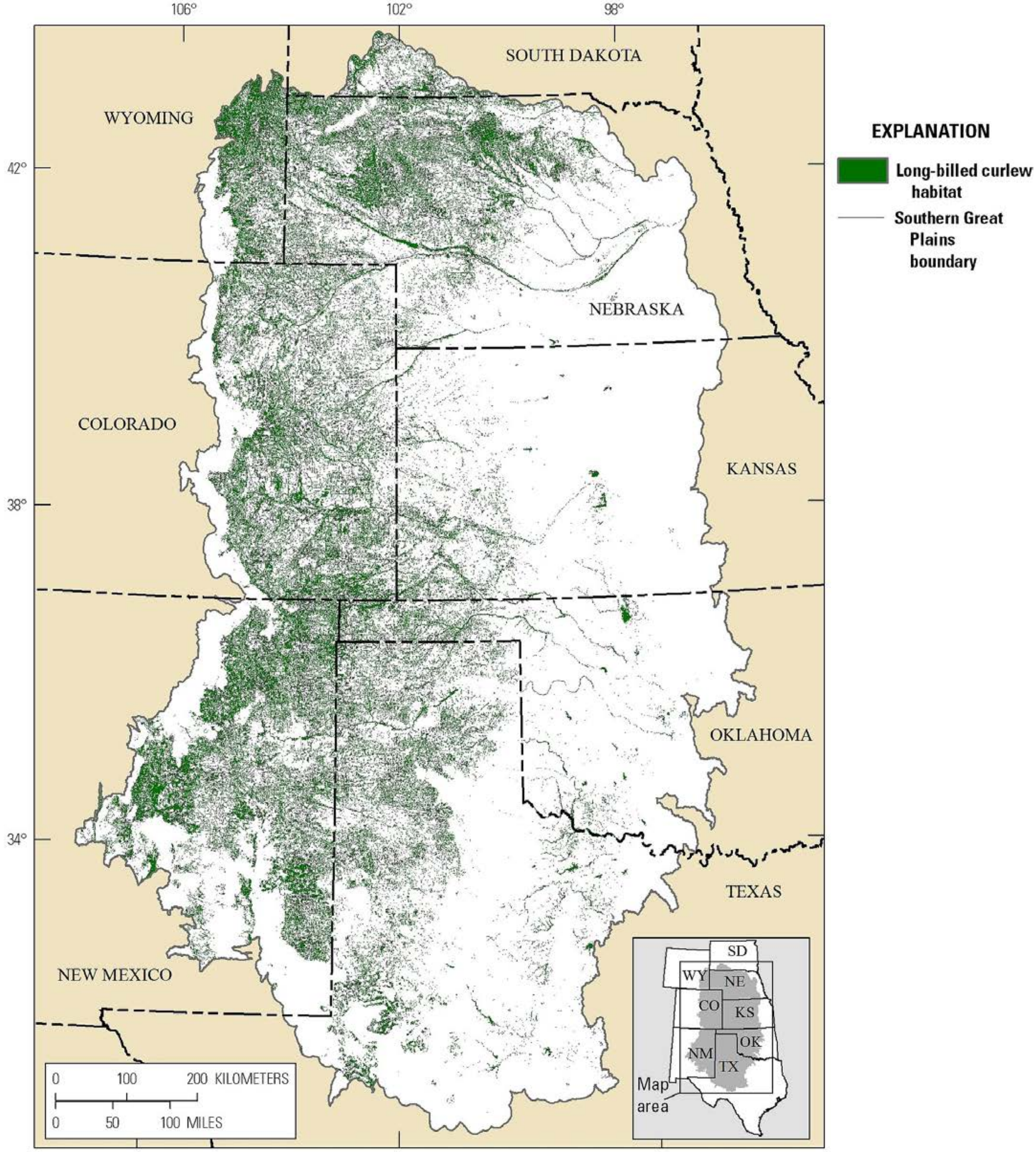


Figure 8–1. Predicted distribution of baseline habitat for long-billed curlews in the Southern Great Plains.

Where does existing development pose the greatest threat to long-billed curlew habitat, and where are the large, relatively undeveloped areas (figs. 8–2 and 8–3)?

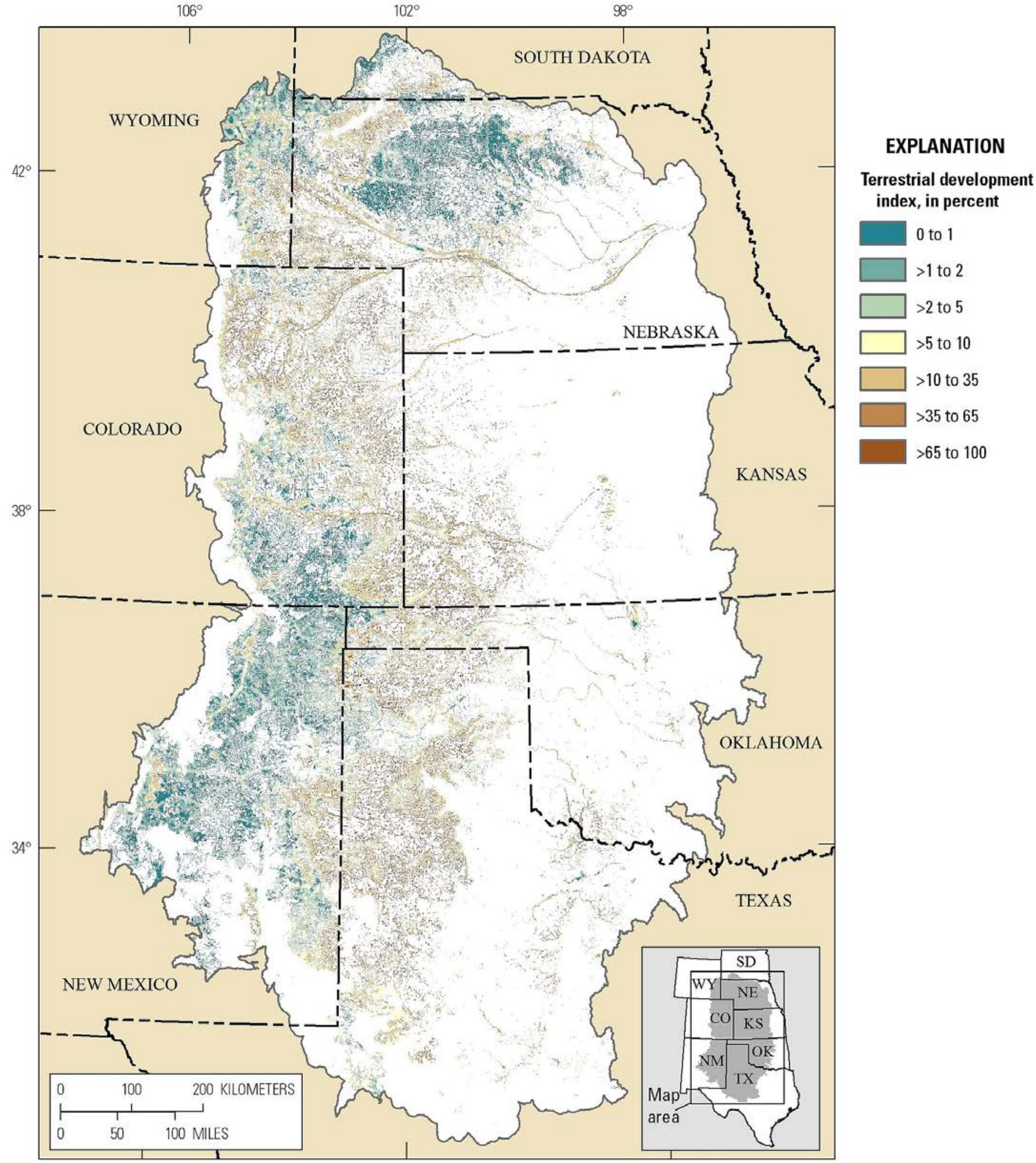


Figure 8–2. Terrestrial development index for long-billed curlew baseline habitat in the Southern Great Plains.

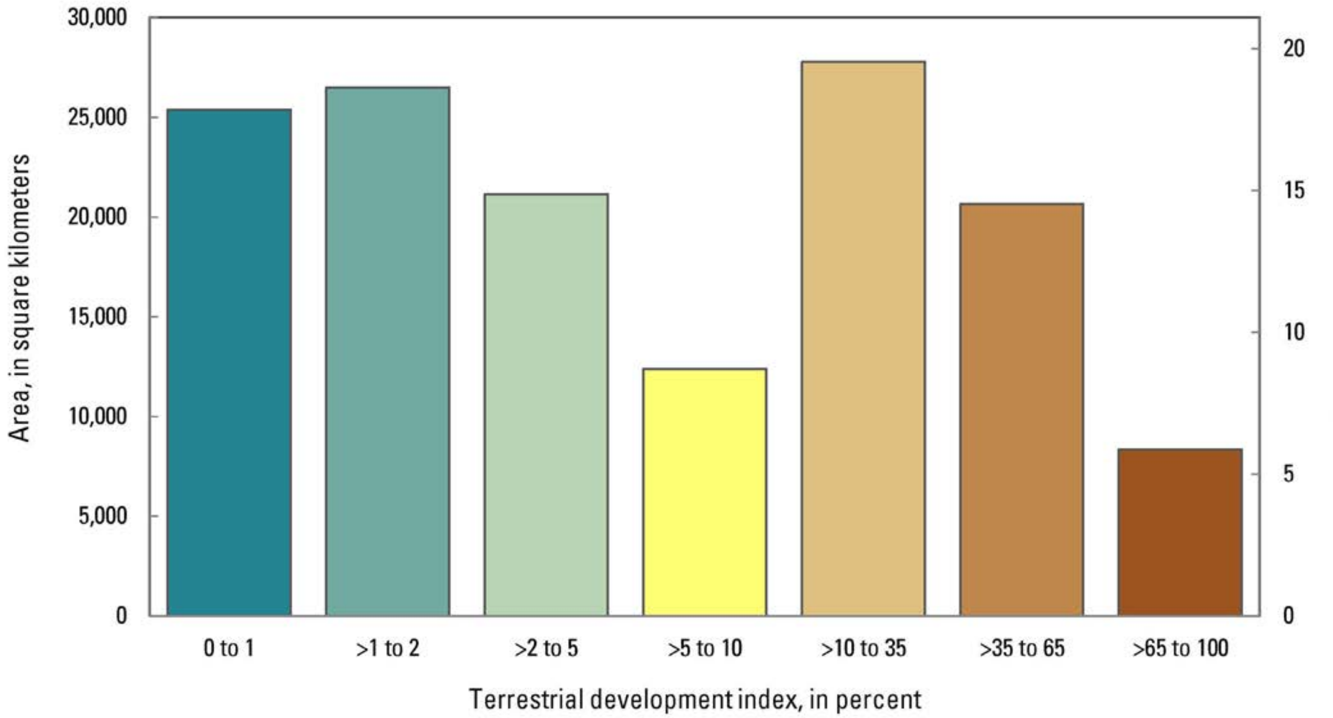


Figure 8–3. Area of long-billed curlew baseline habitat by terrestrial development index class in the Southern Great Plains.

How has development fragmented long-billed curlew habitat (figs. 8–4 and 8–5)?

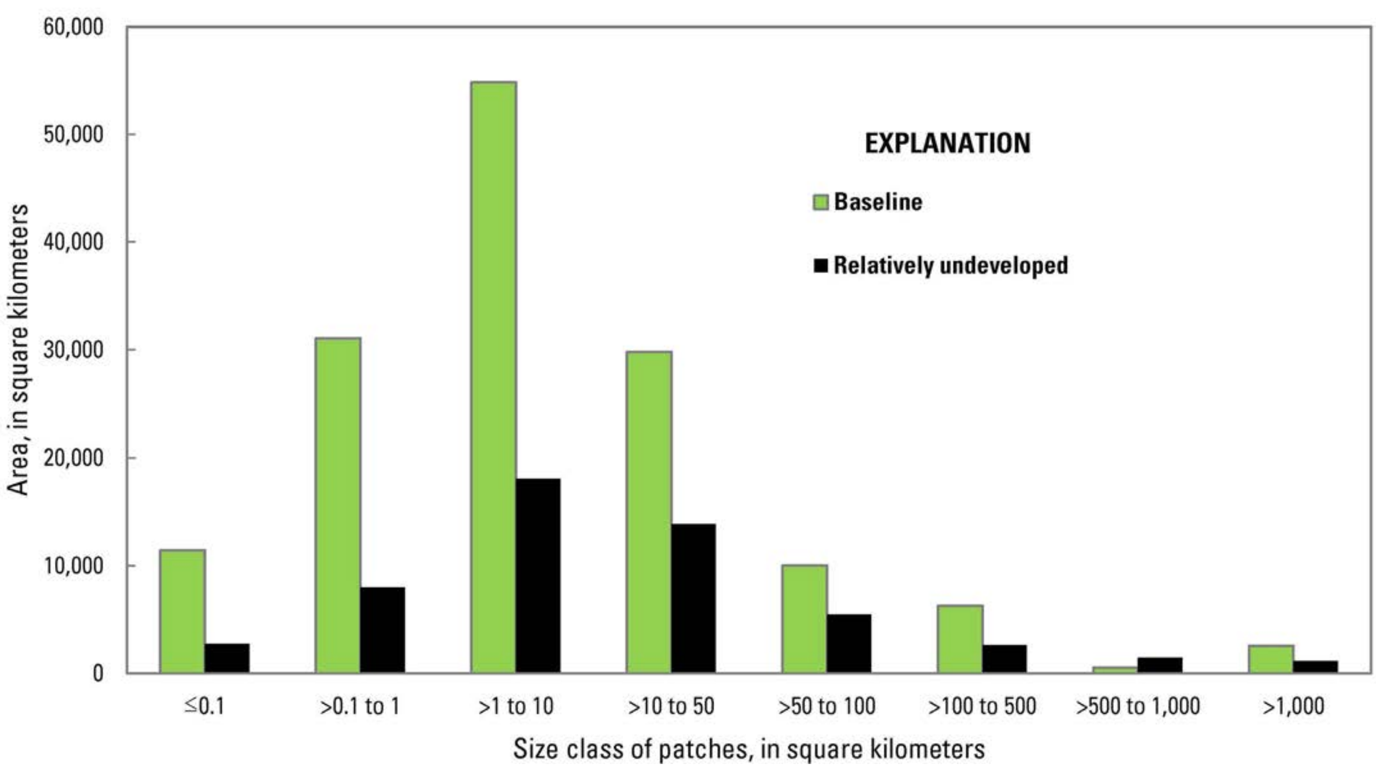


Figure 8–4. Area of long-billed curlew habitat in the Southern Great Plains by patch size class for baseline and relatively undeveloped conditions (terrestrial development index score ≤2 percent).

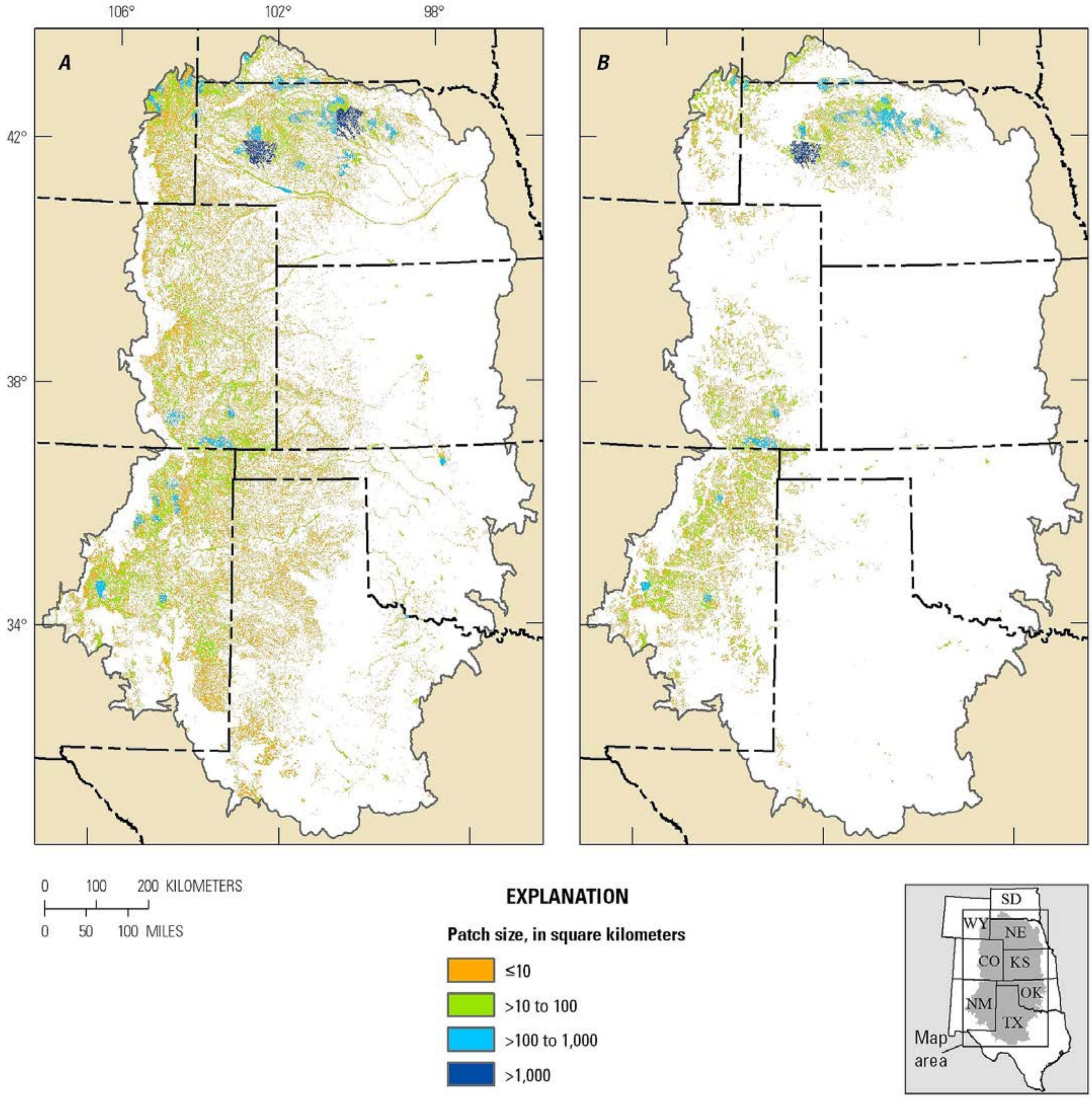


Figure 8–5. Patch size of long-billed curlew habitat in the Southern Great Plains. *A,* Baseline habitat. *B,* Relatively undeveloped habitat (terrestrial development index score ≤2 percent).

Where is long-billed curlew habitat with the highest overall landscape-level rank (fig. 8–6)?

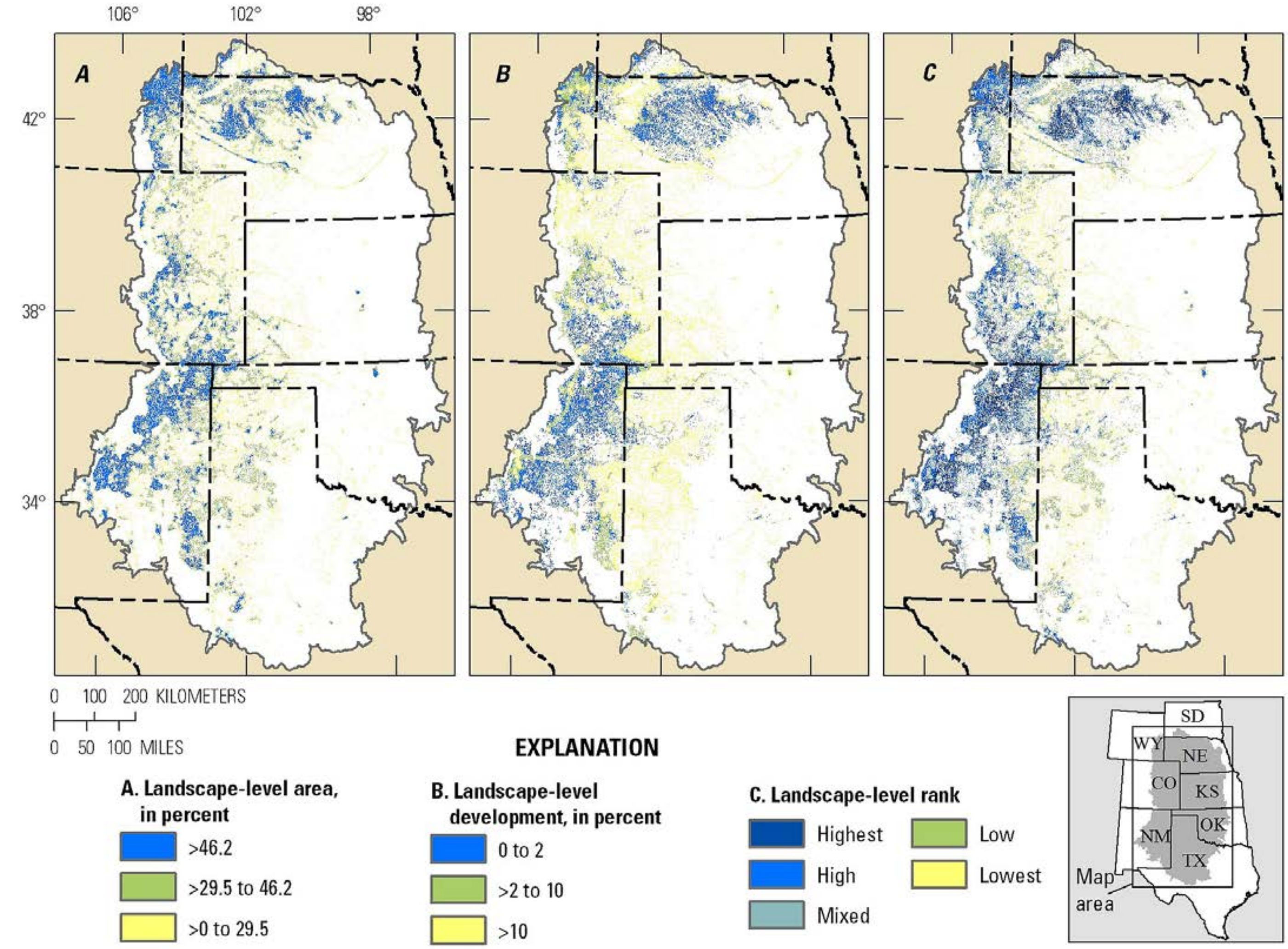


Figure 8–6. Landscape-level summaries for long-billed curlew habitat in the Southern Great Plains. Overall landscape-level rank (*C*) is derived from (*A*) landscape-level habitat area and (*B*) landscape-level development, summarized by a 5-kilometer-radius (3.11-mile) moving window (see table 8–3). Highest overall landscape-level rank corresponds to the largest landscape-level area and the lowest landscape-level development. Lowest overall landscape-level rank corresponds to the smallest landscape-level area and highest landscape-level development. Landscape-level ranks are not intended as standalone summaries and are best interpreted in conjunction with the geospatial datasets used to address core management questions.

Summary

- Baseline habitat for long-billed curlews occurs throughout the SGP but is concentrated along the western and northern portions (fig. 8–1). There are approximately 146,000 km^2 (56,371 mi^2) of baseline habitat in the SGP.
- Long-billed curlew habitat with the lowest development is located predominantly in the northern and southwestern portions of the SGP (fig. 8–2). Approximately 36 percent of its habitat is relatively undeveloped (TDI score ≤2 percent), and 15 percent has low development (TDI scores 2–5 percent). However, 20 percent has very high levels of development (TDI score >35 percent) (fig. 8–3).
- Long-billed curlew habitat is highly fragmented across the entire SGP, and the largest habitat patches occur in the Nebraska Sand Hills (figs. 8–4 and 8–5*A*). Approximately 87 percent of baseline habitat occurs in patches smaller than 50 km^2 (19.3 mi^2) (fig. 8–4). Relatively undeveloped patches are concentrated in Nebraska, southeastern Colorado, and eastern New Mexico (fig. 8–5*B*).
- The largest, most intact areas (the highest overall landscape-level rank) are in the northern and southwestern portions of the SGP (fig. 8–6*C*), where there has been less conversion to croplands than in other areas of the region (Reese and others, 2017).
- The broad-scale summaries provided by the REA are intended to be used in conjunction with local-level information on habitat conditions (Wood and others, 2017).

References Cited

Andres, B.A., Smith, P.A., Morrison, R.I.G., Gratto-Trevor, C.L., Brown, S.C., and Friis, C.A., 2012, Population estimates of North American shorebirds, 2012: Wader Study Group Bulletin, v. 119, no. 3, p. 178–194.

Belovsky, G.E., Joern, A., and Lockwood, J., 2000, Grasshoppers—Plus and minus—The grasshopper problem on a regional basis and a look at beneficial effects of grasshoppers, section VII.16 *of* Cunningham, G.L., and Sampson, M.W., eds., Grasshopper integrated pest management user handbook: U.S. Department of Agriculture, Animal and Plant Health Inspection Service, USDA/APHIS Technical Bulletin 1809, 5 p., accessed March 2014 at https://www.sidney.ars.usda.gov/grasshopper/Handbook/VII/vii_16.htm.

Cochran, J.F., and Anderson, S.H., 1987, Comparison of habitat attributes at sites of stable and declining long-billed curlew populations: Great Basin Naturalist, v. 47, p. 459–466.

Dechant, J.A., Sondreal, M.L., Johnson, D.H., Igl, L.D., Goldade, C.M., Rabie, P.A., and Euliss, B.R., 2002, Effects of management practices on grassland birds—Long-billed curlew: Jamestown, N. Dak., U.S. Geological Survey, Northern Prairie Wildlife Research Center, accessed June 3, 2014, at http://www.npwrc.usgs.gov/resource/literatr/grasbird/download/lbcu.pdf.

Derner, J.D., Lauenroth, W.K., Stapp, P., and Augustine, D.J., 2009, Livestock as ecosystem engineers for grassland bird habitat in the western Great Plains of North America: Rangeland Ecology and Management, v. 62, no. 2, p. 111–118.

Dugger, B.D., and Dugger, K.M., 2002, Long-billed curlew (*Numenius americanus*), *in* Poole, A., ed., The birds of North America: Ithaca, N.Y., Cornell Lab of Ornithology, no. 628, accessed January 2018 at https://birdsna.org/Species-Account/bna/species/lobcur.

Goater, C.P., and Bush, A.O., 1986, Nestling birds as prey of breeding long-billed curlews, *Numenius americanus*: Canadian Field Naturalist, v. 100, p. 263–264.

Jones, S.L., Nations, C.S., Fellows, S.D., and McDonald, L.L., 2008, Breeding abundance and distribution of long-billed curlews (*Numenius americanus*) in North America: Waterbirds, v. 31, p. 1–14.

Page, G.W., Warnock, N., Tibbits, T.L., Jorgensen, D., Hartman, C.A., and Stenzel, L.E., 2014, Annual migratory patterns of long-billed curlews in the American West: Condor, v. 116, p. 50–61.

Pampush, G.J., and Anthony, R.G., 1993, Nest success, habitat utilization and nest-site selection of long-billed curlews in the Columbia Basin, Oregon: Condor, v. 95, p. 957–967.

Redmond, R.L., and Jenni, D.A., 1986, Population ecology of the long-billed curlew (*Numenius americanus*) in western Idaho: Auk, v. 103, p. 755–767.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100

Saalfeld, S.T., Conway, W.C., Haukos, D.A., Rice, M., Jones, S.L., and Fellows, S.D., 2010, Multiscale habitat selection by long-billed curlews (*Numenius americanus*) breeding in the United States: Waterbirds, v. 33, p. 148–161.

Sauer, J.R., Hines, J.E., Fallon, J.E., Pardieck, K.L., Ziolkowski, D.J., Jr., and Link, W.A., 2014, The North American breeding bird survey, results and analysis, 1966–2012 (version 02.19.2014): Laurel, Md., U.S. Geological Survey, Patuxent Wildlife Research Center, accessed March 2014 at http://www.mbr-pwrc.usgs.gov/bbs/.

Skagen, S.K., 2015, Long-billed curlew, chap. 16 *of* Assal, T.J., Melcher, C.P., and Carr, N.B., eds., 2015, Southern Great Plains Rapid Ecoregional Assessment—Pre-assessment report: U.S. Geological Survey Open-File Report 2015–1003, p. 118–126, accessed July 13, 2017, at https://doi.org/10.3133/ofr20151003.

Wood, J.A., Carter, S.K., Litshert, S.E., and Carr, N.B., 2017, Including broad-scale indicators in multiscale natural resource monitoring and assessment programs in the Bureau of Land Management, chap. 3 *of* Carter, S.K., Carr, N.B., Miller, K.H., and Wood, D.J.A., eds., Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, p. 15–37, accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Chapter 9. Interior Least Tern

Introduction

The least tern (*Sternula antillarum*) is the smallest of the North American terns. The interior least tern (*S. a. athalassos*) breeds along rivers and reservoirs of the United States. Genetic distinctions among subspecies of *S. antillarum* remain equivocal (Whittier and others, 2006; Draheim and others, 2010, 2012), but geographic and ecological factors physically separate the interior least tern from the east coast and west coast populations. Least terns experienced severe declines at the turn of the last century because of egg and feather collection until passage of the Migratory Bird Treaty Act of 1916 (Thompson and others, 1997). Degradation and loss of nesting habitat resulting from river channelization and dam building contributed to declines in interior least tern populations (U.S. Fish and Wildlife Service, 1985). More recently, channel management practices, including dike field and dredging operations, have been used to help create and maintain interior least tern habitat (U.S. Fish and Wildlife Service, 2013). The interior least tern was listed as federally endangered in 1985, at which time the interior least tern population was estimated at 1,970 (U.S. Fish and Wildlife Service, 1985, 2013). In 2012, the population was estimated at more than 13,855, and because population recovery goals had been achieved, it was recommended that the subspecies be delisted (U.S. Fish and Wildlife Service, 2013). The nesting habitat requirements of the federally endangered piping plover (*Charadrius melodus*) is similar to that of the interior least tern, and the two species' ranges overlap in the northern Great Plains (U.S. Fish and Wildlife Service, 2009).

The interior least tern historically nested primarily on coarse sandy substrates found along river sandbars (Kirsch, 1996; Lott and others, 2013). The species commonly nests on islands, but it will also use beaches, sand banks, and point bars (Lott and Wiley, 2012; Lott and others, 2013). In many areas, the loss of natural habitats and the availability of novel nesting conditions created and maintained by human activities have led to increased use of shorelines and islands associated with reservoirs, dikes, dredge sites, and gravel pits, as well as industrial sites and gravel rooftops (Butcher and others, 2007; Forys and others, 2013; Stucker and others, 2013; Baasch and others, 2017). The terns primarily feed on small fish and fingerlings and forage in shallow open waters of rivers, marshes, ponds, and reservoirs near colonies (Thompson and others, 1997; Sherfy and others, 2011; Lott and others, 2013).

Least tern. Photograph by Scott Heron (Creative Commons Attribution-ShareAlike 2.0 Generic).

The least tern is a colonial nester that uses ephemeral habitats, and social factors likely play a role in nest-site selection (Kotliar and Burger, 1984; Ward and others, 2011). Least terns often abandon colony sites after predation events or with declines in habitat quality (Burger, 1984; Ward and others, 2011). As a colonial nesting species, the eggs and chicks are vulnerable to predation by a variety of avian and mammalian species and mortality from natural and human disturbances.

The nesting and foraging habitats of interior least terns can be ephemeral and dynamic under natural river flow regimes and vegetation encroachment between flooding events. On rivers, flooding can create and maintain nesting habitat; accordingly least terns readily colonize newly created habitat throughout its range (Sidle and others, 1992). Regulated flows can disrupt the natural processes that create and maintain sand and gravel bars where interior least terns nest (U.S. Fish and Wildlife Service, 1985). During the nesting season, natural flooding events and water releases from dams can cause nesting failure and mortality of unfledged young. Nesting habitat of the interior least tern is also susceptible to vegetation encroachment, particularly from nonnative plant species such as tamarisk (*Tamarix* spp.) that colonize disturbed areas (Schweitzer and Leslie, 1999; Winton and Leslie, 2003). Additional background information on interior least terns can be found in the SGP pre-assessment report (Zeigenfuss, 2015).

Rapid Ecoregional Assessment Components Evaluated for the Interior Least Tern

The key ecological attributes and change agents addressed by core management questions for interior least tern habitat include the amount and distribution, and development (tables 9–1 and 9–2). Invasive woody species were evaluated for riparian and wetland communities (see Reese and others, 2017, chap. 8). Fire occurrence and climate change were evaluated for the entire SGP (see Reese and others, 2017, chap. 3). Overall landscape-level ranking variables are summarized in table 9–3. The core and integrated management questions are listed in table 9–4.

Table 9–1. Key ecological attributes and associated indicators used to address core management questions for interior least terns for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Amount and distribution	Total area	Baseline habitat (breeding)[2]
Landscape dynamics	Fire occurrence	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]Baseline habitat was estimated using data from the U.S. Geological Survey Gap Analysis Program (2011). See chapter 2, “Methods Overview for Species,” for methods and datasets used.

Table 9–2. Anthropogenic change agents and associated indicators used to address core management questions for interior least terns for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Development	Aquatic development index (ADI)	Percentage of baseline habitat in seven development classes by sixth-level watershed
Invasive species	Potential for woody species expansion	See Reese and others (2017, chap. 8)
Climate change	Projected temperature and precipitation	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

Table 9–3. Landscape-level variables used to address the integrated management question for interior least terns. Ranks for landscape-level habitat area and development were combined into an overall landscape-level rank for the Southern Great Plains Rapid Ecoregional Assessment.

[>, greater than]

Landscape-level variables[1]	Description	Relative rank[2]		
		Lowest	Medium	Highest
Area	Area of interior least tern baseline habitat as a percentage of fifth-level watershed	>0–9.0	>9.0–26.0	>26.0
Development	Mean aquatic development index (ADI) score for interior least tern habitat, summarized by fifth-level watershed	0–20	>20–40	>40

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address integrated management questions.

[2]Ranking breakpoints for area of baseline habitat were determined from equal subsets of the data. Ranking breakpoints for the aquatic development index were standardized for all aquatic species and assemblages.

Table 9–4. Management questions addressed for interior least terns for the Southern Great Plains Rapid Ecoregional Assessment.

Core management questions[1]	Results
What is the distribution of baseline habitat for interior least terns?	Figure 9−1
Where does existing development pose the greatest threat to interior least tern habitat, and where are the large, relatively undeveloped areas?	Figures 9−2 and 9−3
Integrated management question[2]	**Results**
Where is interior least tern habitat with the highest overall landscape-level rank?	Figure 9−4

[1]See Reese and others (2017, chap. 11) for management questions that could not be addressed.

[2]See table 9−3.

Management Questions and Results

What is the distribution of baseline habitat for interior least terns (figs. 9–1)?

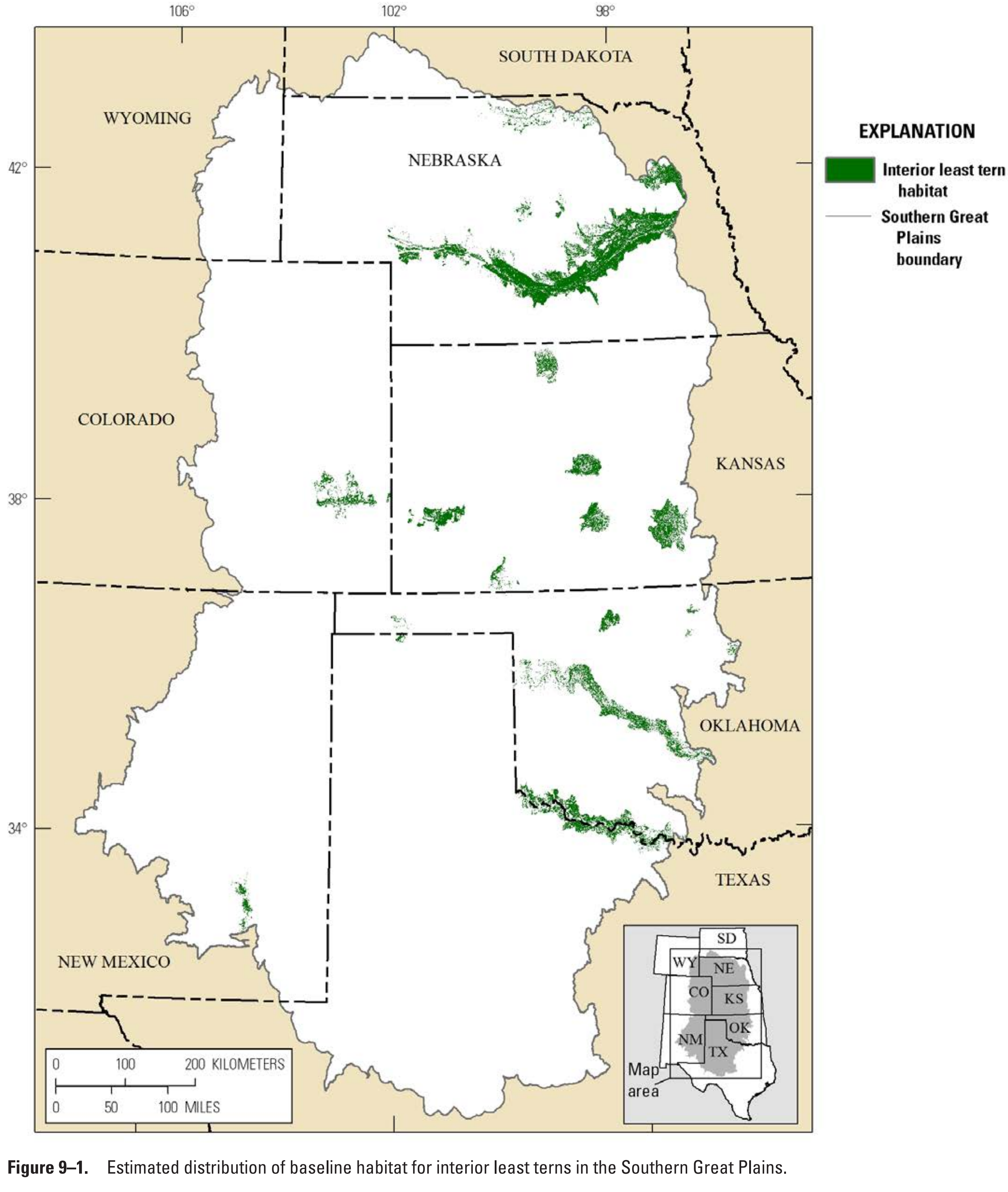


Figure 9–1. Estimated distribution of baseline habitat for interior least terns in the Southern Great Plains.

Where does existing development pose the greatest threat to interior least tern habitat, and where are the large, relatively undeveloped areas (figs. 9–2 and 9–3)?

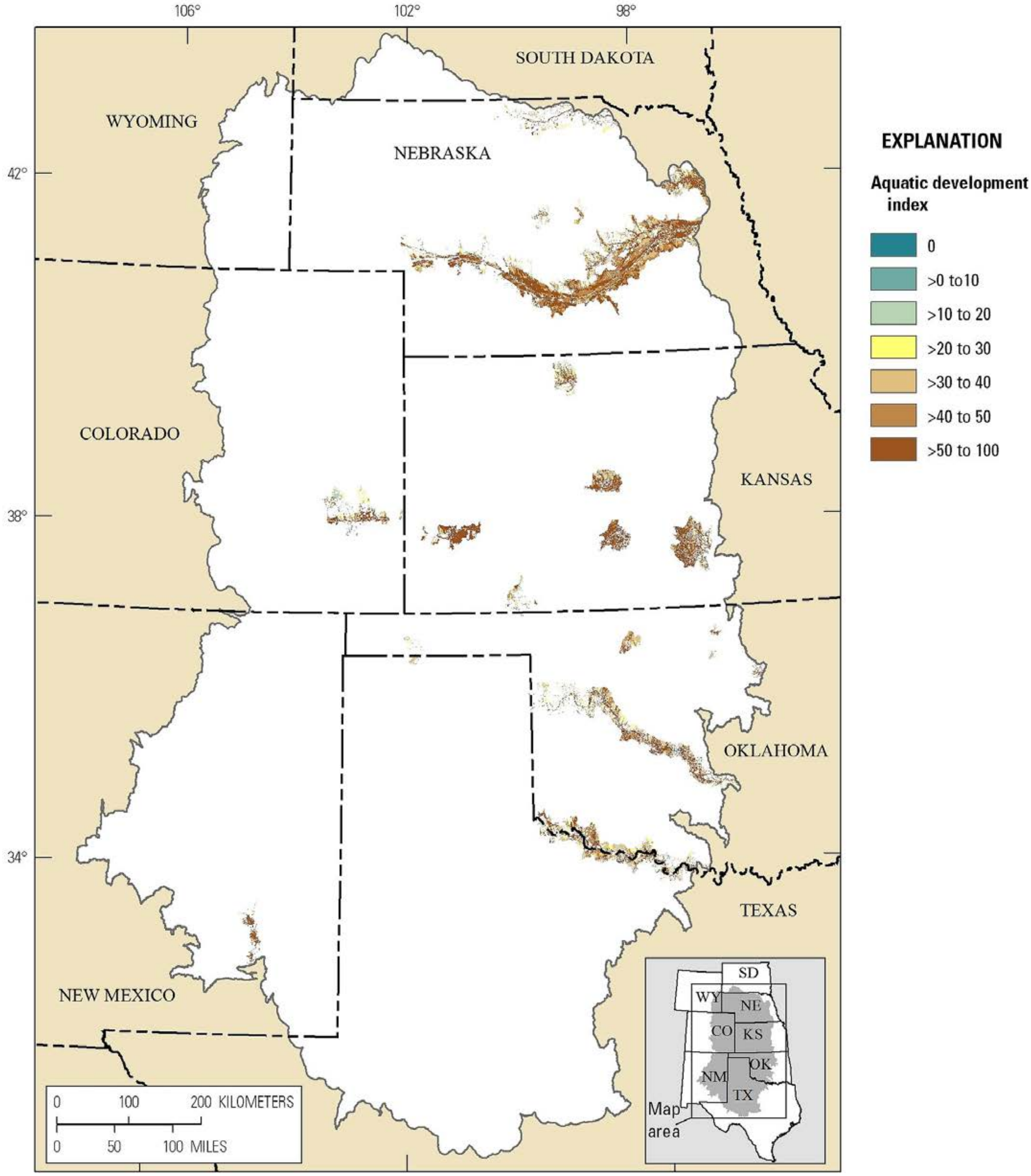


Figure 9–2. Aquatic development index for interior least tern baseline habitat in the Southern Great Plains.

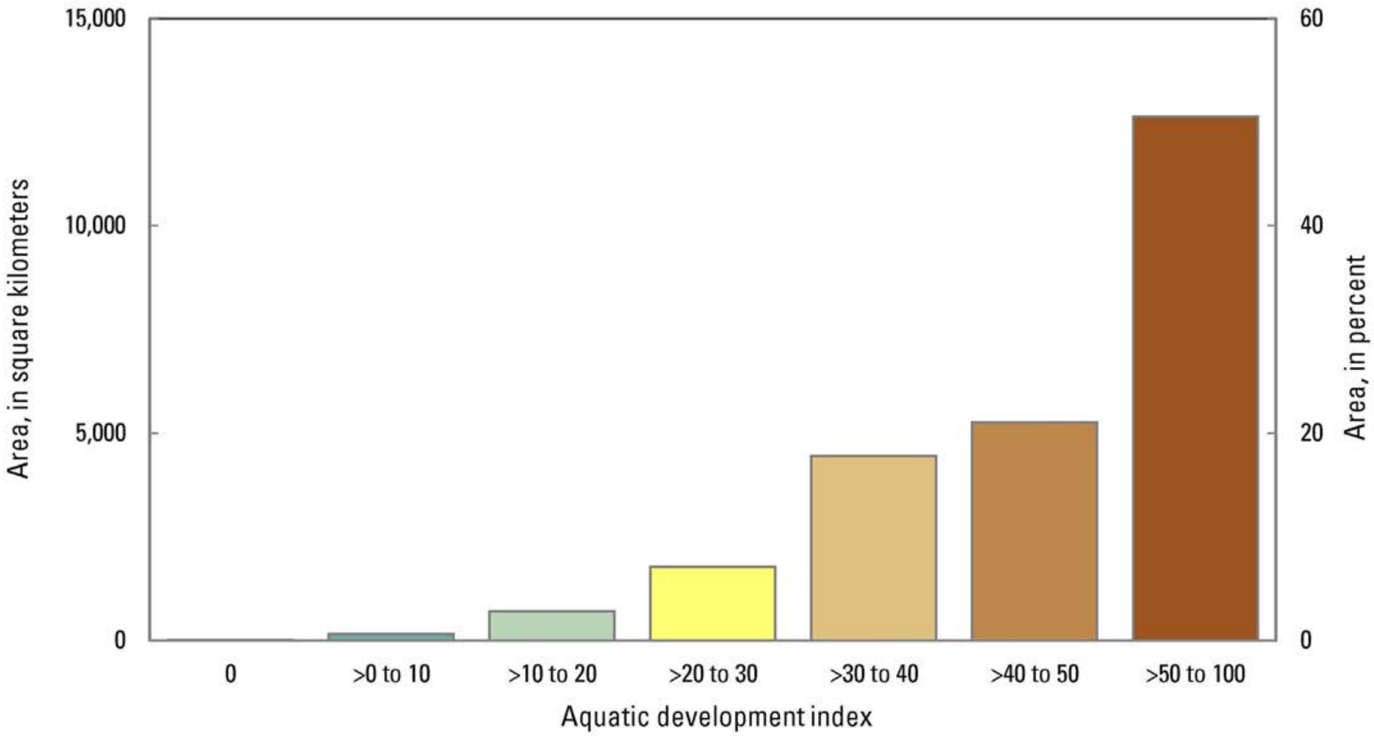


Figure 9–3. Area of interior least tern baseline habitat by aquatic development index class in the Southern Great Plains.

Where is interior least tern habitat with the highest overall landscape-level rank (fig. 9–4)?

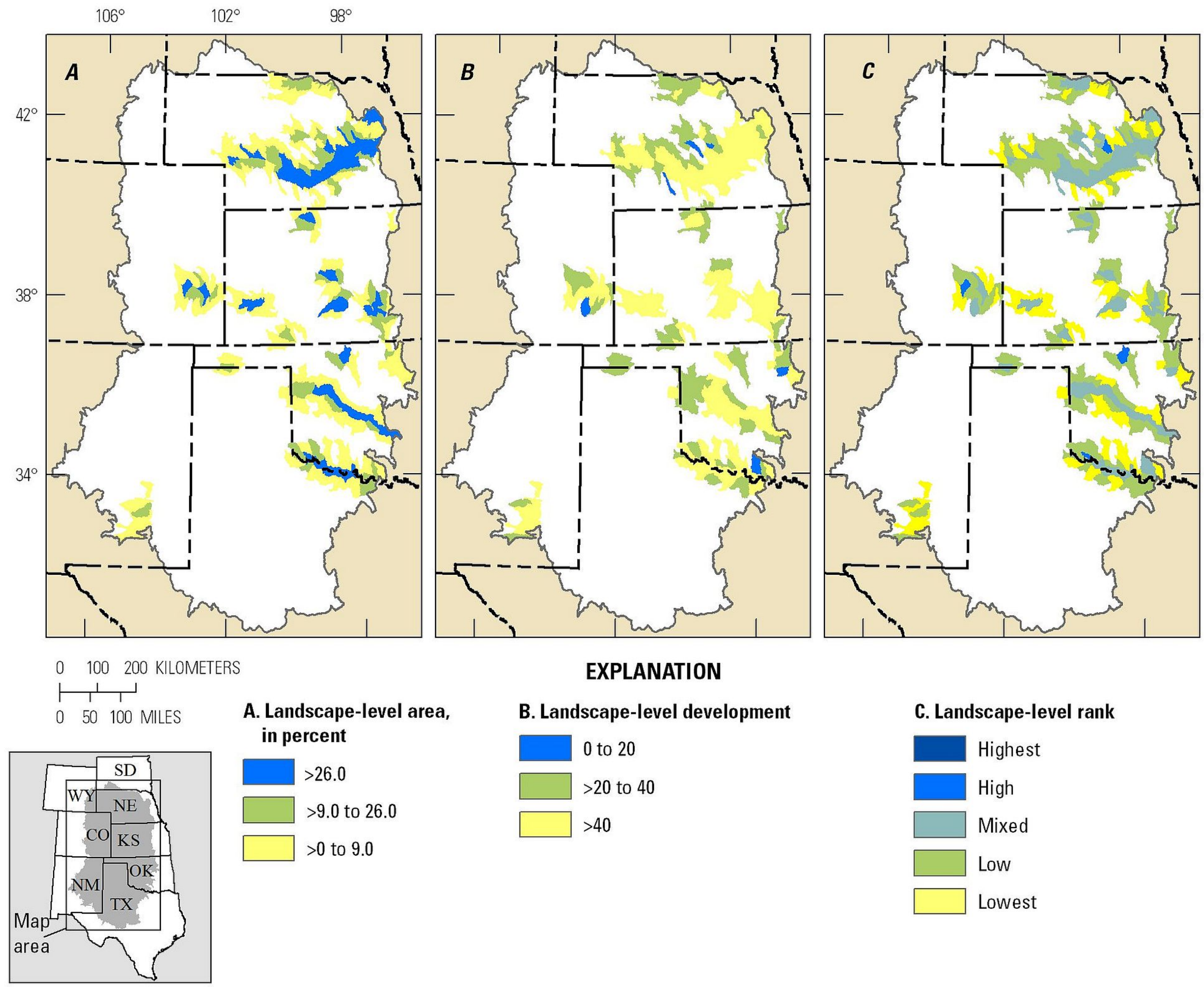


Figure 9–4. Landscape-level summaries for interior least tern habitat in the Southern Great Plains. Overall landscape-level rank (*C*) is derived from (*A*) landscape-level habitat area and (*B*) landscape-level development, summarized by fifth-level watershed (see table 9–3). Highest overall landscape-level rank corresponds to the largest landscape-level area and the lowest landscape-level development. Lowest overall landscape-level rank corresponds to the smallest landscape-level area and highest landscape-level development. None of the watersheds had the highest overall ranking. Landscape-level ranks are not intended as standalone summaries and are best interpreted in conjunction with the geospatial datasets used to address core management questions.

Summary

- Baseline nesting habitat for interior least terns is primarily associated with rivers and reservoirs across the SGP (fig. 9–1). There are approximately 25,600 km^2 (9,884 mi^2) of baseline habitat in the SGP.
- Development levels are very high in almost all interior least tern nesting habitat (fig. 9–2). Approximately 3 percent of this habitat is relatively undeveloped (ADI score ≤20), and 7 percent has low development levels (ADI scores 20–30). However, 72 percent has very high levels of development (ADI score >40) (fig. 9–3). Consequently, habitat for interior least terns is largely restricted to highly altered or novel habitats, all of which typically involve active management by humans to create and maintain.
- The largest, most intact areas (high overall landscape-level rank) are in isolated watersheds in Nebraska, Colorado, and Oklahoma (fig. 9–4*C*). None of the watersheds were assigned the highest overall landscape-level rank—all fifth-level watersheds with the highest landscape-level area of baseline habitat (fig. 9–4*A*) also have high levels of development (fig. 9–4*B*).
- Despite high development levels throughout least tern habitat, the predisposition of least terns to use anthropogenic habitats that resemble natural habitats, in conjunction with active management targeting tern colonies, has helped to support increasing populations in many areas (Brown and others, 2011; U.S. Fish and Wildlife Service, 2013).
- The broad-scale summaries provide by the REA are intended to be used in conjunction with local-level information on habitat conditions (Wood and others, 2017).

References Cited

Baasach, D.M., Farrell, P.D., Farnsworth, J.M., and Smith, C.B., 2017, Nest-site selection by interior least terns and piping plovers at managed, off-channel sites along the central Platte River in Nebraska, USA: Journal of Field Ornithology, v. 88, p. 236–249.

Brown, M.B., Burbach, M.E., Dinan, J., Held, R., Johnson, R., Jorgensen, J.G., Lackey, J.L., Marcus, J., Matkin, G.S., and Thody, C., 2011, Nebraska's Tern and Plover Conservation Partnership—A model for sustainable conservation of threatened and endangered species: Wader Study Group Bulletin, v. 118, p. 22–25.

Burger, J., 1984, Colony stability in least terns: Condor, v. 86, no. 1, p. 61–67.

Butcher, J.A., Neill, R.L., and Boylan, J.T., 2007, Survival of interior least tern chicks hatched on gravel-cover roofs in north Texas: Waterbirds, v. 30, no. 4, p. 595–601.

Draheim, H.M., Baird, P., and Haig, S.M., 2012, Temporal analysis of mtDNA variation reveals decreased genetic diversity in least terns: Condor, v. 114, no. 1, p. 145–154.

Draheim, H.M., Miller, M.P., Baird, P., and Haig, S.M., 2010, Subspecific status and population genetic structure of least terns (*Sternula antillarum*) inferred by mitochondrial DNA control region sequences and microsatellite DNA: Auk, v. 127, p. 807–819.

Forys, E.A., Poppema-Bannon, A., Krajcik, K., and Szelistowski, W.A., 2013, Roof-nesting least terns travel to forage in brackish/marine waters: Southeastern Naturalist, v. 12, no. 1, p. 238–242.

Kirsch, E.M., 1996, Selection and productivity of least terns on the lower Platte River, Nebraska: Wildlife Monographs, no. 132, p. 3–48.

Koenen, M.T., Leslie, D.M., Jr., and Gregory, M., 1996, Habitat changes and success of artificial nests on an alkaline flat: Wilson Bulletin, v. 108, no. 2, p. 292–301.

Kotliar, N.B., and Burger, J., 1984, The use of decoys to attract least terns (*Sterna antillarum*) to abandoned colony sites in New Jersey: Colonial Waterbirds, v. 7, p. 134–138.

Lott, C.A., and Wiley, R.L., 2012, Effects of dam operations on least tern nesting habitat and reproductive success below Keystone Dam on the Arkansas River: U.S. Army Corps of Engineers, Engineering Research and Development Center, Environmental Lab Report ERDC/EL CR–12–4, 100 p., accessed February 28, 2014, at http://el.erdc.usace.army.mil/elpubs/pdf/crel-4.pdf.

Lott, C.A., Wiley, R.L., Fischer, R.A., Hartfield, P.D., and Scott, J.M., 2013, Interior least tern (*Sternula antillarum*) breeding distribution and ecology—Implications for population-level studies and the evaluation of alternative management strategies on large, regulated rivers: Ecology and Evolution, v. 3, no. 10, p. 3613–3627.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100.

Schweitzer, S.A., and Leslie, D.M., Jr., 1999, Nesting habitat of least terns (*Sterna antillarum athalassos*) on an inland alkaline flat: American Midland Naturalist, v. 142, p. 173–180.

Sherfy, M.H., Anteau, M.J., Shaffer, T.L., Sovada, M.A., and Stucker, J.H., 2011, Objectives, priorities, reliable knowledge, and science-based management of Missouri River interior least terns and piping plovers: U.S. Geological Survey Open-File Report 2011–1236, 26 p.

Sidle, J.G., Carlson, D.E., Kirsch, E.M., and Dinan, J.J., 1992, Flooding—Mortality and habitat renewal for least terns and piping plovers: Colonial Waterbirds, v. 15, no. 1, p. 132–136.

Stucker, J.H., Buhl, D.A., and Sherfy, M.A., 2013, Consequences of least tern (*Sternula antillarum*) microhabitat nest-site selection on natural and mechanically constructed sandbars in the Missouri River: Auk, v. 130, no. 4, p. 753–763.

Thompson, B.C., Jackson, J.A., Burger, J., Hill, L.A., Kirsch, E.M., and Atwood, J.L., 1997, Least tern (*Sterna antillarum*), *in* Poole, A., ed., The birds of North America: Ithaca, N.Y., Cornell Lab of Ornithology, no. 290, accessed February 2014 at http://bna.birds.cornell.edu/bna/species/290.

U.S. Fish and Wildlife Service, 1985, Endangered and threatened wildlife and plants—Interior population of the least tern determined to be endangered [Final rule]: Federal Register, v. 50, no. 102, p. 21784–21792.

U.S. Fish and Wildlife Service, 2009, Piping plover (*Charadrius melodus*)—5-year review—Summary and evaluation: U.S. Fish and Wildlife Service, 206 p., accessed February 25, 2014, at http://www.fws.gov/northeast/endangered/pdf/piping_plover_five_year_review_and_summary.pdf.

U.S. Fish and Wildlife Service, 2013, Interior least tern (*Sternula antillarum*)—5-year review—Summary and evaluation: U.S. Fish and Wildlife Service, 71 p., accessed February 25, 2014, at http://www.fws.gov/southeast/5yearReviews/5yearreviews/interiorLeastTern5yrReivew102413.pdf.

U.S. Geological Survey Gap Analysis Program, 2011, National GAP vertebrate species distribution model: U.S. Geological Survey database, accessed October 5, 2015, at http://gapanalysis.usgs.gov/species.

Ward, M.P., Semel, B., Jablonski, C., Deutsch, C., Giammaria, V., Miller, S.B., and McGuire, B.M., 2011, Consequences of using conspecific attraction in avian conservation—A case study of endangered colonial waterbirds: Waterbirds, v. 34, no. 4, p. 476–480.

Whittier, J.B., Leslie, D.M., Jr., and Van den Bussche, R.A., 2006, Genetic variation among subspecies of least tern (*Sterna antillarum*)—Implications for conservation: Waterbirds, v. 29, p. 176–184.

Winton, B.R., and Leslie, D.M., Jr., 2003, Nest sites and conservation of endangered interior least terns *Sterna antillarum athalassos* on an alkaline flat in the south-central Great Plains (USA): Acta Ornithologica, v. 38, p. 135–141.

Wood, J.A., Carter, S.K., Litshert, S.E., and Carr, N.B., 2017, Including broad-scale indicators in multiscale natural resource monitoring and assessment programs in the Bureau of Land Management, chap. 3 *of* Carter, S.K., Carr, N.B., Miller, K.H., and Wood, D.J.A., eds., Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, p. 15–37, accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Zeigenfuss, L.C., 2015, Interior least tern, chap. 14 *of* Assal, T.J., Melcher, C.P., and Carr, N.B., eds., 2015, Southern Great Plains Rapid Ecoregional Assessment—Pre-assessment report: U.S. Geological Survey Open-File Report 2015–1003, p. 118–126, accessed July 13, 2017, at https://doi.org/10.3133/ofr20151003.

Chapter 10. Burrowing Owl

Introduction

The burrowing owl (*Athene cunicularia*) is a small, ground-dwelling, diurnal owl. The burrowing owl occurs throughout the western portions of the SGP, but historically it occurred throughout the SGP (Klute and others, 2003). Northern populations are migratory, but in New Mexico and Texas the birds may remain year round (Poulin and others, 2011). Its winter range is largely south of the SGP in Texas, Louisiana, Mexico, and Central America. Burrowing owl populations appear to be declining at the margins of its breeding range in North America (Macias-Duarte and Conway, 2016), presumably as a result of declines in prairie dogs and conversion to agriculture, and the owls have been extirpated from much of their original range in Canada and eastern portions of the Central Plains (Poulin and others, 2011). The U.S. Fish and Wildlife Service considers the burrowing owl a species of conservation concern (Klute and others, 2003).

The burrowing owl nests in well-drained grasslands and prairies, shrub-steppe, and deserts, but it may also use agricultural lands and open sites created by human activities (Poulin and others, 2011). Year-round habitat is characterized by very short vegetation with a significant component of bare ground and the presence of elevated perch sites, such as mounds, shrubs, or fence posts, but otherwise lacking trees (Panella, 2013; Thiele and others, 2013). Burrowing owls nest and roost in underground burrows excavated by other species, especially prairie dogs (Desmond and others, 2000; Poulin and others, 2011; Augustine and Baker, 2013; Alverson and Dinsmore, 2014; Ray and others, 2016). Their prey includes a wide variety of invertebrates, rodents, birds, and small herptiles (Conrey, 2010; Poulin and others, 2011). Burrowing owls exhibit unusual behaviors that may enhance access to prey during incubation and brooding; for example, they cover the entrances to their burrows with animal dung, which attracts dung beetles and other insects that the owls consume (Levy and others, 2004). They also have been reported caching food in their burrows (Poulin and others, 2011). Grazing by prairie dogs and fire can help to maintain open, short vegetation preferred by burrowing owls (Milchunas and others, 1998; Klute and others, 2003; Poulin and others, 2011).

Development can destroy or degrade burrowing owl habitat (Orth and Kennedy, 2001; Klute and others, 2003; Poulin and others, 2011; Panella, 2013). Burrowing owls avoid roads with vehicle speeds greater than 80 kilometers per hour (Scobie and others, 2014). The species is vulnerable to mortality from vehicles as a consequence of their low-flying habits and the use of roadsides and adjacent fence posts for foraging and perching (Poulin and others, 2011; Panella, 2013). Additionally, burrowing owls are killed by collisions with wind turbines (Smallwood and Thelander, 2008). Disturbance from human activities can lead to reduced nesting productivity (Poulin and others, 2011). Mammalian predators of burrowing owls use roads and other infrastructure associated with energy development as travel corridors, and their avian predators use powerlines as perches (Klute and others, 2003; Poulin and others, 2011; Panella, 2013). Agricultural activities, including cultivation, prairie dog control efforts, and pesticide use, can all negatively affect burrowing owl populations (Klute and others, 2003; Poulin and others, 2011; Panella, 2013; Justice-Allen and Loyd, 2017). Although owls typically breed on native grasslands, they may also use irrigated farmland, presumably because of greater prey densities associated with these lands (Poulin and others, 2011). In some cases, however, agricultural lands may represent habitat sinks (Conway and others, 2006; Berardelli and others, 2010). The value of agricultural lands may depend on the size and extent of the cultivated lands relative to native grasslands (Restani and others, 2008). Pesticides can cause mortality directly through poisoning and indirectly by reducing prey populations (Poulin and others, 2011; Justice-Allen and Loyd, 2017). In some cases, plague epizootics that lead to prairie dog die-offs can negatively affect burrowing owls (Klute and others, 2003; Conrey, 2010; Alverson and Dinsmore, 2014). Additional background information on burrowing owls can be found in the SGP pre-assessment report (Melcher, 2015).

Rapid Ecoregional Assessment Components Evaluated for the Burrowing Owl

The key ecological attributes and change agents addressed by core management questions for burrowing owl habitat include amount and distribution, landscape structure, and development (tables 10–1 and 10–2). Fire occurrence was evaluated for the entire SGP (see Reese and others, 2017, chap. 3). Invasive woody species and climate change were evaluated for grassland communities (see Reese and others, 2017, chap. 4). Overall landscape-level ranking variables are summarized in table 10–3. The core and integrated management questions are listed in table 10–4.

Burrowing owl. Photograph by Ron Knight (Creative Commons Attribution 2.0 Generic).

Table 10–1. Key ecological attributes and associated indicators used to address core management questions for burrowing owls for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Amount and distribution	Total area	Baseline habitat (breeding)[2]
Landscape structure	Patch size	Patch sizes for baseline habitat
Landscape dynamics	Fire occurrence	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]MaxEnt was used to predict baseline habitat. See chapter 2, "Methods Overview for Species," for methods and datasets used.

Table 10–2. Anthropogenic change agents and associated indicators used to address core management questions for burrowing owls for the Southern Great Plains Rapid Ecoregional Assessment.

[km, kilometer; mi, mile]

Attributes	Variables	Indicators[1]
Development	Terrestrial development index (TDI)	Percentage of baseline habitat in seven development classes based on a 2.5-km (1.55-mi) moving window
	Index of fragmentation	Patch sizes for relatively undeveloped[2] habitat
Invasive species	Potential for woody species expansion	See Reese and others (2017, chap. 4)
Climate change	Potential distribution of grasslands	See Reese and others (2017, chap. 4)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]Terrestrial development index score less than or equal to 2 percent.

Table 10–3. Landscape-level variables used to address the integrated management question for burrowing owls. Ranks for landscape-level habitat area and development were combined into an overall landscape-level rank for the Southern Great Plains Rapid Ecoregional Assessment.

[>, greater than; km, kilometer; mi, mile]

Landscape-level variables[1]	Description	Relative rank[2]		
		Lowest	Medium	Highest
Area	Percentage of baseline habitat within a 5-km-radius (3.11-mi) moving window	>0–41.0	>41.0–70.0	>70.0
Development	Mean terrestrial development index (TDI) score for baseline habitat within a 5km-radius (3.11-mi) moving window	0–2	>2–10	>10

[1]See Reese and others (2017, chap. 2) for methods and datasets used to address integrated management questions.

[2]Ranking breakpoints for area of baseline habitat were determined from equal subsets of the data. Ranking breakpoints for terrestrial development index scores were standardized for all terrestrial conservation elements.

Table 10–4. Management questions addressed for burrowing owls for the Southern Great Plains Rapid Ecoregional Assessment.

Core management questions[1]	Results
What is the distribution of baseline habitat for burrowing owls?	Figure 10−1
Where does existing development pose the greatest threat to burrowing owl habitat, and where are the large, relatively undeveloped areas?	Figures 10−2 and 10−3
How has development fragmented burrowing owl habitat?	Figures 10−4 and 10−5
Integrated management question[2]	**Results**
Where is burrowing owl habitat with the highest overall landscape-level rank?	Figure 10−6

[1]See Reese and others (2017, chap. 11) for management questions that could not be addressed.

[2]See table 10−3.

Management Questions and Results

What is the distribution of baseline habitat for burrowing owls (fig. 10–1)?

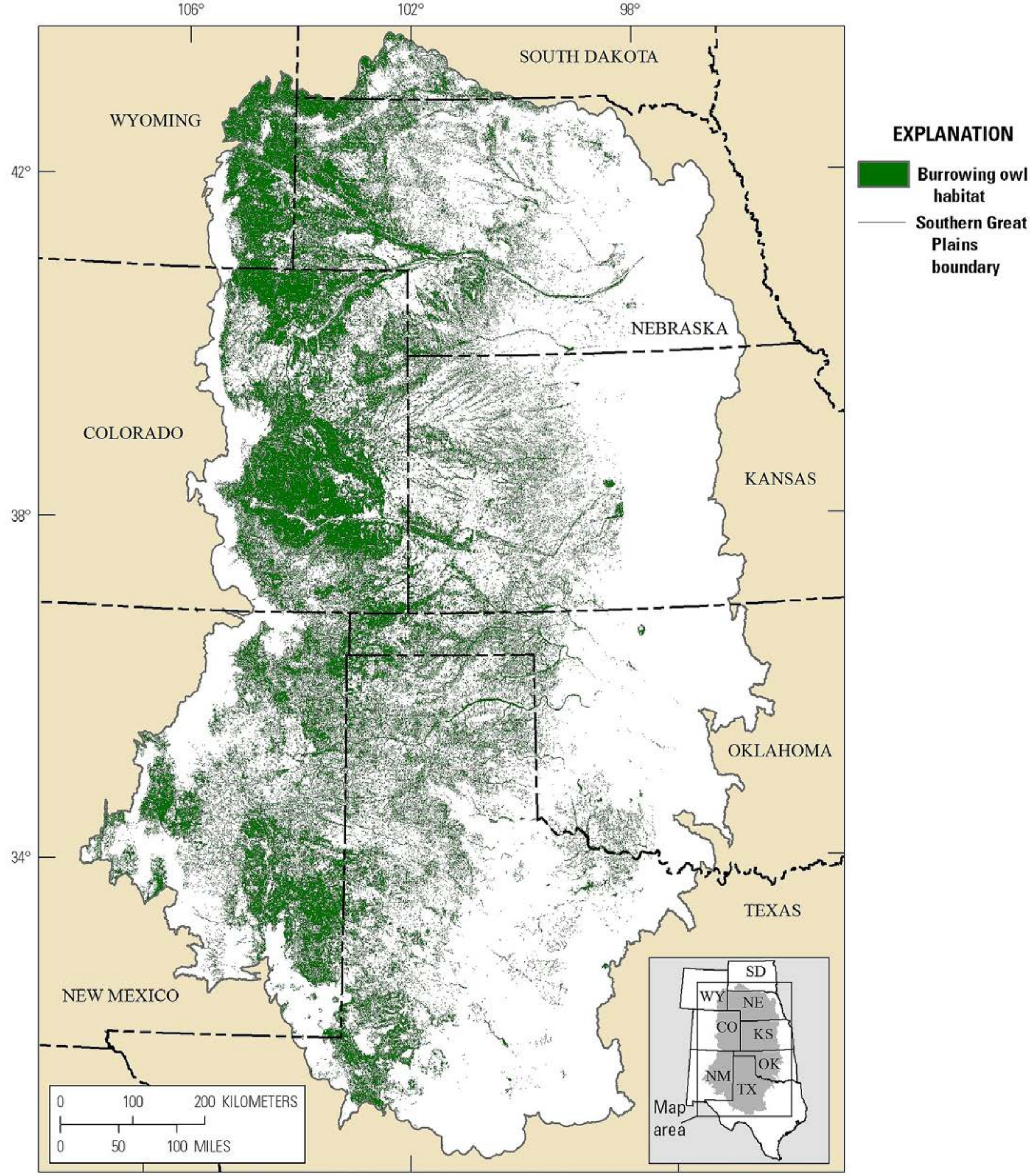


Figure 10–1. Predicted distribution of baseline habitat for burrowing owls in the Southern Great Plains.

Where does existing development pose the greatest threat to burrowing owl habitat, and where are the large, relatively undeveloped areas (figs. 10–2 and 10–3)?

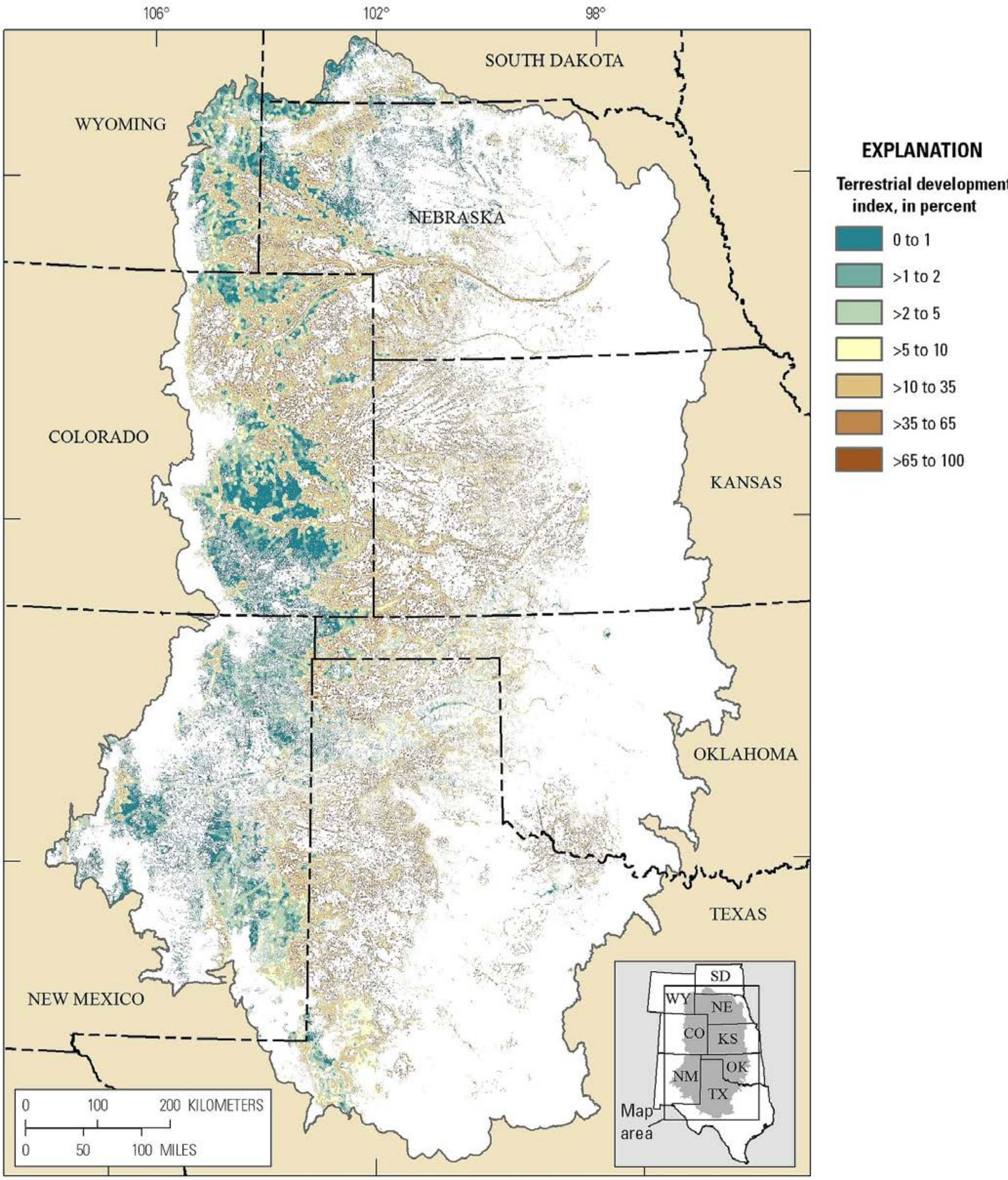


Figure 10–2. Terrestrial development index for burrowing owl baseline habitat in the Southern Great Plains.

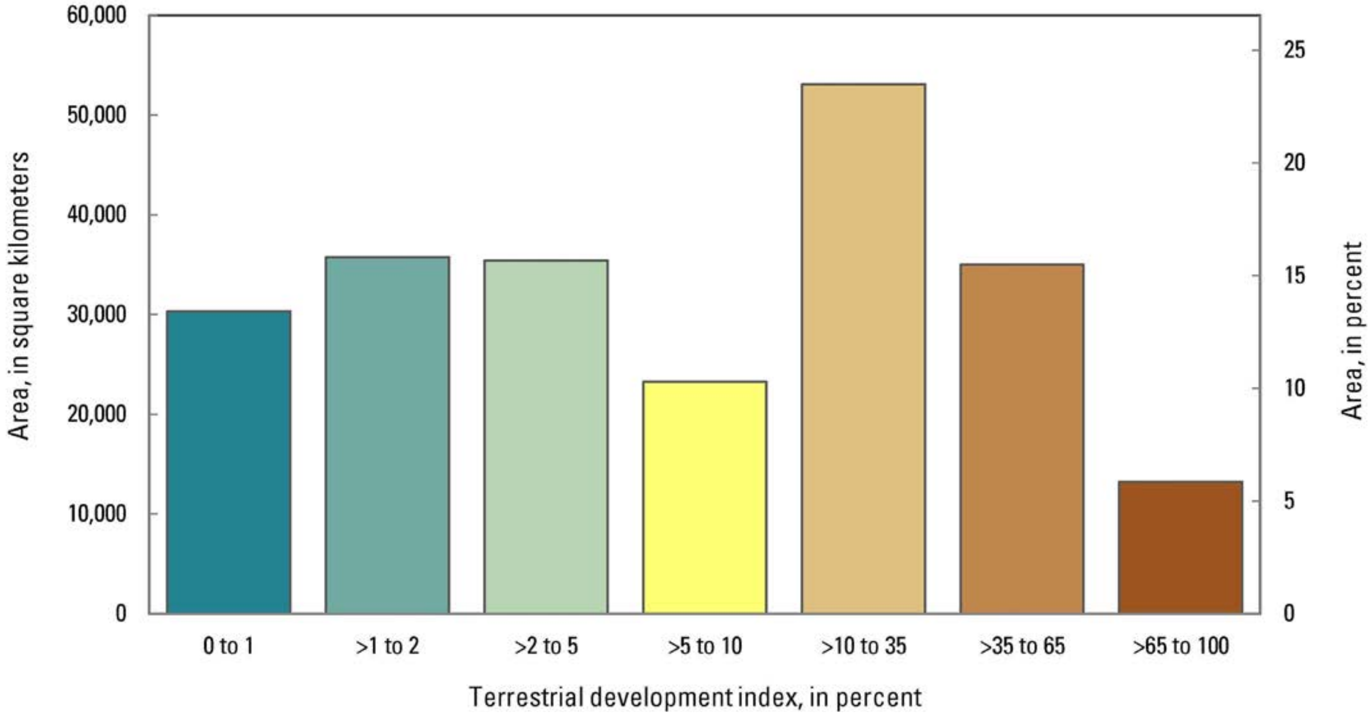


Figure 10–3. Area of burrowing owl baseline habitat by terrestrial development index class in the Southern Great Plains.

How has development fragmented burrowing owl habitat (figs. 10–4 and 10–5)?

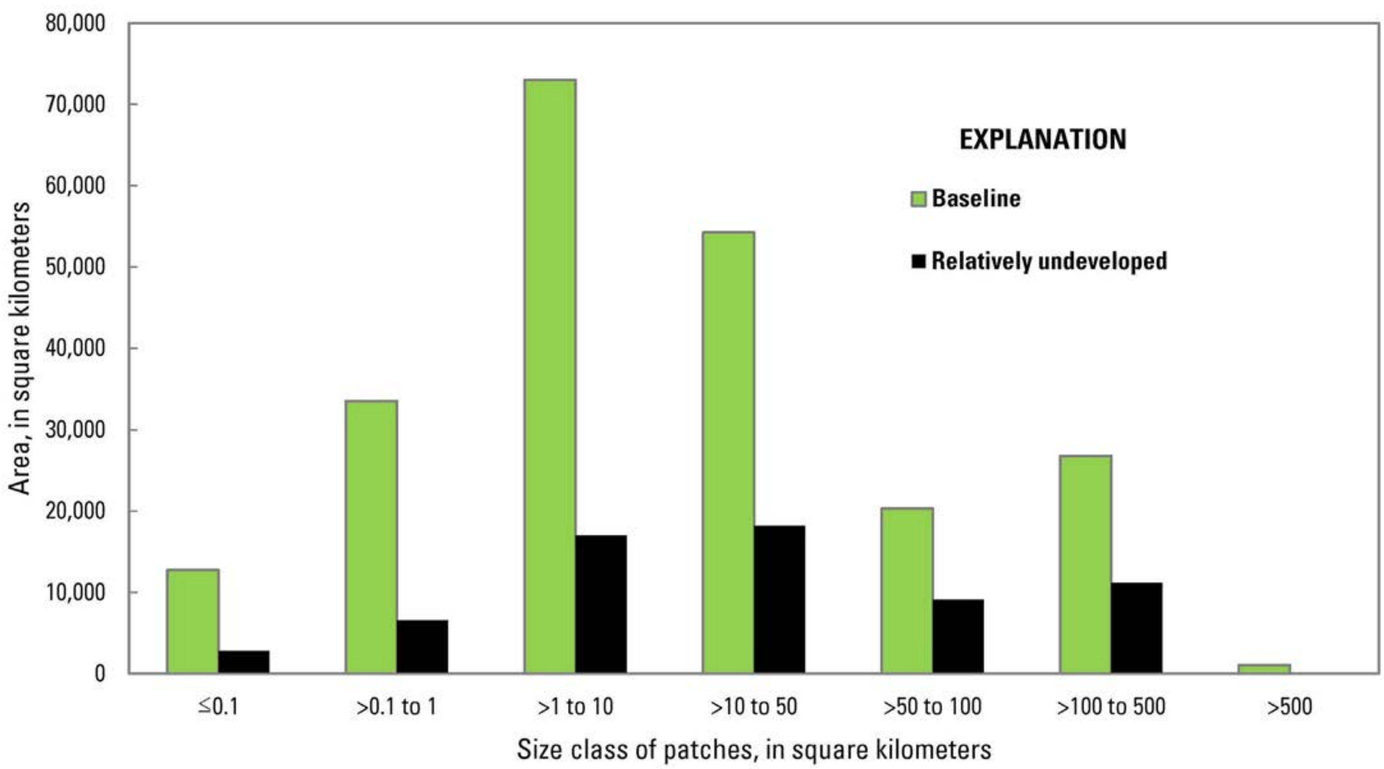


Figure 10–4. Area of burrowing owl habitat in the Southern Great Plains by patch size class for baseline and relatively undeveloped conditions (terrestrial development index score ≤2 percent).

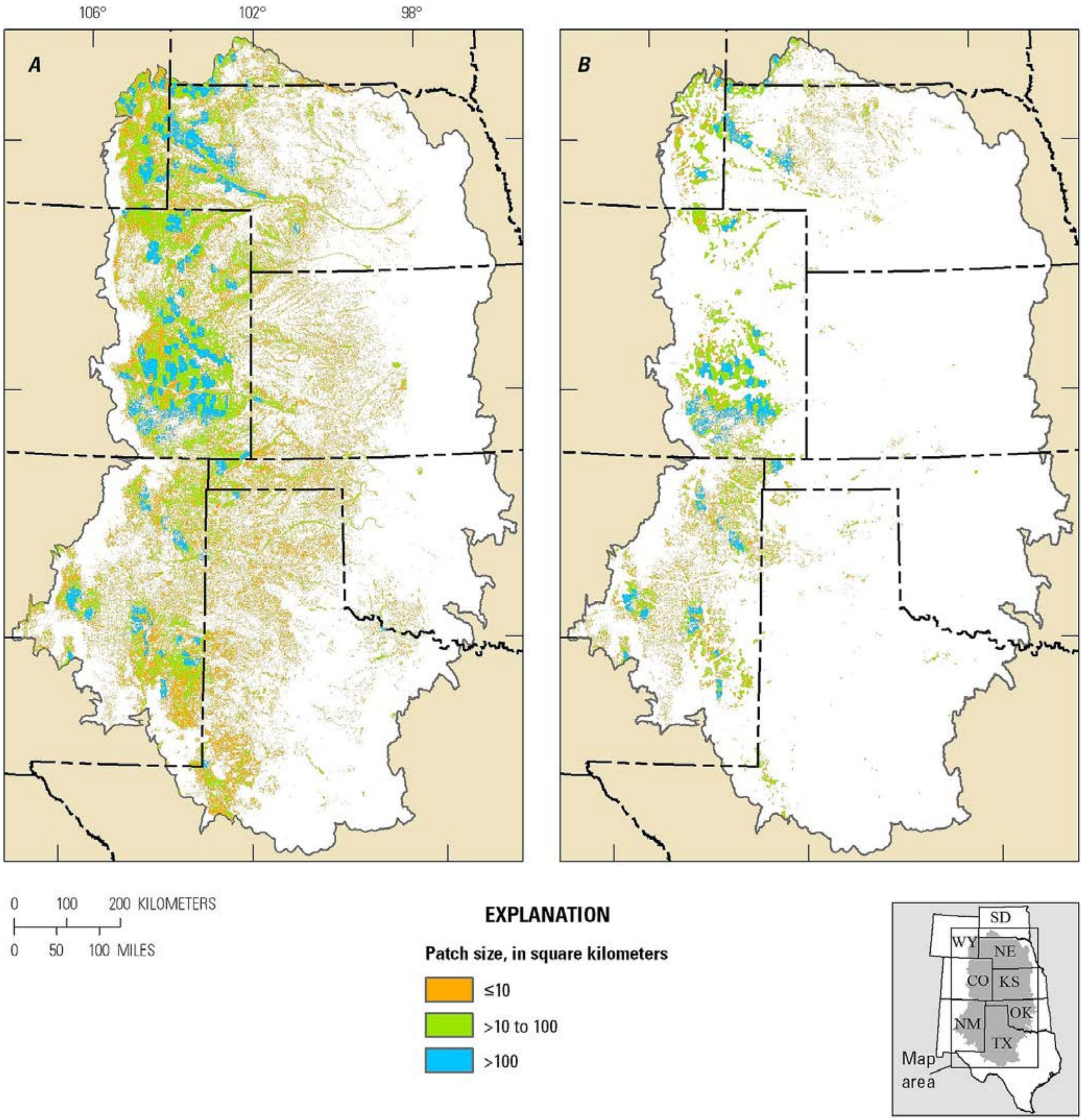


Figure 10–5. Patch size of burrowing owl habitat in the Southern Great Plains. *A,* Baseline habitat. *B,* Relatively undeveloped habitat (terrestrial development index score ≤2 percent).

Where is burrowing owl habitat with the highest overall landscape-level rank (fig. 10–6)?

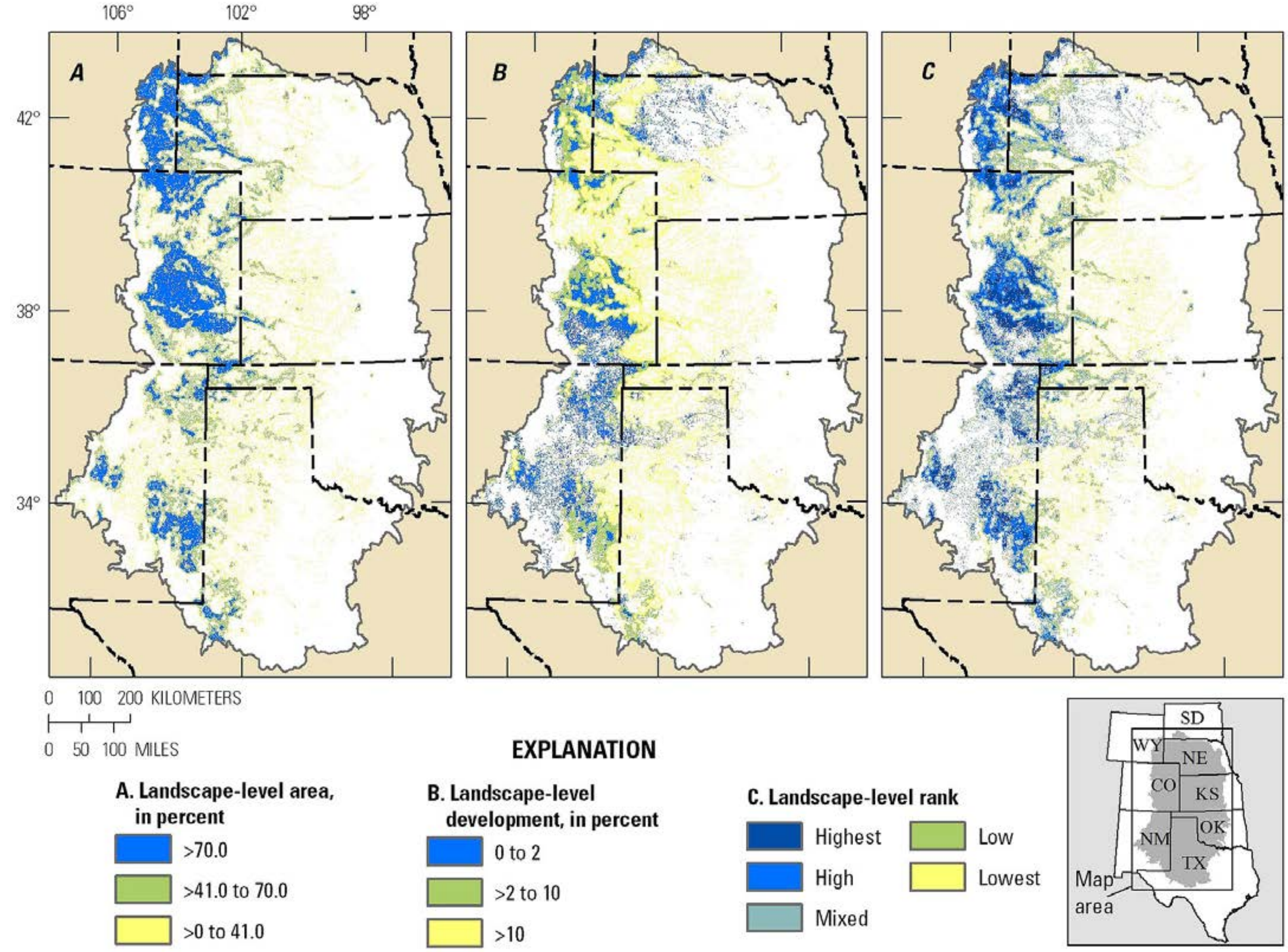


Figure 10–6. Landscape-level summaries for burrowing owl habitat in the Southern Great Plains. Overall landscape-level rank (*C*) is derived from (*A*) landscape-level habitat area and (*B*) landscape-level development, summarized by a 5-kilometer-radius (3.11-mile) moving window (see table 10–3). Highest overall landscape-level rank corresponds to the largest landscape-level area and the lowest landscape-level development. Lowest overall landscape-level rank corresponds to the smallest landscape-level area and highest landscape-level development. Landscape-level ranks are not intended as standalone summaries and are best interpreted in conjunction with the geospatial datasets used to address core management questions.

Summary

- Baseline habitat for burrowing owls occurs throughout much of the SGP except in the easternmost portions of the region (fig. 10–1). There are approximately 222,000 km^2 (85,715 mi^2) of baseline habitat in the SGP.
- Burrowing owl habitat with the lowest development levels is concentrated in southeastern Colorado, eastern New Mexico, and the northern portions of the SGP (fig. 10–2). Approximately 29 percent of their habitat is relatively undeveloped (TDI score ≤2 percent), and 16 percent has low development (TDI scores 2–5 percent) (fig. 10–3). However, 21 percent of the habitat has very high development (TDI scores >35 percent).
- Burrowing owl habitat in the central and eastern portions of the SGP (in Texas, Oklahoma, Kansas, and southern Nebraska) is highly fragmented, primarily by agriculture (figs. 10–4 and 10–5*A*). Approximately 78 percent of baseline habitat occurs in patches smaller than 50 km^2 (19.3 mi^2) (fig. 10–4). Most of the larger, relatively undeveloped patches of habitat are in southeastern Colorado (fig. 10–5*B*).
- The largest, most intact areas (the highest overall landscape-level rank) are in the western extent of the shortgrass prairie (fig. 10–6*C*), where there has been less conversion to croplands than in other areas of the region (Reese and others, 2017).
- The broad-scale summaries provided by the REA are intended to be used in conjunction with local-level information on habitat conditions (Wood and others, 2017).

References Cited

Alverson, K.M., and Dinsmore, S.J., 2014, Factors affecting burrowing owl occupancy of prairie dog colonies: The Condor, v. 116, p. 242–250.

Augustine, D.J., and Baker, B.W., 2013, Associations of grassland bird communities with black-tailed prairie dogs in the North American Great Plains: Conservation Biology, v. 27, p. 324–334.

Berardelli, D., Desmond, M.J., and Murray, L., 2010, Reproductive success of burrowing owls in urban and grassland habitats in southern New Mexico: Wilson Journal of Ornithology, v. 122, no. 1, p. 51–59.

Conrey, R.C.Y., 2010, Breeding success, prey use, and mark-resight estimation of burrowing owls nesting on black-tailed prairie dog towns—Plague affects a non-susceptible raptor: Fort Collins, Colo., Colorado State University, Ph.D. dissertation, 218 p.

Conway, C.J., Garcia, V., Smith, M.D., Ellis, L.A., and Whitney, J.L., 2006, Comparative demography of burrowing owls in agricultural and urban landscapes in southeastern Washington: Journal of Field Ornithology, v. 77, no. 3, p. 280–290.

Desmond, M.J., Savidge, J.A., and Eskridge, K.M., 2000, Correlations between burrowing owl and black-tailed prairie dog declines—A 7-year analysis: Journal of Wildlife Management, v. 64, no. 4, p. 1067–1075.

Justice-Allen, A., and Loyd, K.A., 2017, Mortality of western burrowing owls (*Athene cunicularia hypugaea*) associated with brodifacoum exposure: Journal of Wildlife Diseases, v. 53, p. 165–169.

Klute, D.S., Ayers, L.W., Green, M.T., Howe, W.H., Jones, S.L., Shaffer, J.A., Sheffield, S.R., and Zimmerman, T.S., 2003, Status assessment and conservation plan for the western burrowing owl in the United States: U.S. Fish and Wildlife Service Biological Technical Publication BTP–R6001–2003, 108 p.

Levy, D.J., Duncan, R.S., and Levins, C.F., 2004, Use of dung as a tool by burrowing owls: Nature, v. 431, p. 39.

Macias-Duarte, A., and Conway, C.J., 2016, Distributional changes in the western burrowing owl (*Athene cunicularia hypugaea*) in North America from 1967 to 2008: Journal of Raptor Research, v. 49, p. 75–83.

Melcher, C.P., 2015, Burrowing owl, chap. 12 *of* Assal, T.J., Melcher, C.P., and Carr, N.B., eds., 2015, Southern Great Plains Rapid Ecoregional Assessment—Pre-assessment report: U.S. Geological Survey Open-File Report 2015–1003, p. 118–126, accessed July 13, 2017, at https://doi.org/10.3133/ofr20151003.

Milchunas, D.G., Lauenroth, W.K., and Burke, I.C., 1998, Livestock grazing—Animal and plant biodiversity of shortgrass steppe and the relationship to ecosystem function: Oikos, v. 83, no. 1, p. 65−74.

Orth, P.B., and Kennedy, P.L., 2001, Do land-use patterns influence nest-site selection by burrowing owls (*Athene cunicularia hypugaea*) in northeastern Colorado?: Canadian Journal of Zoology, v. 79, p. 1038–1045.

Panella, M.J., 2013, Western burrowing owl (*Athene cunicularia hypugaea*)—A species conservation assessment for the Nebraska Natural Legacy Project: Lincoln, Nebr., Nebraska Game and Parks Commission, Wildlife Division, 23 p.

Poulin, R., Todd, L.D., Haug, E.A., Millsap, B.A., and Martell, M.S., 2011, Burrowing owl (*Athene cunicularia*), *in* Poole, A., ed., The birds of North America: Ithaca, N.Y., Cornell Lab of Ornithology, no. 61, 20 p., accessed January 2018, at https://birdsna.org/Species-Account/bna/species/burowl.

Ray, J.D., McIntyre, N.E., Wallace, M.C., Teaschner, A.P., and Schoenhals, M.G., 2016, Factors influencing burrowing owl abundance in prairie dog colonies on the southern High Plains of Texas: Journal of Raptor Research, v. 50, p. 186–193.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100.

Restani, M., Davies, J.M., and Newton, W.E., 2008, Importance of agricultural landscapes to nesting burrowing owls in the Northern Great Plains, USA: Landscape Ecology, v. 23, p. 977–987.

Scobie, C., Bayne, E., and Wellicome, T., 2014, Influence of anthropogenic features and traffic disturbance on burrowing owl diurnal roosting behavior: Endangered Species Research, v. 24, p. 73–83.

Smallwood, K.S., and Thelander, C., 2008, Bird mortality in the Altamont Pass Wind Resource Area, California: Journal of Wildlife Management, v. 72, no. 1, p. 215–223.

Thiele, J.P., Bakker, K.K., and Dieter, C.D., 2013, Multiscale nest site selection by burrowing owls in western South Dakota: Wilson Journal of Ornithology, v. 125, p. 763–774.

Wood, J.A., Carter, S.K., Litshert, S.E., and Carr, N.B., 2017, Including broad-scale indicators in multiscale natural resource monitoring and assessment programs in the Bureau of Land Management, chap. 3 *of* Carter, S.K., Carr, N.B., Miller, K.H., and Wood, D.J.A., eds., Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, p. 15–37, accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Chapter 11. Black-Tailed Prairie Dog

Introduction

The black-tailed prairie dog (*Cynomys ludovicianus*) is a colonial, burrowing rodent of open grasslands of the Great Plains. Once abundant and widespread, black-tailed prairie dogs have been extirpated from a large proportion of their estimated historical range, including much of the SGP (Proctor and others, 2006; U.S. Fish and Wildlife Service, 2009), and many remaining populations occur in isolated colonies and complexes at low densities (U.S. Fish and Wildlife Service, 2009). Although the black-tailed prairie dog has been petitioned repeatedly for listing under the Endangered Species Act, the species has not been designated as threatened because population size may have been underestimated (U.S. Fish and Wildlife Service, 2004, 2009). Others have challenged this conclusion, and the need for protective status continues to be debated (Miller and Reading, 2012; Sidle and others, 2012; Rauscher and others, 2013).

Habitat for black-tailed prairie dogs is characterized by short vegetation, and prairie dogs will often clip tall vegetation, presumably to facilitate predator detection (Hoogland, 1995). Black-tailed prairie dogs are highly social and breed in colonies. The primary natural sources of mortality for prairie dogs are predation, infanticide, drought, and winter mortality (Hoogland, 1995). The most notable predator is the black-footed ferret (*Mustela nigripes*), an endangered carnivore that specializes on prairie dogs (Miller and Reading, 2012).

The dynamics of prairie dog colonies are influenced by precipitation, grazing, sylvatic plague, and fire, as well as interactive effects of these factors (Augustine and others, 2007; Lauenroth and Burke, 2008; Grassel and others, 2016; Eads and Biggins, 2017). In turn, prairie dogs create large patches of altered vegetation similar in many respects to those created by native ungulates (Whicker and Detling, 1988). In some areas, the black-tailed prairie dog can function as a keystone species by providing habitat conditions, such as burrows and low vegetation cover, required or used by other species (Kotliar and others, 2006). Indeed, four species of management concern in the SGP associate closely with prairie dogs: the ferruginous hawk (see chap. 4), burrowing owl (see chap. 10), mountain plover (see chap. 7), and black-footed ferret (Kotliar and others, 2006). The keystone role of prairie dogs depends, in part, on colony size and dynamics (Kotliar, 2000). The degree of isolation of colonies may affect colony dynamics, but the relationships between the probability of colony extinction and intercolony distance are unclear (Lomolino and Smith, 2001; Stapp and others, 2004; Johnson and others, 2011). Colony persistence can be greatest among larger colonies, presumably because large colonies provide better protection from predators and stochastic events than small colonies and because they may be encountered more readily by dispersing prairie dogs (Hoogland, 1995; Lomolino and Smith, 2001; Snäll and others, 2008).

The primary threats to prairie dogs include sylvatic plague, habitat conversion, and ongoing prairie dog control. Epizootic outbreaks of plague occur about every 5–14 years and kill most, if not all, prairie dogs in affected areas (Cully and others, 2006), but interceding, enzootic periods of the plague cycle also cause chronic mortality in their populations (Biggins and others, 2010; Eads and Biggins, 2017). Conversion of grasslands to croplands and urbanization have also led to habitat loss and fragmentation (Johnson and Collinge, 2004; U.S. Fish and Wildlife Service, 2009; Magle and others, 2010; Beals and others, 2015). In some areas, prairie dogs are actively killed by poisoning, and millions are shot every year (Reeve and Vosburgh, 2006).

Other human activities may negatively affect prairie dogs. Fire suppression can enhance shrub expansion on grasslands, which can impede colony expansion and interfere with predator detection by prairie dogs (Van Auken, 2000; Long and others, 2006). Energy development could reduce the amount of habitat available to prairie dogs or fragment their colonies (U.S. Fish and Wildlife Service, 2009), although some disturbance activities may create potential habitat. Rotating wind turbines may negatively affect food consumption by prairie dogs by increasing the time spent engaged in antipredator behaviors (Johnson and others, 2000; Rabin and others, 2006; Biggins and others, 2012). Additional background information on black-tailed prairie dogs can be found in the SGP pre-assessment report (Eads, 2015).

Rapid Ecoregional Assessment Components Evaluated for the Black-Tailed Prairie Dog

The key ecological attributes and change agents addressed by core management questions for the black-tailed prairie dog habitat include amount and distribution, landscape structure, and development (tables 11–1 and 11–2). Fire occurrence was evaluated for the entire SGP (see Reese and others, 2017, chap. 3). Invasive woody species and climate changes were evaluated for grassland communities (see Reese and others, 2017, chap. 4). Overall landscape-level ranking variables are summarized in table 11–3. The core and integrated management questions are listed in table 11–4.

Black-tailed prairie dog. Photograph by Larry Smith (Creative Commons Attribution 2.0 Generic).

Table 11–1. Key ecological attributes and associated indicators used to address core management questions for black-tailed prairie dogs for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Amount and distribution	Total area	Baseline habitat (year round)[2]
Landscape structure	Patch size	Patch sizes for baseline habitat
Landscape dynamics	Fire occurrence	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]MaxEnt was used to predict baseline habitat. See chapter 2, "Methods Overview for Species," for methods and datasets used.

Table 11–2. Anthropogenic change agents and associated indicators used to address core management questions for black-tailed prairie dogs for the Southern Great Plains Rapid Ecoregional Assessment.

[km, kilometer; mi, mile]

Attributes	Variables	Indicators[1]
Development	Terrestrial development index (TDI)	Percentage of baseline habitat in seven development classes based on a 2.5-km (1.55-mi) moving window
	Index of fragmentation	Patch sizes for relatively undeveloped[2] habitat
Invasive species	Potential for woody species expansion	See Reese and others (2017, chap. 4)
Climate change	Potential distribution of grasslands	See Reese and others (2017, chap. 4)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]Terrestrial development index score less than or equal to 2 percent.

Table 11–3. Landscape-level variables used to address the integrated management question for black-tailed prairie dogs. Ranks for landscape-level habitat area and development were combined into an overall landscape-level rank for the Southern Great Plains Rapid Ecoregional Assessment.

[>, greater than; km, kilometer; mi, mile]

Landscape-level variables[1]	Description	Relative rank[2]		
		Lowest	Medium	Highest
Area	Percentage of baseline habitat within a 5-km-radius (3.11-mi) moving window	>0–45.0	>45.0–73.7	>73.7
Development	Mean terrestrial development index (TDI) score for baseline habitat within a 5km-radius (3.11-mi) moving window	0–2	>2–10	>10

[1]See Reese and others (2017, chap. 2) for methods and datasets used to address integrated management questions.

[2]Ranking breakpoints for area of baseline habitat were determined from equal subsets of the data. Ranking breakpoints for terrestrial development index scores were standardized for all terrestrial conservation elements.

Table 11–4. Management questions addressed for black-tailed prairie dogs for the Southern Great Plains Rapid Ecoregional Assessment.

Core management questions[1]	Results
What is the distribution of baseline habitat for black-tailed prairie dogs?	Figure 11−1
Where does existing development pose the greatest threat to black-tailed prairie dog habitat, and where are the large, relatively undeveloped areas?	Figures 11−2 and 11−3
How has development fragmented black-tailed prairie dog habitat?	Figures 11−4 and 11−5
Integrated management question[2]	**Results**
Where is black-tailed prairie dog habitat with the highest overall landscape-level rank?	Figure 11−6

[1]See Reese and others (2017, chap. 11) for management questions that could not be addressed.

[2]See table 11−3.

Management Questions and Results

What is the distribution of baseline habitat for black-tailed prairie dogs (fig. 11–1)?

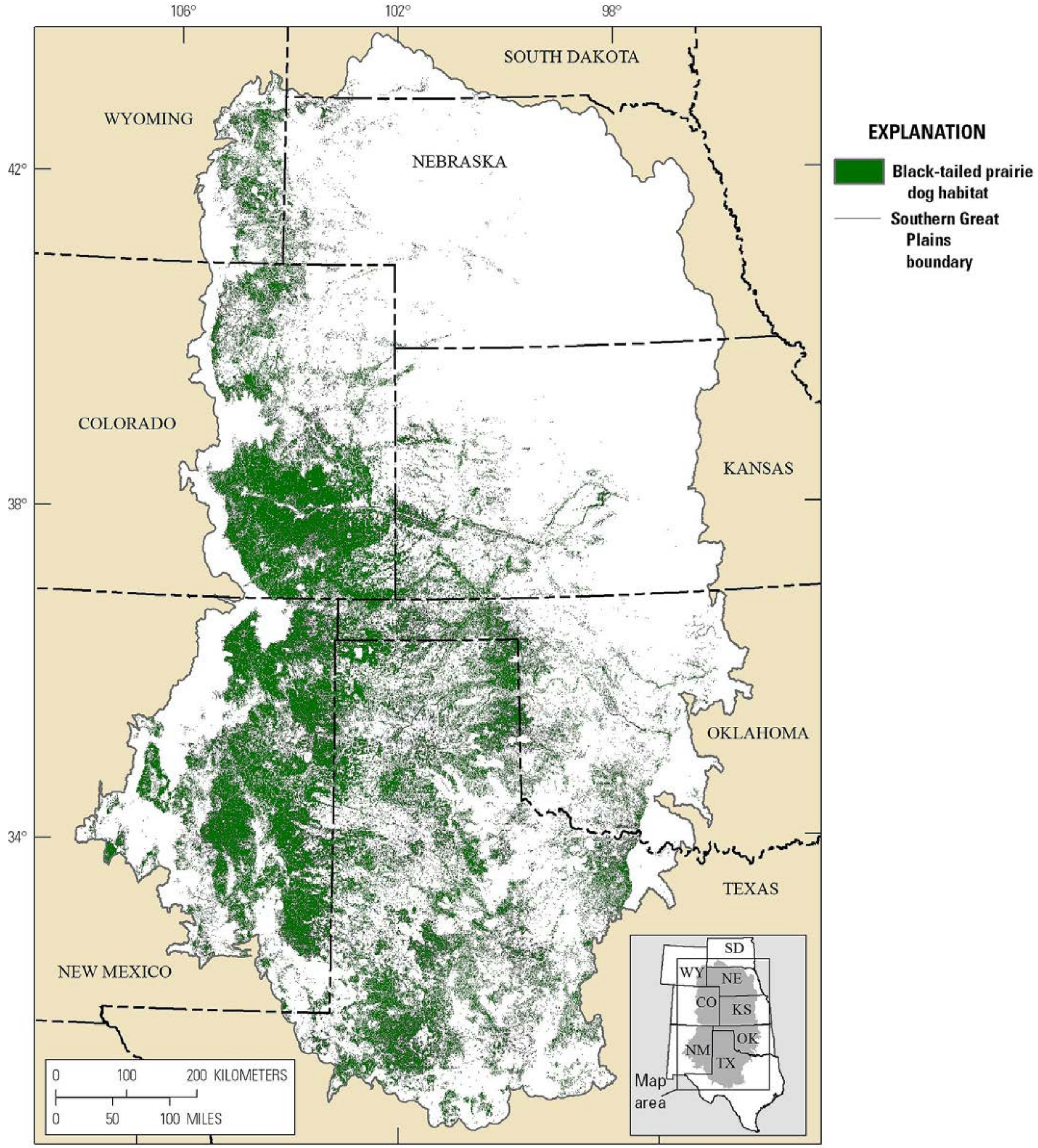


Figure 11–1. Predicted distribution of baseline habitat for black-tailed prairie dogs in the Southern Great Plains.

Where does existing development pose the greatest threat to black-tailed prairie dog habitat, and where are the large, relatively undeveloped areas (figs. 11–2 and 11–3)?

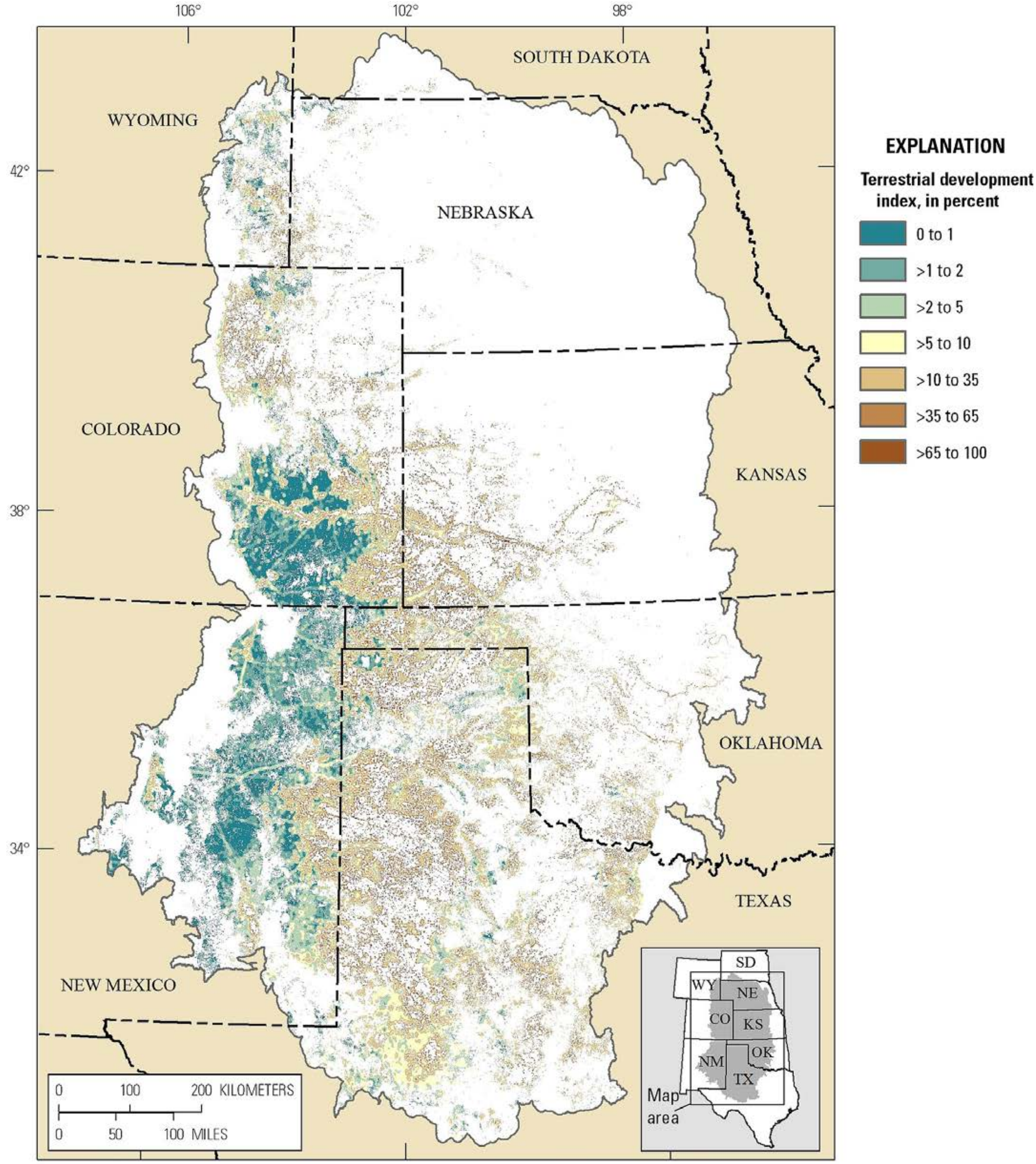


Figure 11–2. Terrestrial development index for black-tailed prairie dog baseline habitat in the Southern Great Plains.

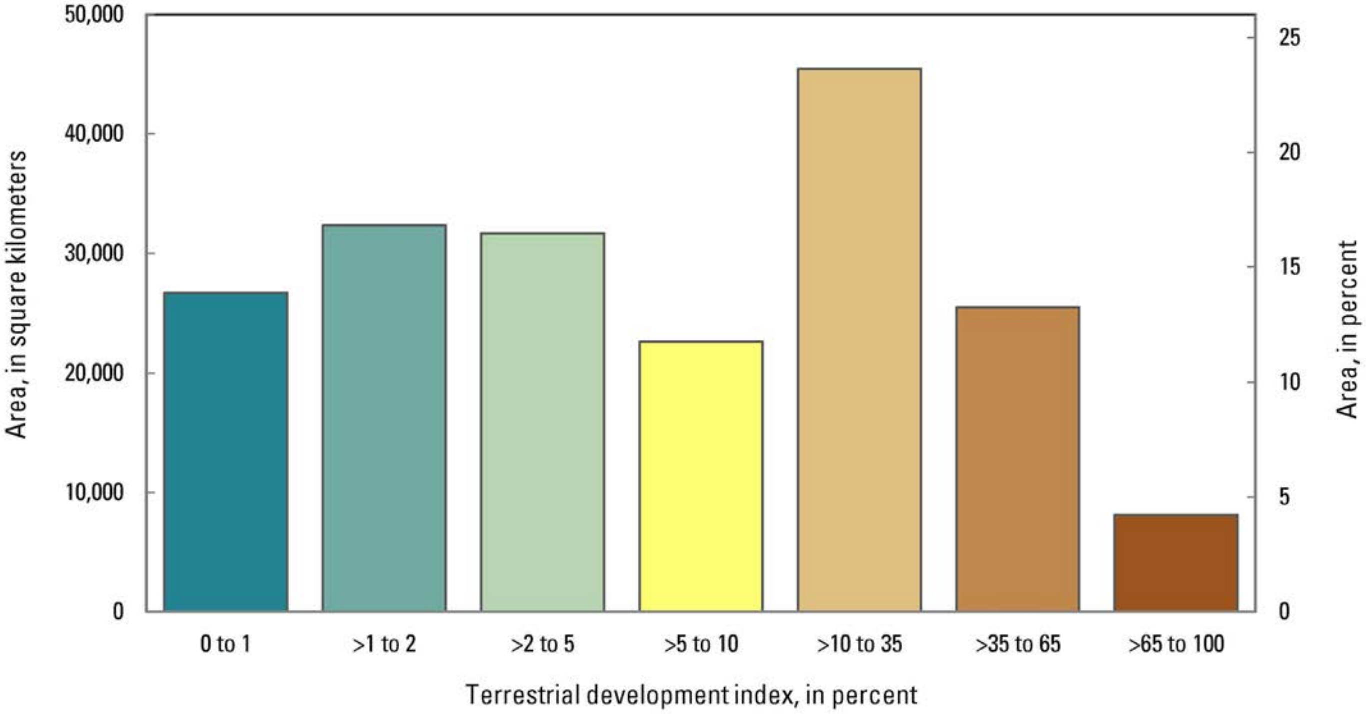


Figure 11–3. Area of black-tailed prairie dog baseline habitat by terrestrial development index class in the Southern Great Plains.

How has development fragmented black-tailed prairie dog habitat (figs. 11–4 and 11–5)?

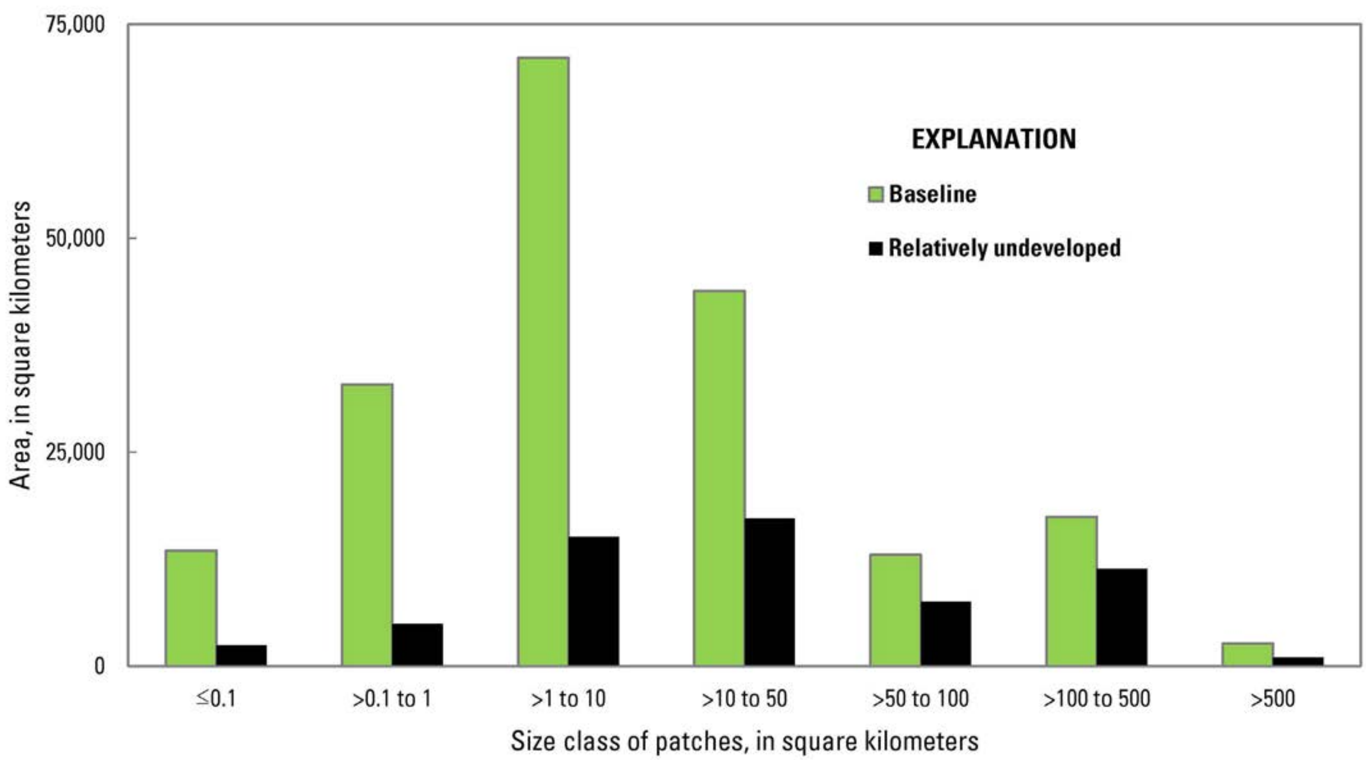


Figure 11–4. Area of black-tailed prairie dog habitat in the Southern Great Plains by patch size class for baseline and relatively undeveloped conditions (terrestrial development index score ≤2 percent).

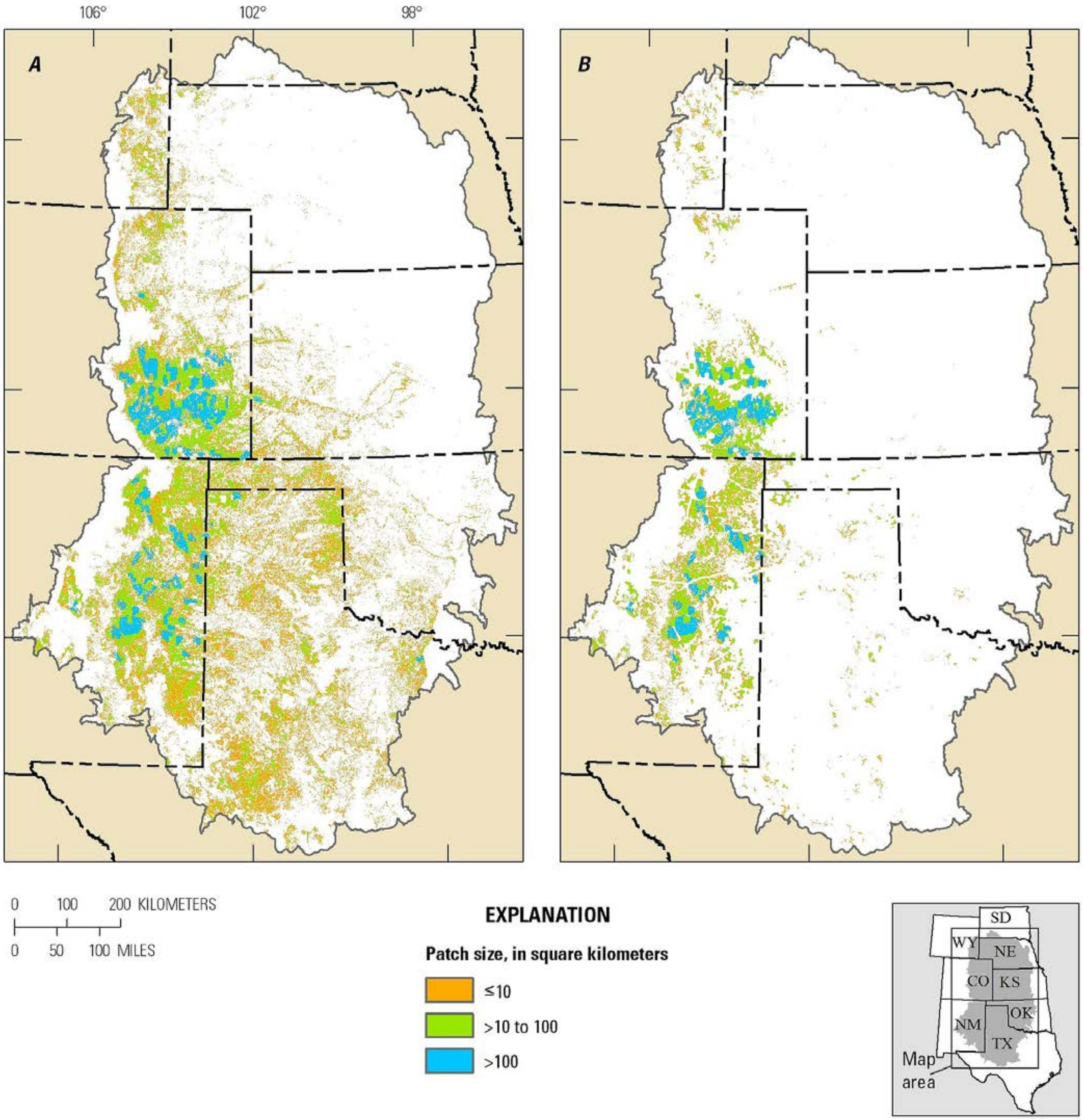


Figure 11–5. Patch size of black-tailed prairie dog habitat in the Southern Great Plains. *A,* Baseline habitat. *B,* Relatively undeveloped habitat (terrestrial development index score ≤2 percent).

Where is black-tailed prairie dog habitat with the highest overall landscape-level rank (fig. 11–6)?

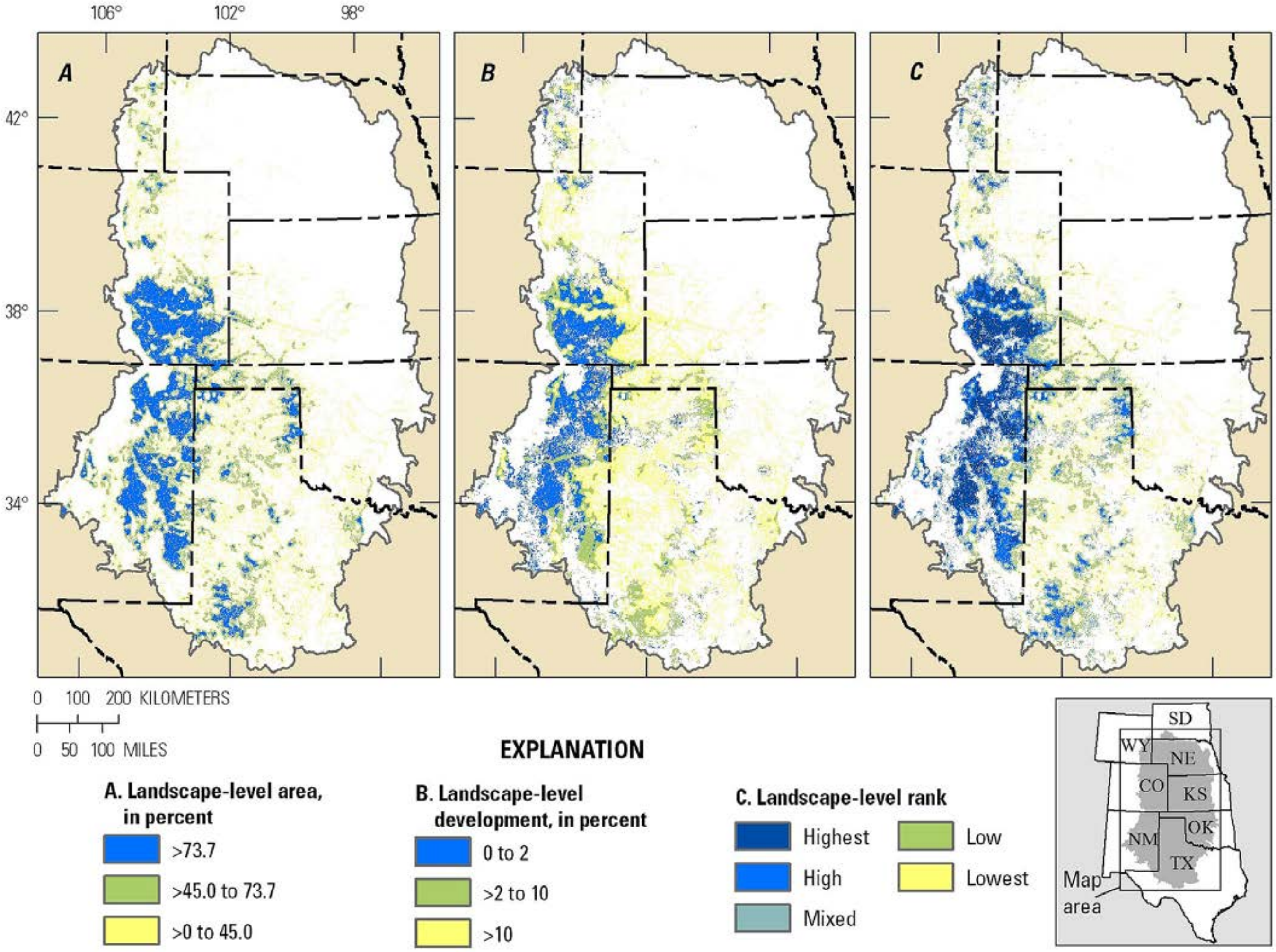


Figure 11–6. Landscape-level summaries for black-tailed prairie dog habitat in the Southern Great Plains. Overall landscape-level rank (*C*) is derived from (*A*) landscape-level habitat area and (*B*) landscape-level development, summarized by a 5-kilometer-radius (3.11-mile) moving window (see table 11–3). Highest overall landscape-level rank corresponds to the largest landscape-level area and the lowest landscape-level development. Lowest overall landscape-level rank corresponds to the smallest landscape-level area and highest landscape-level development. Landscape-level ranks are not intended as standalone summaries and are best interpreted in conjunction with the geospatial datasets used to address core management questions.

Summary

- Baseline habitat for black-tailed prairie dogs is largely restricted to the western and southern extent of the SGP (fig. 11–1). There are approximately 194,000 km^2 (74,904 mi^2) of baseline habitat in the SGP.
- Black-tailed prairie dog habitat with the lowest development is concentrated in the southeastern portions of Colorado and eastern New Mexico (fig. 11–2). Approximately 31 percent of its habitat is relatively undeveloped (TDI score ≤2 percent), and more than 16 percent has low development (TDI scores 2–5 percent) (fig. 11–3). However, 17 percent of their habitat has very high development (TDI scores >35 percent).
- Black-tailed prairie dog habitat is highly fragmented in Texas, Oklahoma, and northeastern Colorado (figs. 11–4 and 11–5*A*). Approximately 83 percent of baseline habitat occurs in patches smaller than 50 km^2 (19.3 mi^2) (fig. 11–4). The largest remaining relatively undeveloped patches are concentrated in southeastern Colorado and scattered across eastern New Mexico (fig. 11–5*B*).
- The largest, most intact areas (the highest overall landscape-level rank) are in the western extent of the shortgrass prairie (fig. 11–6*C*), where there has been less conversion to croplands than in other areas of the region (Reese and others, 2017).
- The broad-scale summaries provided by the REA are intended to be used in conjunction with local-level information on habitat conditions (Wood and others, 2017). In particular, the prevalence of plague can affect the suitability of habitat for the persistence of prairie dog colonies.

References Cited

Augustine, D.J., Cully, J.F., Jr., and Johnson, T.L., 2007, Influence of fire on black-tailed prairie dog colony expansion in shortgrass steppe: Rangeland Ecology and Management, v. 60, p. 538–542.

Beals, S.C., Preston, D.L., Wessman, C.A., and Seastedt, T.R., 2015, Resilience of a novel ecosystem after the loss of a keystone species—Plague epizootics and urban prairie dog management: Ecosphere, v. 6, no. 9, article 157, 13 p., accessed January 2018, at http://onlinelibrary.wiley.com/doi/10.1890/ES15-00244.1.

Biggins, D.E., Godbey, J.L., Gage, K.L., Carter, L.G., and Montenieri, J.A., 2010, Vector control improves survival of three species of prairie dogs (*Cynomys*) in areas considered enzootic for plague: Vector-Borne and Zoonotic Diseases, v. 10, p. 17–26.

Cully, J.F., Biggins, D.E., and Seery, D.B., 2006, Conservation of prairie dogs in areas with plague, *in* Hoogland, J.L., ed., Conservation of the black-tailed prairie dog—Saving North America's western grasslands: Washington, D.C., Island Press, p. 157–168.

Eads, D.A., 2015, Black-tailed prairie dogs, chap. 20 *of* Assal, T.J., Melcher, C.P., and Carr, N.B., eds., 2015, Southern Great Plains Rapid Ecoregional Assessment—Pre-assessment report: U.S. Geological Survey Open-File Report 2015–1003, p. 118–126, accessed July 13, 2017, at https://doi.org/10.3133/ofr20151003.

Eads, D.A., and Biggins, D.E., 2017, Paltry past-precipitation—Predisposing prairie dogs to plague?: Journal of Wildlife Management, v. 81, p. 990–998.

Grassel, S.M., Rachlow, J.L., and Williams, C.J., 2016, Reproduction by black-tailed prairie dogs and black-footed ferrets—Effects of weather and food availability: Western North American Naturalist, v. 76, p. 405–416.

Hoogland, J.L., 1995, The black-tailed prairie dog—Social life of a burrowing mammal: Chicago, Ill., University of Chicago Press, 557 p.

Johnson, G.D., Young, D.P., Jr., Erickson, W.P., Derby, C.E., Strickland, M.D., Good, R.E., and Kern, J.W., 2000, Final report—Wildlife monitoring studies for the SeaWest Windpower project, Carbon County, Wyoming—1995–1999: San Diego, Calif., SeaWest Energy Corporation and Rawlins, Wyo., Bureau of Land Management, prepared by Western EcoSystems Technology, Inc., Cheyenne, Wyo., 195 p.

Johnson, T.L., Cully, J.F., Jr., Collinge, S.K., Ray, C., Frey, C.M., and Sandercock, B.K., 2011, Spread of plague among black-tailed prairie dogs is associated with colony spatial characteristics: Journal of Wildlife Management, v. 75, no. 2, p. 357–368.

Johnson, W.C., and Collinge, S.K., 2004, Landscape effects on black-tailed prairie dog colonies: Biological Conservation, v. 115, p. 487–497.

Kotliar, N.B., 2000, Application of the new keystone-species concept to prairie dogs—How well does it work?: Conservation Biology, v. 14, p. 1715–1721.

Kotliar, N.B., Miller, B.J., Reading, R.P., and Clark, T.W., 2006, The prairie dog as a keystone species, *in* Hoogland, J.L., ed., Conservation of the black-tailed prairie dog—Saving North America's western grasslands: Washington, D.C., Island Press, p. 53–64.

Lauenroth, W.K., and Burke, I.C., 2008, Ecology of the shortgrass steppe—A long-term perspective: Oxford, United Kingdom, Oxford University Press, 536 p.

Lomolino, M.V., and Smith, G.A., 2001, Dynamic biogeography of prairie dog (*Cynomys ludovicianus*) towns near the edge of their range: Journal of Mammalogy, v. 82, p. 937–945.

Long, D., Bly-Honness, K., Truett, J.C., and Seery, D.B., 2006, Establishment of new prairie dog colonies by translocation, *in* Hoogland, J.L., ed., Conservation of the black-tailed prairie dog—Saving North America's western grasslands: Washington, D.C., Island Press, p. 188–209.

Magle, S.B., Ruell, E.W., Antolin, M.F., and Crooks, K.R., 2010, Population genetic structure of black-tailed prairie dogs, a highly interactive species, in fragmented urban habitat: Journal of Mammalogy, v. 91, p. 326–335.

Miller, B.J., and Reading, R.P., 2012, Challenges to black-footed ferret recovery—Protecting prairie dogs: Western North American Naturalist, v. 72, p. 228–240.

Proctor, J., Haskins, B., and Forrest, S.C., 2006, Focal areas for conservation of prairie dogs and the grassland ecosystem, *in* Hoogland, J.L., ed., Conservation of the black-tailed prairie dog—Saving North America's western grasslands: Washington, D.C., Island Press, p. 232–247.

Rabin, L.A., Coss, R.G., and Owings, D.H., 2006, The effects of wind turbines on antipredator behavior in California ground squirrels (*Spermophilus beecheyi*): Biological Conservation, v. 131, p. 410–420.

Rauscher, R.L., Story, S.J., Gude, J.A., and Russell, R.E., 2013, Estimation of black-tailed prairie dog colonies in Montana: Wildlife Society Bulletin, v. 37, p. 608–615.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100.

Reeve, A.F., and Vosburgh, T.C., 2006, Recreational shooting of prairie dogs, *in* Hoogland, J.L., ed., Conservation of the black-tailed prairie dog—Saving North America's western grasslands: Washington, D.C., Island Press, p. 139–156.

Sidle, J.G., Augustine, D.J., Johnson, D.H., Miller, S.D., Cully, J.F., and Reading, R.P., 2012, Aerial surveys adjusted by ground surveys to estimate area occupied by black-tailed prairie dog colonies: Wildlife Society Bulletin, v. 36, p. 248–256.

Snäll, T., O'Hara, R.B., Ray, C., and Collinge, S.K., 2008, Climate-driven spatial dynamics of plague among prairie dog colonies: American Naturalist, v. 171, no. 2, p. 238–248.

Stapp, P., Antolin, M.F., and Ball, M., 2004, Patterns of extinction in prairie dog metapopulations—Plague outbreaks follow El Nino events: Frontiers in Ecology and the Environment, v. 2, no. 5, p. 235–240.

U.S. Fish and Wildlife Service, 2004, Endangered and threatened wildlife and plants—Finding for the resubmitted petition to list the black-tailed prairie dog as threatened [Finding on a resubmitted petition]: Federal Register, v. 69, no. 159, p. 51217–51226.

U.S. Fish and Wildlife Service, 2009, Endangered and threatened wildlife and plants—12-Month finding on a petition to list the black-tailed prairie dog as threatened or endangered [Notice of a 12-month petition finding]: Federal Register, v. 74, no. 231, p. 63343–63366.

Van Auken, O.W., 2000, Shrub invasions of North American semiarid grasslands: Annual Review of Ecology and Systematics, v. 31, p. 197–215.

Whicker, A.D., and Detling, J.K., 1988, Ecological consequences of prairie dog disturbances: BioScience, v. 38, no. 11, p. 778–785.

Wood, J.A., Carter, S.K., Litshert, S.E., and Carr, N.B., 2017, Including broad-scale indicators in multiscale natural resource monitoring and assessment programs in the Bureau of Land Management, chap. 3 *of* Carter, S.K., Carr, N.B., Miller, K.H., and Wood, D.J.A., eds., Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, p. 15–37, accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Chapter 12. Tree-Roosting Bat Assemblage

Introduction

Seventeen bat species regularly occur in the SGP (see Hayes, 2015, for the full list of species occurring in the SGP). Bats roost primarily in caves, rock crevices, and trees, as well as human-created structures such as houses and bridges. Maternity roost sites are relatively warm and usually are located near reliable foraging areas (Dalquest and others, 1990; Miller, 2011), whereas winter hibernacula are relatively cold and stable (Humphrey and Kunz, 1976; Prendergast and others, 2010). Caves used for maternity and wintering roosts are not abundant in the SGP, and large segments of regional bat populations of nonmigratory, cave-dwelling species may be concentrated at a few roost sites (Prendergast and others, 2010), with the remaining individuals dispersed among talus slopes and rock crevices. Rock crevice-roosting bats, such as the pallid bat (*Antrozous pallidus*), roost in sheltered areas among along cliffs, escarpments, or boulder fields (Bogan and others, 2003). Tree-roosting bats roost individually or in small groups in trees or other vegetation (Carter and others, 2003).

Several bat species of the SGP are considered species of conservation concern: eastern red bat (*Lasiurus borealis*), hoary bat (*L. cinereus*), silver-haired bat (*Lasionycteris noctivagans*), little brown myotis (*Myotis lucifugus*), northern myotis (*M. septentrionalis*), and tricolored bat (*Perimyotis subflavus*) (Hammerson and others, 2017). For the Southern Great Plains REA, we evaluated only the tree-roosting bat assemblage because of the lack of information on cave- and rock-roosting species. We had sufficient data to evaluate three species: eastern red bat, hoary bat, and silver-haired bat.

Human activities can both positively and negatively affect bats. For example, in some areas of the SGP, human activities have led to the expansion of habitat features used by bats, including trees, surface water, and buildings and mines used for roosting (Sparks and Choate, 2000). However, human disturbance at roost sites in caves, trees, buildings, or mines can lead to some species abandoning their roosts (Pierson and others, 1999; Hayes and others, 2015). Bat populations are vulnerable to rapid population declines and often take years or decades to recover, primarily because of low reproductive rates (Hutson and others, 2001; O'Shea and others, 2003).

Although habitat loss and disturbance can have adverse effects on bat populations, energy development and white-nose syndrome are responsible for unprecedented levels of bat mortality (U.S. Fish and Wildlife Service, 2012) and likely pose the greatest near-term threats to bat populations in the SGP. Energy development can have a variety of negative effects on bats, including mortality from collisions with infrastructure, disturbance, habitat loss, and contamination of food and water resources. Wind energy poses significant threats to bats, especially tree-roosting bats (Arnett and Baerwald, 2013); tens to hundreds of thousands of bats may die annually after colliding with wind turbines in North America (Cryan, 2011; Ellison, 2012). Seasonal movements may increase vulnerability of bats to mortality from wind turbines (Hayes and others, 2015).

Bat populations, especially those that hibernate in large congregations, are increasingly at risk from white-nose syndrome (WNS) (Lorch and others, 2016). Since the discovery of WNS in 2006, some bat populations (such as the little brown bat) may have declined by more than 75 percent (Hayes, 2015). It is estimated that more than 5.5 million bats of several species died between 2006 and 2011, leading to regional population collapses and vulnerability to extinction for some species (U.S. Fish and Wildlife Service, 2012). Cave-roosting bats of the SGP are at risk from the spread of WNS because of the nearly continuous distribution of cave and karst habitats where WNS has been confirmed (Culver and others, 1999; Veni, 2002). The fungus that causes WNS has been confirmed in the SGP (White-nose Syndrome Response Team, 2018). Additional background information on the bat assemblage can be found in the SGP pre-assessment report (Hayes, 2015).

Rapid Ecoregional Assessment Components Evaluated for the Tree-Roosting Bat Assemblage

The key ecological attributes and change agents addressed by core management questions for the tree-roosting bat assemblage (hereafter referred to as bat assemblage) habitat include amount and distribution, landscape structure, and development (tables 12–1 and 12–2). Fire occurrence and climate change were evaluated for the entire SGP (see Reese and others, 2017, chap. 3). Overall landscape-level ranking variables are summarized in table 12–3. The core and integrated management questions are listed in table 12–4.

Hoary bat. Photograph by Adam Searcy (Creative Commons Attribution-ShareAlike 2.0 Generic).

Table 12–1. Key ecological attributes and associated indicators used to address core management questions for the tree-roosting bat assemblage for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Amount and distribution	Total area	Baseline habitat (non-winter)[2]
Landscape structure	Patch size	Patch sizes for baseline habitat
Landscape dynamics	Fire occurrence	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]Baseline habitat was predicted using MaxEnt. See chapter 2, "Methods Overview for Species," for methods and datasets used.

Table 12–2. Anthropogenic change agents and associated indicators used to address core management questions for the tree-roosting bat assemblage for the Southern Great Plains Rapid Ecoregional Assessment.

[km, kilometer; mi, mile]

Attributes	Variables	Indicators[1]
Development	Terrestrial development index (TDI)	Percentage of baseline habitat in seven development classes based on a 2.5-km (1.55-mi) moving window
	Index of fragmentation	Patch sizes for relatively undeveloped[2] habitat
Climate change	Projected temperature and precipitation	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]Terrestrial development index score less than or equal to 2 percent.

Table 12–3. Landscape-level variables used to address the integrated management question for the tree-roosting bat assemblage. Ranks for landscape-level habitat area and development were combined into an overall landscape-level rank for the Southern Great Plains Rapid Ecoregional Assessment.

[>, greater than; km, kilometer; mi, mile]

Landscape-level variables[1]	Description	Relative rank[2]		
		Lowest	Medium	Highest
Area	Percentage of baseline habitat within a 5-km-radius (3.11-mi) moving window	>0–45.4	>45.4–74.3	>74.3
Development	Mean terrestrial development index (TDI) score for baseline habitat within a 5km-radius (3.11-mi) moving window	0–2	>2–10	>10

[1]See Reese and others (2017, chap. 2) for methods and datasets used to address integrated management questions.

[2]Ranking breakpoints for area of baseline habitat were determined from equal subsets of the data. Ranking breakpoints for terrestrial development index scores were standardized for all terrestrial conservation elements.

Table 12–4. Management questions addressed for the tree-roosting bat assemblage for the Southern Great Plains Rapid Ecoregional Assessment.

Core management questions[1]	Results
What is the distribution of baseline habitat for the bat assemblage?	Figure 12−1
Where does existing development pose the greatest threat to the bat assemblage habitat, and where are the large, relatively undeveloped areas?	Figures 12−2 and 12−3
How has development fragmented the bat assemblage habitat?	Figures 12−4 and 12−5
Integrated management question[2]	**Results**
Where is the bat assemblage habitat with the highest overall landscape-level rank?	Figure 12−6

[1]See Reese and others (2017, chap. 11) for management questions that could not be addressed.

[2]See table 12−3.

Management Questions and Results

What is the distribution of baseline habitat for the bat assemblage (fig. 12–1)?

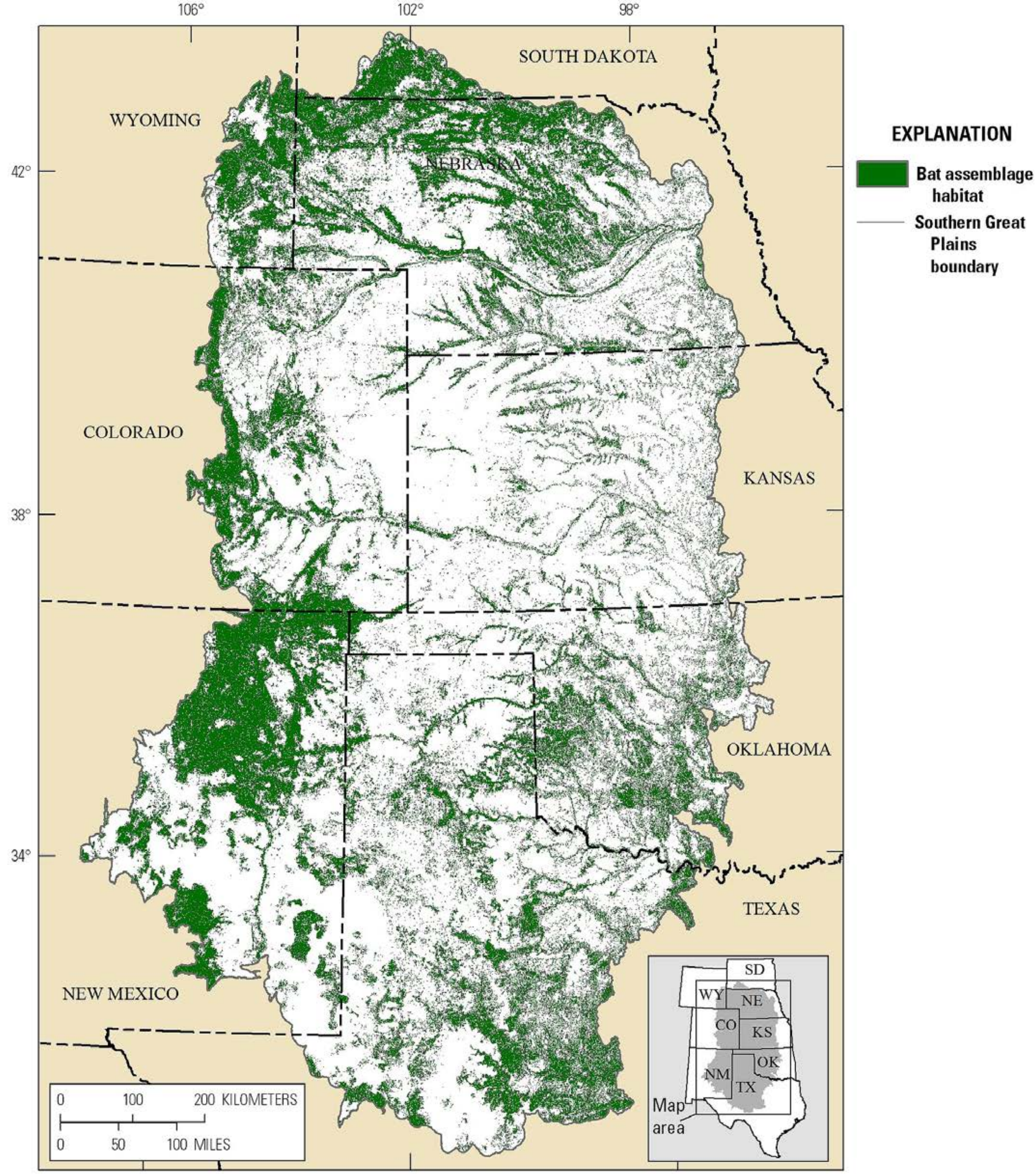


Figure 12–1. Predicted distribution of baseline habitat for the bat assemblage in the Southern Great Plains.

Where does existing development pose the greatest threat to the bat assemblage habitat, and where are the large, relatively undeveloped areas (figs. 12–2 and 12–3)?

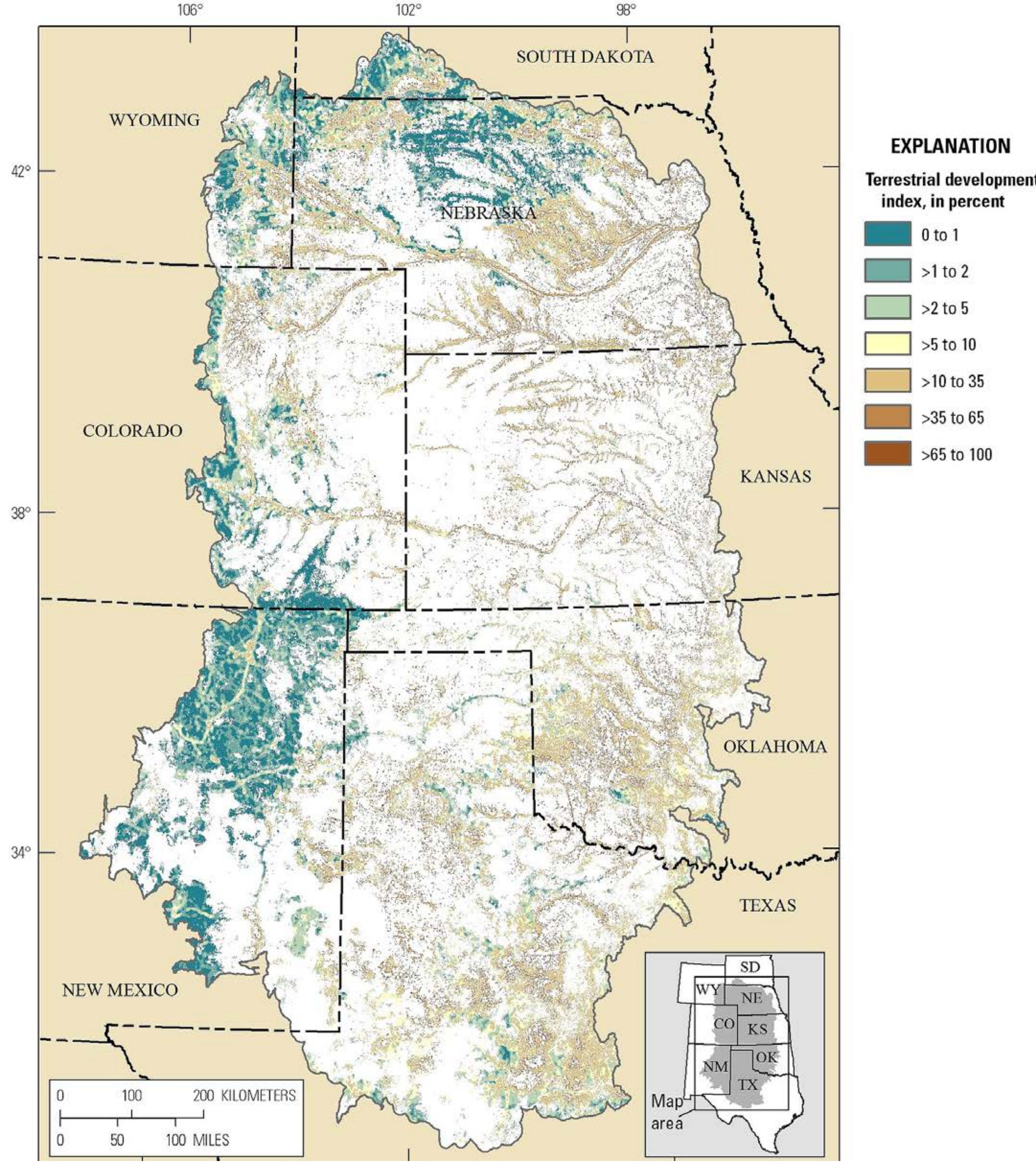


Figure 12–2. Terrestrial development index for bat assemblage baseline habitat in the Southern Great Plains.

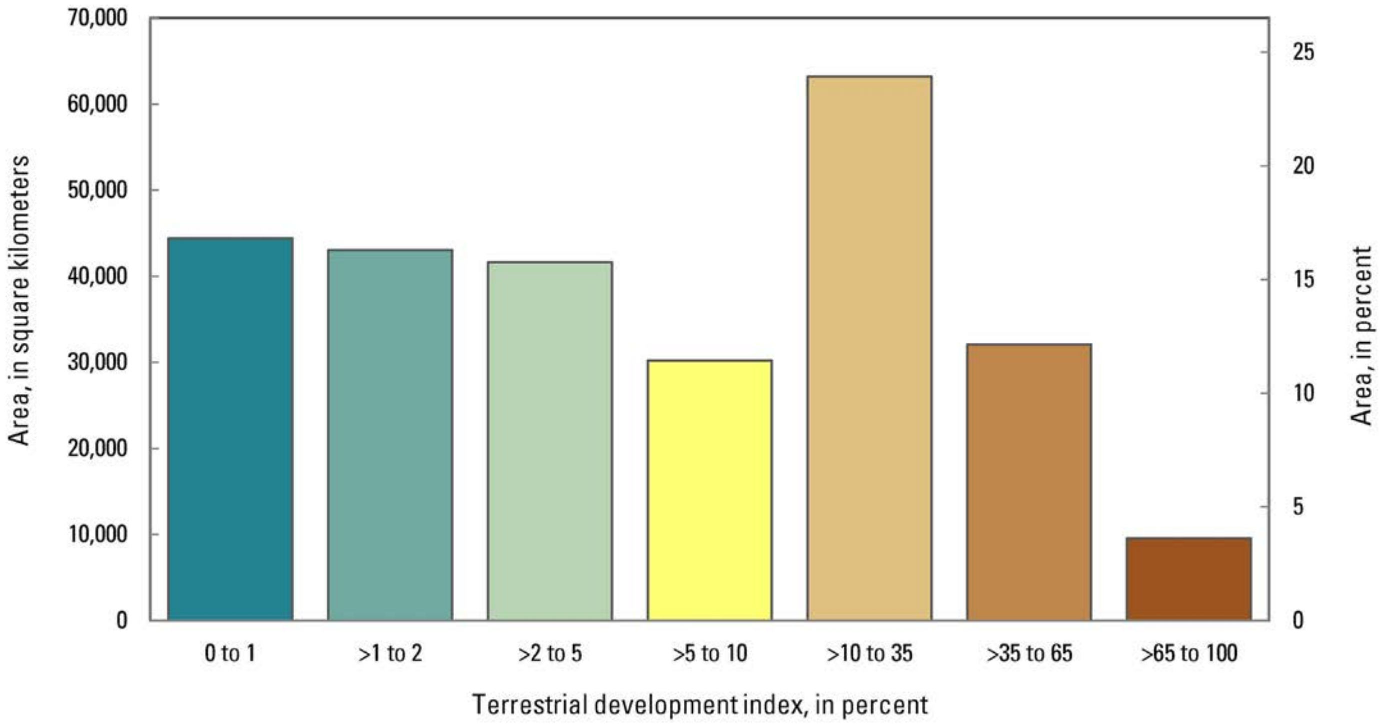


Figure 12–3. Area of the bat assemblage baseline habitat by terrestrial development index class in the Southern Great Plains.

How has development fragmented the bat assemblage habitat (figs. 12–4 and 12–5)?

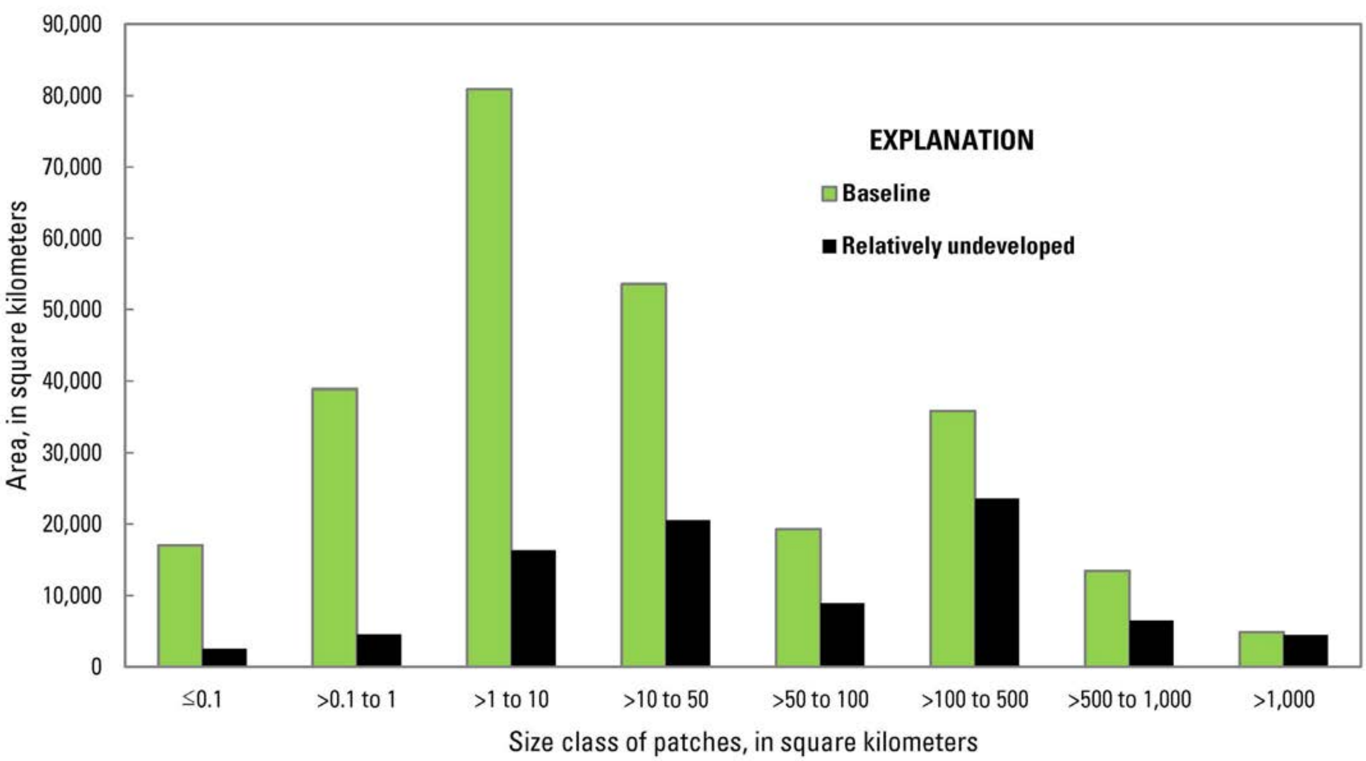


Figure 12–4. Area of the bat assemblage habitat in the Southern Great Plains by patch size class for baseline and relatively undeveloped conditions (terrestrial development index score ≤2 percent).

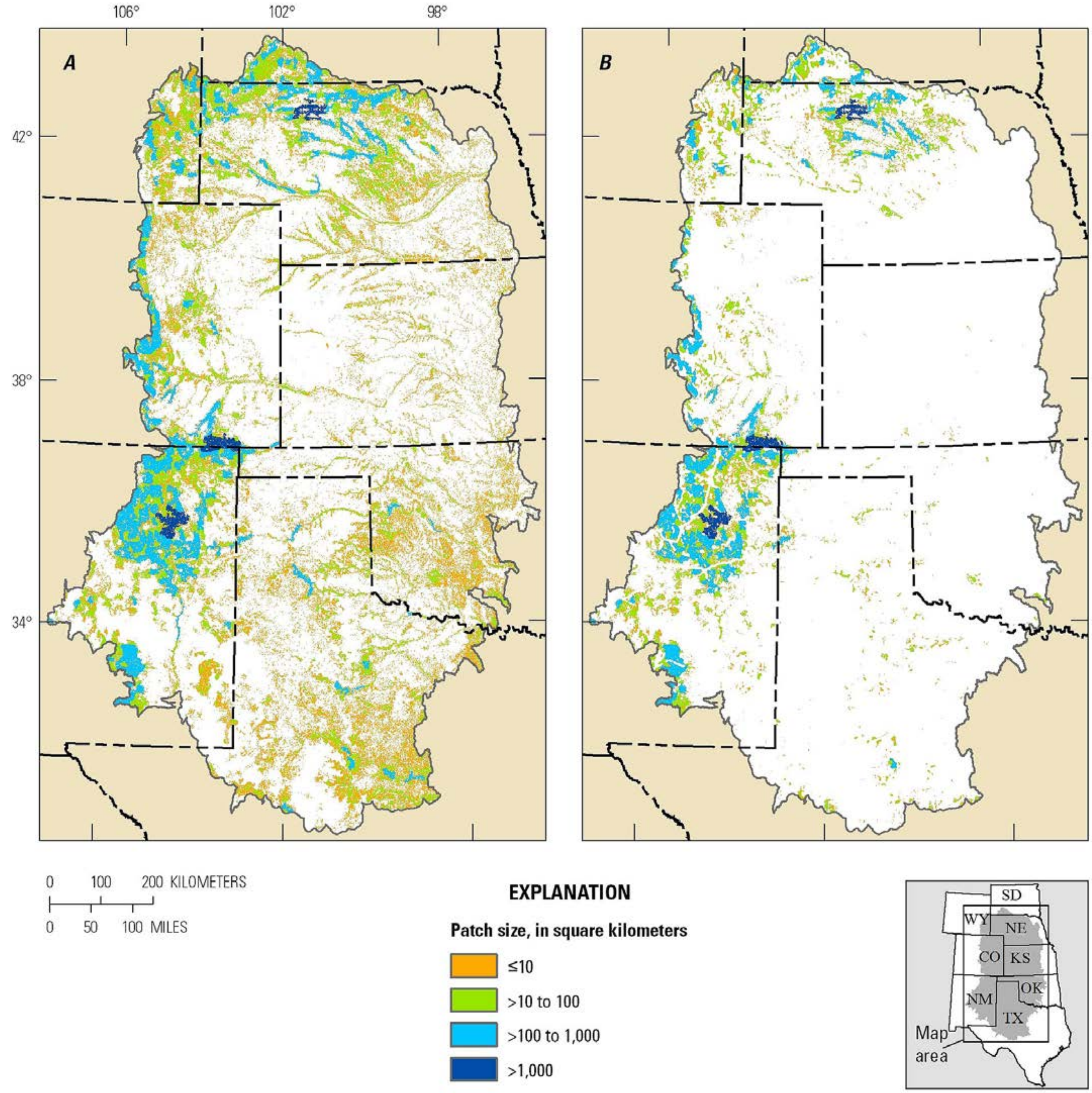


Figure 12–5. Patch size of the bat assemblage habitat in the Southern Great Plains. *A*, Baseline habitat. *B*, Relatively undeveloped habitat (terrestrial development index score ≤2 percent).

Where is the bat assemblage habitat with the highest overall landscape-level rank (fig. 12–6)?

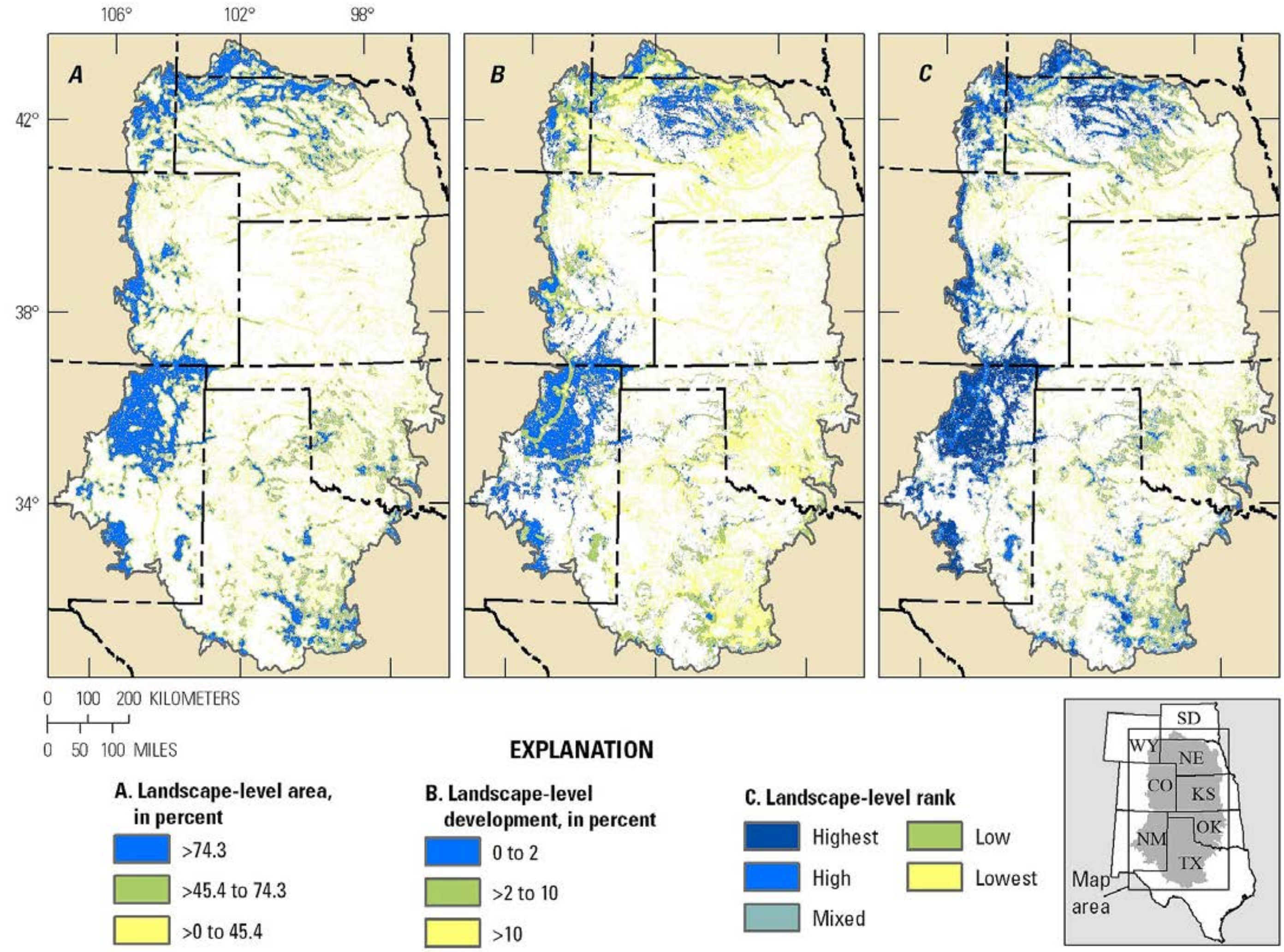


Figure 12–6. Landscape-level summaries for the bat assemblage habitat in the Southern Great Plains. Overall landscape-level rank (*C*) is derived from (*A*) landscape-level habitat area and (*B*) landscape-level development, summarized by a 5-kilometer-radius (3.11-mile) moving window (see table 12–3). Highest overall landscape-level rank corresponds to the largest landscape-level area and the lowest landscape-level development. Lowest overall landscape-level rank corresponds to the smallest landscape-level area and highest landscape-level development. Landscape-level ranks are not intended as standalone summaries and are best interpreted in conjunction with the geospatial datasets used to address core management questions.

Summary

- Baseline habitat for the tree-roosting bat assemblage is distributed throughout the SGP. There are approximately 264,000 km^2 (101,931 mi^2) of predicted habitat.
- Habitat for the tree-roosting bat assemblage with the lowest development is primarily concentrated in eastern New Mexico and the northern SGP (fig. 12–2). Approximately one-third of their habitat is relatively undeveloped (TDI score ≤2 percent), and 16 percent has low development (TDI scores 2–5 percent) (fig. 12–3). However, 16 percent of their habitat has very high development (TDI scores >35 percent).
- Habitat for the bat assemblage is highly fragmented throughout Texas, Oklahoma, Kansas, and southeastern Nebraska (figs. 12–4 and 12–5*A*). Approximately 72 percent of baseline habitat occurs in patches smaller than 50 km^2 (19.3 mi^2) (fig. 12–4). The largest relatively undeveloped patches are concentrated in northeastern New Mexico and southern Colorado (fig. 12–5*B*).
- The largest, most intact areas (the highest overall landscape-level rank) are in the western and northern portions of the SGP (fig. 10–6*C*).
- The broad-scale summaries provided by the REA are intended to be used in conjunction with local-level information on habitat conditions (Wood and others, 2017)

References Cited

Arnett, E.B., and Baerwald, E.F., 2013, Impacts of wind energy development on bats—Implications for conservation, *in* Adams, R.A., and Petersen, S.C., eds., Bat evolution, ecology and conservation: New York, Springer, p. 435–456.

Bogan, M.A., Cryan, P.M., Valdez, E.W., Ellison, L.E., and O’Shea, T.J., 2003, Western crevice and cavity-roosting bats, *in* O’Shea, T.J., and Bogan, M.A., eds., Monitoring trends in bat populations of the United States and territories—Problems and prospects: U.S. Geological Survey Information and Technology Report 2003–0003, p. 69–78.

Carter, T.C., Menzel, M.A., and Saugey, D.A., 2003, Population trends of solitary foliage-roosting bats, *in* O’Shea, T.J., and Bogan, M.A., eds., Monitoring trends in bat populations of the United States and territories—Problems and prospects: U.S. Geological Survey Information and Technology Report 2003–0003, p. 41–48.

Cryan, P.M., 2011, Wind turbines as landscape impediments to the migratory connectivity of bats: Environmental Law, v. 41, p. 355–370.

Culver, D.C., Hobbs, H.H., III, Christman, M.C., and Master, L.L., 1999, Distribution map of caves and cave animals in the United States: Journal of Cave and Karst Studies, v. 61, p. 139–140.

Dalquest, W.W., Stangl, F.B., and Jones, J.K., Jr., 1990, Mammalian zoogeography of a Rocky Mountain–Great Plains interface in New Mexico, Oklahoma, and Texas: Lubbock, Tex., Texas Tech University Press, Special Publication Number 34, 78 p.

Ellison, L.E., 2012, Bats and wind energy—A literature synthesis and annotated bibliography: U.S. Geological Survey Open-File Report 2012–1110, 57 p., accessed February 2014 at http://pubs.usgs.gov/of/2012/1110/OF12-1110.pdf.

Hammerson, G.A., Kling, M., Harkness, M., Ormes, M., and Young, B.E., 2017, Strong geographic and temporal patterns in conservation status of North American bats: Biological Conservation, v. 212, p. 144–152.

Hayes, M.A., 2015, Bat species assemblage, chap. 19 *of* Assal, T.J., Melcher, C.P., and Carr, N.B., eds., 2015, Southern Great Plains Rapid Ecoregional Assessment—Pre-assessment report: U.S. Geological Survey Open-File Report 2015–1003, p. 118–126, accessed July 13, 2017, at https://doi.org/10.3133/ofr20151003.

Hayes, M.A., Cryan, M.A., and Wunder, M.B., 2015, Seasonally-dynamic presence-only species distribution models for cryptic migratory bat impacted by wind energy development: PLoS ONE, v. 10, no. 7, article e0132599, 20 p., accessed May 16, 2016, at http://journals.plos.org/plosone/article?id=10.1371/journal.pone.0132599

Humphrey, S.R., and Kunz, T.H., 1976, Ecology of a Pleistocene relict, the western big-eared bat (*Plecotus townsendii*), in the southern Great Plains: Journal of Mammalogy, v. 57, p. 470–494.

Hutson, A.M., Mickleburgh, S.P., and Racey, P.A., 2001, Microchiropteran bats—Global status survey and conservation action plan: Gland, Switzerland, and Cambridge, United Kingdom, International Union for Conservation of Nature, Species Survival Commission, Chiropteran Specialist Group, 259 p.

Lorch, J.M., Palmer, J.M., Lindner, D.L., Ballmann, A.E., George, K.G., Griffin, K., Knowles, S., Huckabee, J.R., Haman, K.H., Anderson, C.D., Becker, P.A., Buchanan, J.B., Foster, J.T., and Blehert, D.S., 2016, First detection of bat white-nose syndrome in western North America: mSphere, v. 1, no. 4, article e00148-16, 5 p., accessed January 25, 2018, at https://doi.org/10.1128/mSphere.00148-16.

Miller, J.C., 2011, Habitat relationships and conservation of bats within the Red Hills of Kansas and Oklahoma: Emporia, Kans., Emporia State University, M.S. thesis, 120 p.

O'Shea, T.J., Bogan, M.A., and Ellison, L.E., 2003, Monitoring trends in bat populations of the United States and territories—Status of the science and recommendations for the future: Wildlife Society Bulletin, v. 31, p. 16–29.

Pierson, E.D., Wackenhut, M.C., Altenbach, J.S., Bradley, P., Call, P., Genter, D.L., Harris, C.E., Keller, B.L., Lengus, B., Lewis, L., Luce, B., Navo, K.W., Perkins, J.M., Smith, S., and Welch, L., 1999, Species conservation assessment and strategy for Townsend's big-eared bat (*Corynorhinus townsendii townsendii* and *Corynorhinus townsendii pallescens*): Boise, Idaho, Idaho Conservation Effort, Idaho Department of Fish and Game, 66 p.

Prendergast, J.A., Jensen, W.E., and Roth, S.D., 2010, Trends in abundance of hibernating bats in a karst region of the southern Great Plains: Southwestern Naturalist, v. 55, p. 331–339.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100.

Sparks, D.W., and Choate, J.R., 2000, Distribution, natural history, and conservation of bats in Kansas, *in* Choate, J.R., ed., Reflections of a naturalist—Papers honoring Professor Eugene D. Fleharty: Hays, Kans., Fort Hays State University, Sternberg Museum of Natural History, Fort Hays Studies special issue no. 1, p. 173–228.

U.S. Fish and Wildlife Service, 2012, North American bat death toll exceeds 5.5 million from white-nose syndrome: U.S. Fish and Wildlife Service news release, January 17, 2012, 2 p., accessed February 2014 at http://www.batcon.org/pdfs/USFWS_WNS_Mortality_2012_NR_FINAL.pdf.

Veni, G., 2002, Revising the karst map of the United States: Journal of Cave and Karst Studies, v. 64, p. 45–50.

White-Nose Syndrome Response Team, 2018, White-nose syndrome occurrence map—by year (2018): U.S. Fish and Wildlife Service White-Nose Syndrome Reponse Team, accessed June 1, 2018, at https://www.whitenosesyndrome.org.

Wood, J.A., Carter, S.K., Litshert, S.E., and Carr, N.B., 2017, Including broad-scale indicators in multiscale natural resource monitoring and assessment programs in the Bureau of Land Management, chap. 3 *of* Carter, S.K., Carr, N.B., Miller, K.H., and Wood, D.J.A., eds., Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, p. 15–37, accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Chapter 13. Swift Fox

Introduction

The swift fox (*Vulpes velox*) is native to the shortgrass and mixed-grass prairies of the Great Plains (Egoscue, 1979; Olson and Lindzey, 2002a; Harrison and Schmitt, 2003; Gese and Thompson, 2014). Conversion of native grasslands to cropland, trapping for the fur trade, and indiscriminant poisoning of other mammals led to population declines across its range (Kilgore, 1969; Egoscue, 1979; Allardyce and Sovada, 2003; Sovada and others, 2009). Following reintroductions and the cessation of poisoning campaigns, populations began dramatic natural recoveries (Soper, 1964; Allardyce and Sovada, 2003; Sovada and Carbyn, 2003; Russell, 2006; Sovada and others, 2009; Cullingham and Moehrenschlager, 2013). Currently, most populations are apparently stable or increasing (Olson and Lindzey, 2002a, b; Stephens and Anderson, 2005; Moehrenschlager and others, 2013).

Swift foxes typically inhabit areas of flat to gently rolling topography dominated by short grasses. Populations are occasionally found in areas with scattered shrubs (Olson and Lindzey, 2002a, b; Dark-Smiley and Keinath, 2003; Thompson and Gese, 2007; Thompson and others, 2008) and mixed agriculture (Cutter, 1958; Kilgore, 1969; Sovada and others, 2003). It is generally assumed that short, open cover typical of swift fox habitat enhances detection of predators, especially coyotes (*Canis latrans*), their primary predator (Russell, 2006; Sovada and others, 2009).

The conversion of native prairie to cropland has greatly fragmented and reduced swift fox habitat (Sovada and others, 2009; Schwalm and others, 2014). Conversion to croplands was most prevalent in mixed-grass prairie, and most of the remaining swift fox habitat occurs within shortgrass prairie, including areas used for grazing and dry-land farming (Matlack and others, 2000; Kamler and others, 2003; Sovada and others, 2009). Swift foxes appear to prefer native grasslands and rangelands over cultivated areas (Kamler and others, 2003), but they often inhabit non-irrigated croplands (Sovada and others, 2003; Stephens and Anderson, 2005). Swift foxes may construct dens in fallow fields, but generally they avoid irrigated cropland and Conservation Reserve Program land, which is often planted with mid-height or tallgrass species (Jackson and Choate, 2000; Kamler and others, 2003; Sovada and others, 2003, 2009). Much of the current swift fox habitat occurs on rangeland, and livestock grazing can be beneficial to swift foxes by reducing vegetation height and cover (Stephens and Anderson, 2005). Swift foxes may benefit from a mosaic of vegetation structures maintained by variable grazing intensity and other disturbances across the landscape, which may not be fully replicated by some grazing practices.

Roads and energy development can negatively affect swift foxes. Swift foxes appear to be fairly tolerant of roads, and it has been suggested that this tolerance may be in response to reduced predation pressure from coyotes, which tend to avoid roads (Kamler and others, 2003). In areas where roads have fragmented habitat, mortality from vehicles can exceed mortality from coyote predation (Sovada and others, 2003; Meyer, 2009). The effects of energy development on swift foxes are largely unknown (Moehrenschlager and others, 2004), but oil and gas, wind, and biofuels production have the potential to further exacerbate effects of habitat loss and fragmentation (Moehrenschlager and Sovada, 2004; Committee on the Status of Endangered Wildlife in Canada, 2009).

The effects of fire suppression on the swift fox in native grasslands have likely been mostly negative, and prescribed fire may generally benefit the swift fox, particularly in areas where shrub densities have increased as the result of fire exclusion (Thompson and others, 2008; Meyer, 2009; Gese and Thompson, 2014). Additional background information on the swift fox can be found in the SGP pre-assessment report (Carr and Melcher, 2015).

Rapid Ecoregional Assessment Components Evaluated for the Swift Fox

The key ecological attributes and change agents addressed by core management questions for swift fox habitat include amount and distribution, landscape structure, and development (tables 13–1 and 13–2). Invasive woody species and climate change were evaluated for grassland communities (see Reese and others, 2017, chap. 4). Fire occurrence and potentially altered vegetation (including invasive herbaceous plants) were evaluated for the entire SGP (see Reese and others, 2017, chap. 3). Overall landscape-level ranking variables are summarized in table 13–3. The core and integrated management questions are listed in table 13–4.

Swift fox. Photograph by Tony Ifland, U.S. Fish and Wildlife Service (Creative Commons Attribution 2.0 Generic).

Table 13–1. Key ecological attributes and associated indicators used to address core management questions for swift foxes for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Amount and distribution	Total area	Baseline habitat (year round)[2]
Landscape structure	Patch size	Patch sizes for baseline habitat
Landscape dynamics	Fire occurrence	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]Baseline habitat was estimated using data from Sovada and others (2009). See chapter 2, "Methods Overview for Species," for methods and datasets used.

Table 13–2. Anthropogenic change agents and associated indicators used to address core management questions for swift foxes for the Southern Great Plains Rapid Ecoregional Assessment.

[km, kilometer; mi, mile]

Attributes	Variables	Indicators[1]
Development	Terrestrial development index (TDI)	Percentage of baseline habitat in seven development classes based on a 2.5-km (1.55-mi) moving window
	Index of fragmentation	Patch sizes for relatively undeveloped[2] habitat
Invasive species	Potential for woody species expansion	See Reese and others (2017, chap. 4)
Climate change	Projected temperature and precipitation	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]Terrestrial development index score less than or equal to 2 percent.

Table 13–3. Landscape-level variables used to address the integrated management question for swift foxes. Ranks for landscape-level habitat area and development were combined into an overall landscape-level rank for the Southern Great Plains Rapid Ecoregional Assessment.

[>, greater than; km, kilometer; mi, mile]

Landscape-level variables[1]	Description	Relative rank[2]		
		Lowest	Medium	Highest
Area	Percentage of baseline habitat within a 5-km-radius (3.11-mi) moving window	>0–51.2	>51.2–75.3	>75.3
Development	Mean terrestrial development index (TDI) score for baseline habitat within a 5km-radius (3.11-mi) moving window	0–2	>2–10	>10

[1]See Reese and others (2017, chap. 2) for methods and datasets used to address integrated management questions.

[2]Ranking breakpoints for area of baseline habitat were determined from equal subsets of the data. Ranking breakpoints for terrestrial development index scores were standardized for all terrestrial conservation elements.

Table 13–4. Management questions addressed for swift foxes for the Southern Great Plains Rapid Ecoregional Assessment.

Core management questions[1]	Results
What is the distribution of baseline habitat for swift foxes?	Figure 13−1
Where does existing development pose the greatest threat to swift fox habitat, and where are the large, relatively undeveloped areas?	Figures 13−2 and 13−3
How has development fragmented swift fox habitat?	Figures 13−4 and 13−5
Integrated management question[2]	**Results**
Where is swift fox habitat with the highest overall landscape-level rank?	Figure 13−6

[1]See Reese and others (2017, chap. 11) for management questions that could not be addressed.

[2]See table 13−3.

Management Questions and Results

What is the distribution of baseline habitat for swift foxes (fig. 13–1)?

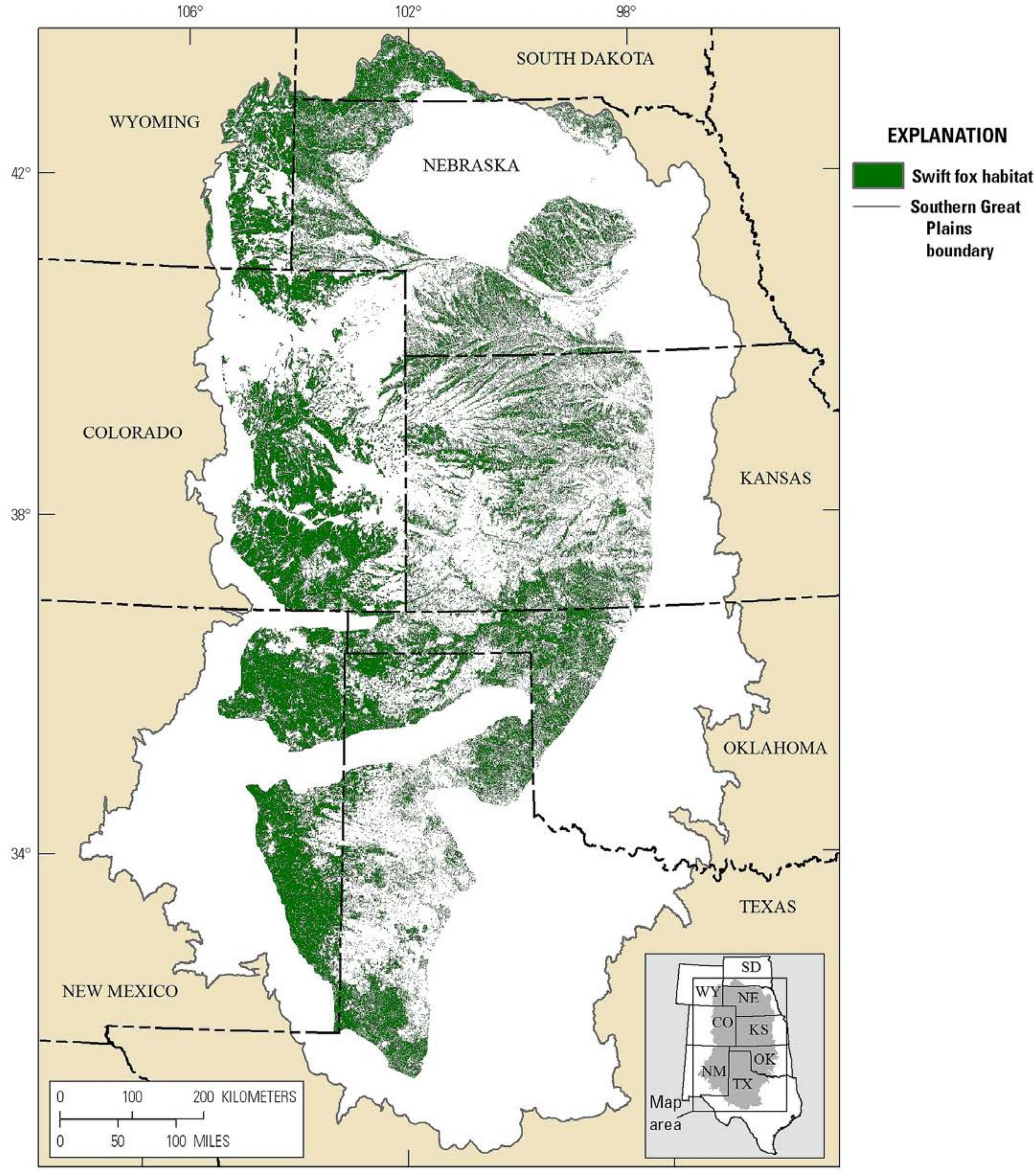


Figure 13–1. Distribution of baseline habitat for swift foxes in the Southern Great Plains.

Where does existing development pose the greatest threat to swift fox habitat, and where are the large, relatively undeveloped areas (figs. 13–2 and 13–3)?

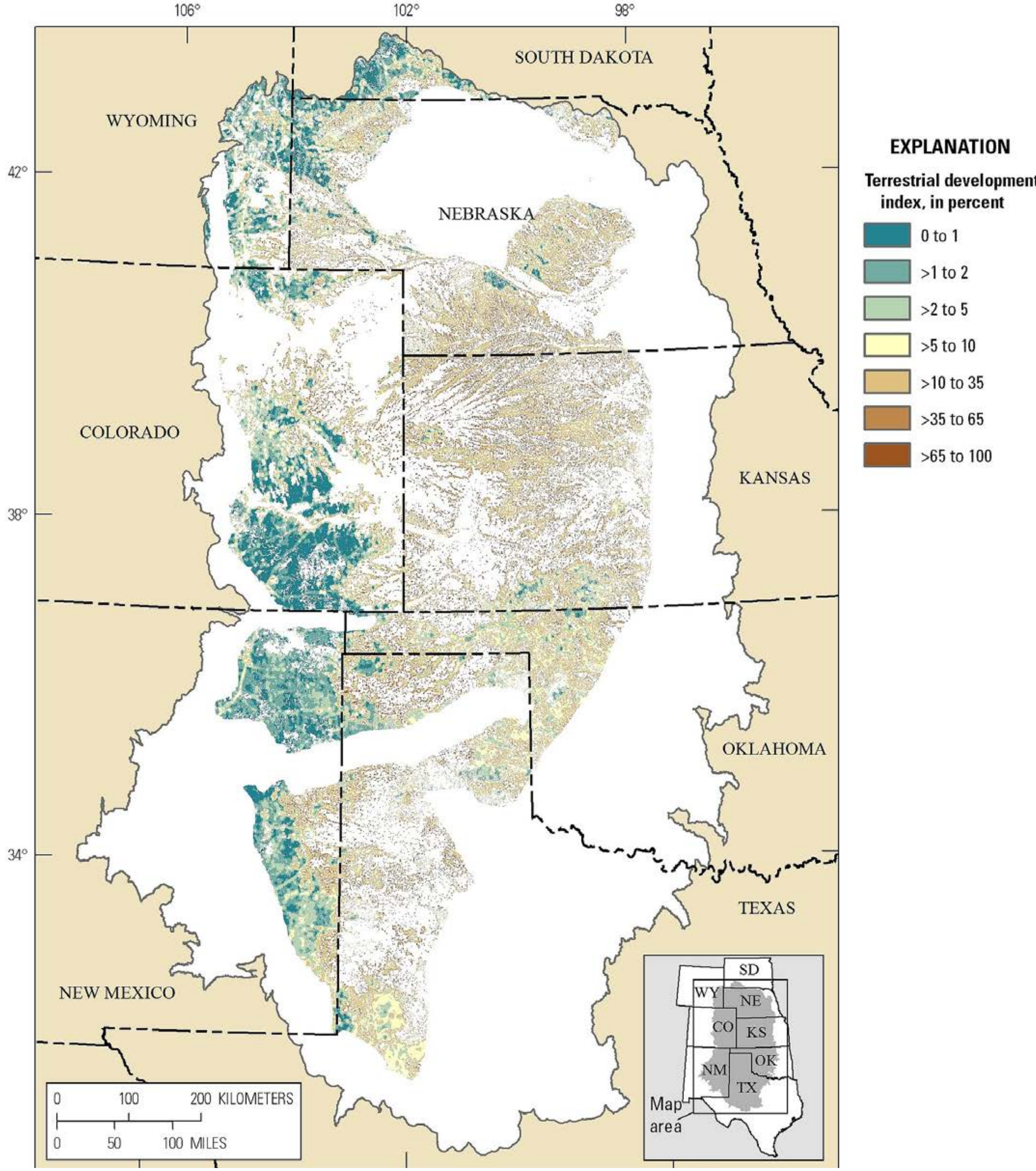


Figure 13–2. Terrestrial development index for swift fox baseline habitat in the Southern Great Plains.

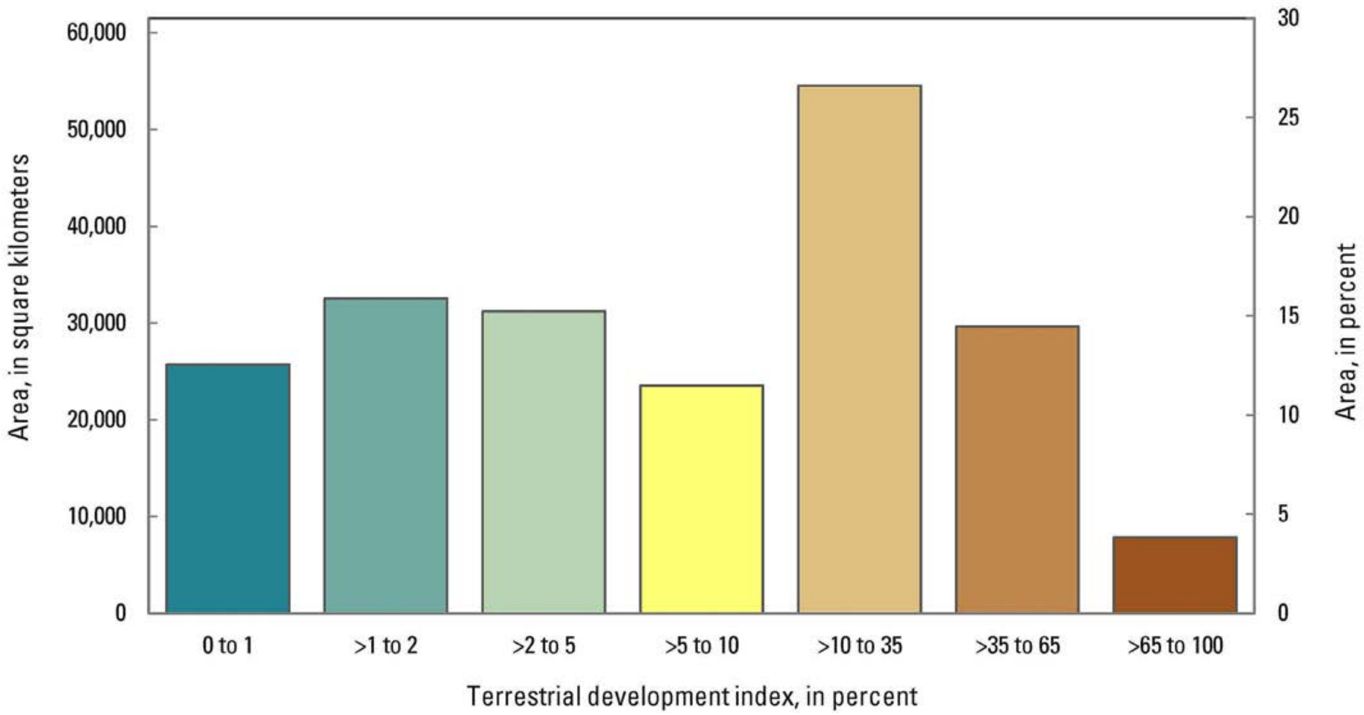


Figure 13–3. Area of swift fox baseline habitat by terrestrial development index class in the Southern Great Plains.

How has development fragmented swift fox habitat (figs. 13–4 and 13–5)?

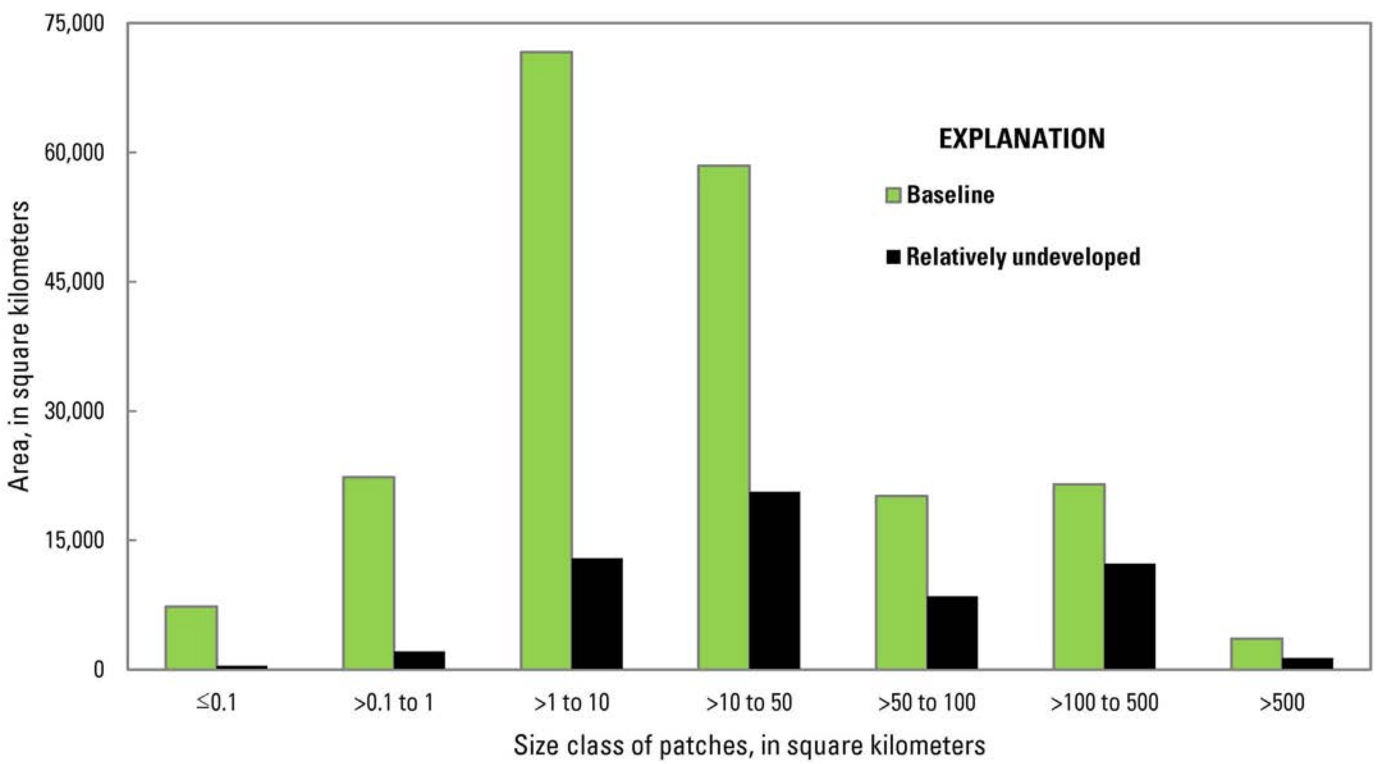


Figure 13–4. Area of swift fox habitat in the Southern Great Plains by patch size class for baseline and relatively undeveloped conditions (terrestrial development index score ≤2 percent).

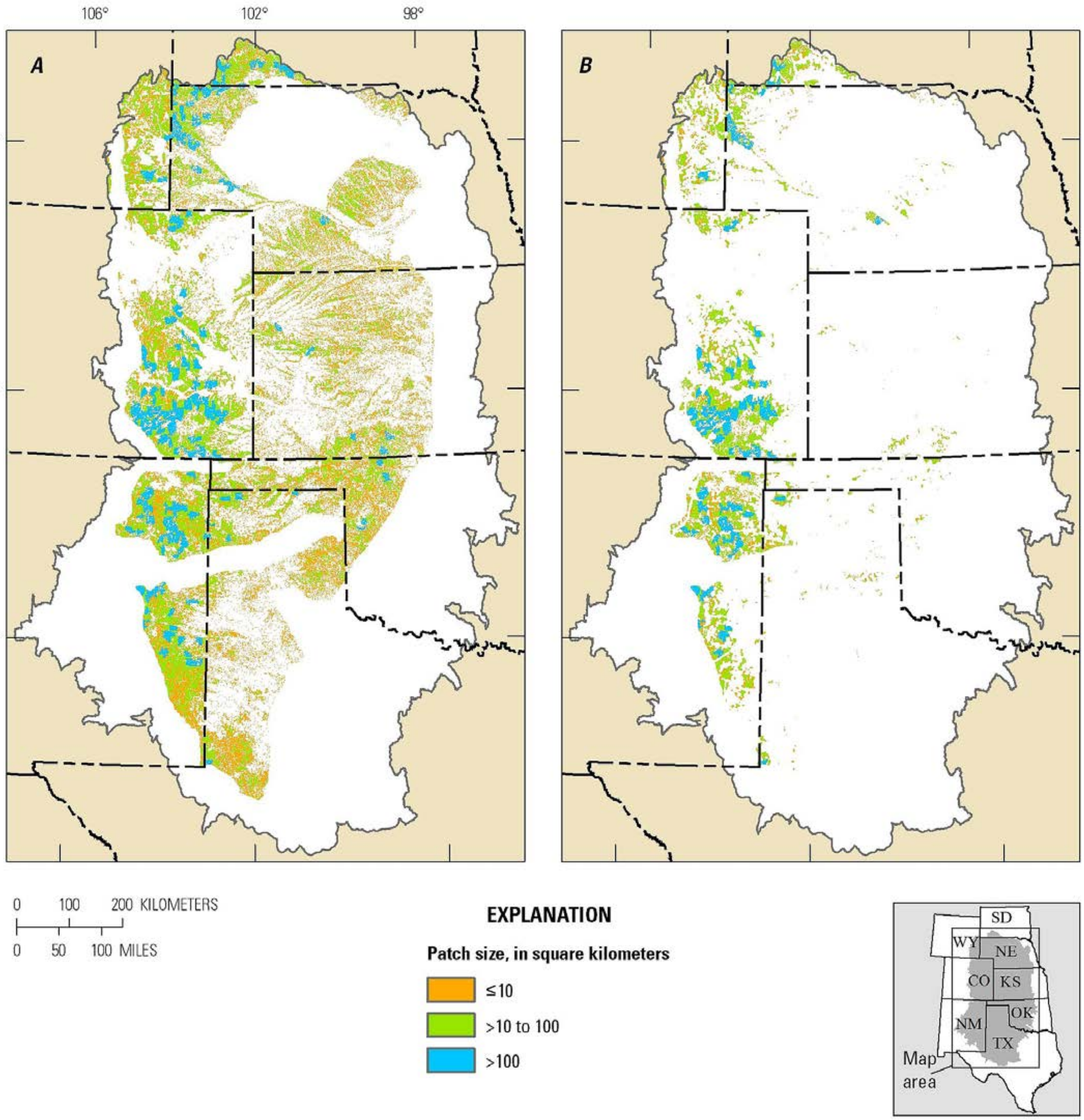


Figure 13–5. Patch size of swift fox habitat in the Southern Great Plains. *A,* Baseline habitat. *B,* Relatively undeveloped habitat (terrestrial development index score ≤2 percent).

Where is swift fox habitat with the highest overall landscape-level rank (fig. 13–6)?

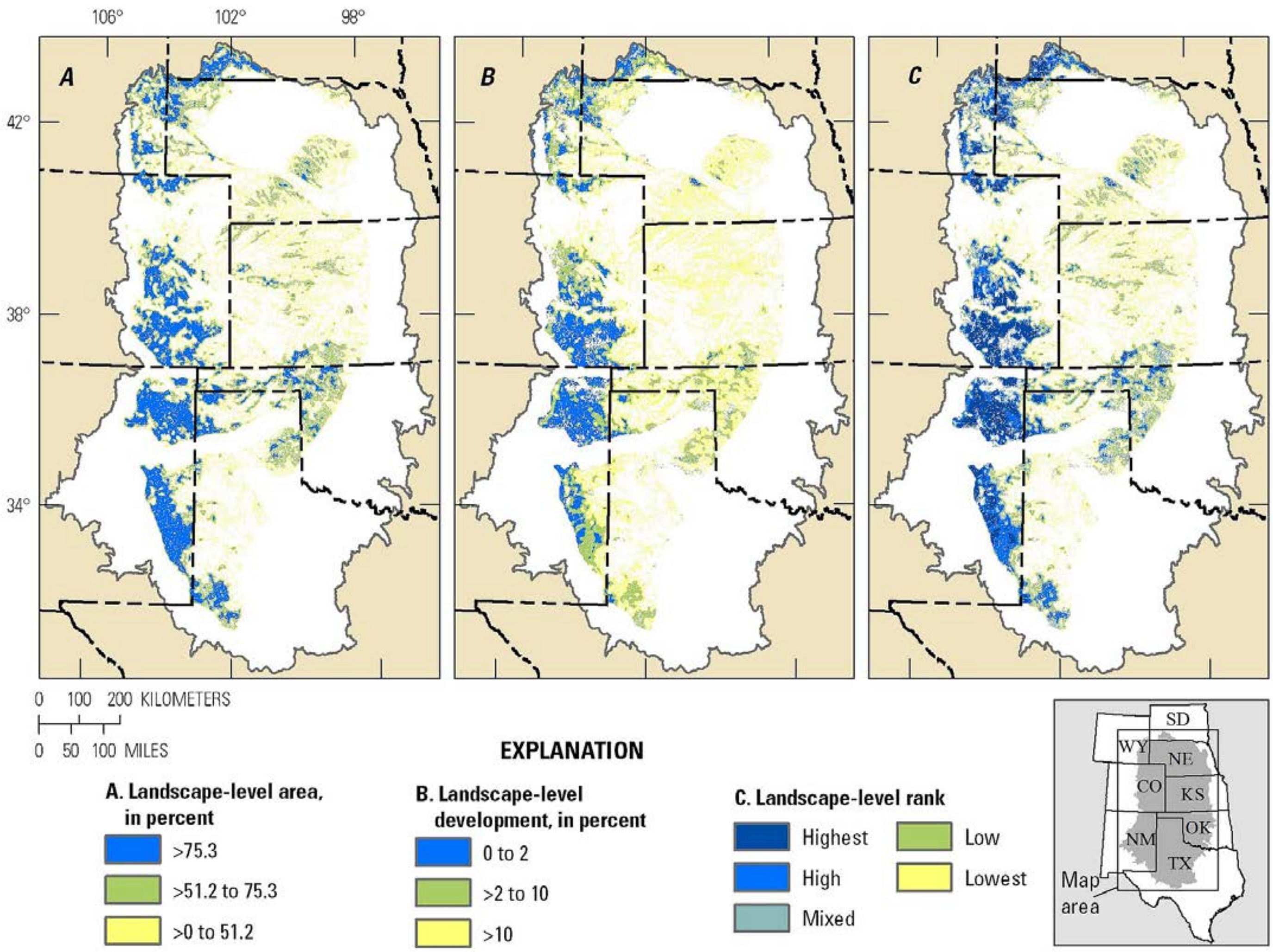


Figure 13–6. Landscape-level summaries for swift fox habitat in the Southern Great Plains. Overall landscape-level rank (*C*) is derived from (*A*) landscape-level habitat area and (*B*) landscape-level development, summarized by a 5-kilometer-radius (3.11-mile) moving window (see table 13–3). Highest overall landscape-level rank corresponds to the largest landscape-level area and the lowest landscape-level development. Lowest overall landscape-level rank corresponds to the smallest landscape-level area and highest landscape-level development. Landscape-level ranks are not intended as standalone summaries and are best interpreted in conjunction with the geospatial datasets used to address core management questions.

Summary

- Baseline habitat for the swift fox spans almost the entire distribution of shortgrass and mixed-grass prairie of the SGP (fig. 13–1). There are approximately 205,000 km^2 (79,151 mi^2) of baseline habitat in the SGP.
- Swift fox habitat with the lowest development is concentrated in southeastern Colorado, eastern New Mexico, and the northeastern portions of the SGP (fig. 13–2). Approximately 28 percent of its habitat is relatively undeveloped (TDI score ≤2 percent), and 15 percent has low development (TDI scores 2–5 percent) (fig. 13–3). However, 18 percent of its habitat has very high development (TDI scores >35 percent).
- Swift fox habitat throughout Texas, Oklahoma, Kansas, and southern Nebraska is highly fragmented, primarily by agriculture (figs. 13–4 and 13–5). Approximately 78 percent of baseline habitat occurs in patches smaller than 50 km^2 (19.3 mi^2) (fig. 13–4). The largest relatively undeveloped habitat patches are in southeastern Colorado and northeastern New Mexico (fig. 13–5*B*).
- The largest, most intact areas (the highest overall landscape-level rank) are in the western extent of the shortgrass prairie (fig. 13–6*C*), where there has been less conversion to croplands than in other areas of the region (Reese and others, 2017).
- The broad-scale summaries provided by the REA are intended to be used in conjunction with local-level information on habitat conditions (Wood and others, 2017).

References Cited

Allardyce, D., and Sovada, M.A., 2003, Review of the ecology, distribution, and status of swift foxes in the United States, *in* Sovada, M.A., and Carbyn, L., eds., The swift fox—Ecology and conservation of swift foxes in a changing world: Regina, Saskatchewan, Canada, University of Regina Press, Canadian Plains Proceedings 34, p. 3–18.

Carr, N.B., and Melcher, C.P., 2015, Swift fox, chap. 22 *of* Assal, T.J., Melcher, C.P., and Carr, N.B., eds., 2015, Southern Great Plains Rapid Ecoregional Assessment—Pre-assessment report: U.S. Geological Survey Open-File Report 2015–1003, p. 118–126, accessed July 13, 2017, at https://doi.org/10.3133/ofr20151003.

Committee on the Status of Endangered Wildlife in Canada, 2009, Committee on the Status of Endangered Wildlife in Canada assessment and status report on the swift fox *Vulpes velox* in Canada: Ottawa, Canada, Committee on the Status of Endangered Wildlife in Canada, 49 p., accessed February 2014 at http://publications.gc.ca/collections/collection_2011/ec/CW69-14-108-2010-eng.pdf.

Cullingham, C.I., and Moehrenschlager, A., 2013, Temporal analysis of genetic structure to assess population dynamics of reintroduced swift foxes: Conservation Biology, v. 27, no. 6, p. 1389−1398.

Cutter, W.L., 1958, Denning of swift fox in northern Texas: Journal of Mammalogy, v. 39, p.70–74.

Dark-Smiley, D.N., and Keinath, D.A., 2003, Species assessment for swift fox (*Vulpes velox*) in Wyoming: Cheyenne, Wyo., Bureau of Land Management, Wyoming State Office, 51 p., accessed February 2014 at http://www.blm.gov/pgdata/etc/medialib/blm/wy/wildlife/animal-assessmnts.Par.72741.File.dat/SwiftFox.pdf.

Egoscue, H.J., 1979, *Vulpes velox*: Mammalian Species, v. 122, p. 1–5.

Gese, E.M., and Thompson, C.M., 2014, Does habitat heterogeneity in a multi-use landscape influence survival rates and density of a native mesocarnivore?: PLoS ONE, v. 9, no. 6, article e100500, 9 p., accessed October 13, 2017, at https://doi.org/10.1371/journal.pone.0100500.

Harrison, R.L., and Schmitt, C.G., 2003, Current swift fox distribution and habitat selection within areas of historical occurrence in New Mexico, *in* Sovada, M.A., and Carbyn, L., eds., The swift fox—Ecology and conservation of swift foxes in a changing world: Regina, Saskatchewan, Canada, University of Regina Press, Canadian Plains Proceedings 34, p. 71–77.

Jackson, V.L., and Choate, J.R., 2000, Dens and den sites of the swift fox, *Vulpes velox*: Southwestern Naturalist, v. 45, p. 212–220.

Kamler, J.F., Ballard, W.B., Fish, E.B., Lemons, P.R., Mote, K.P., and Perchellet, C.C., 2003, Habitat use, home ranges, and survival of swift foxes in a fragmented landscape—Conservation implications: Journal of Mammalogy, v. 84, p. 989–995.

Kilgore, D.L., Jr., 1969, An ecological study of swift fox (*Vulpes velox*) in the Oklahoma Panhandle: American Midland Naturalist, v. 81, p. 512–534.

Matlack, R.S., Gipson, P.S., and Kaufman, D.W., 2000, The swift fox in rangeland and cropland in western Kansas—Relative abundance, mortality, and body size: Southwestern Naturalist, v. 45, p. 221–225.

Meyer, R., 2009, *Vulpes velox*: U.S. Department of Agriculture Forest Service Fire Effects Information System, accessed January 30, 2014, at http://www.fs.fed.us/database/feis/animals/mammal/vuve/all.html.

Moehrenschlager, A., Cypher, B.L., Ralls, K., List, R., and Sovada, M.A., 2004, Swift and kit foxes—Comparative ecology and conservation priorities of swift and kit foxes, *in* MacDonald, D.W., and Sillero-Zurbi, C., eds., The biology and conservation of wild canids: Oxford, United Kingdom, Oxford University Press, p. 185−198.

Moehrenschlager, A., and Sovada, M.A., 2004, Swift fox—*Vulpes velox*, *in* Sillero-Zubiri, C., Hoffman, M., and Macdonald, D.W., eds., Canids—Foxes, wolves, jackals, and dogs. Status survey and conservation action plan: Gland, Switzerland, and Cambridge, United Kingdom, International Union for Conservation of Nature Species Survival Commission, Canid Specialist Group, p. 109–116, accessed June 1, 2014, at http://www.carnivoreconservation.org/files/actionplans/canids.pdf.

Moehrenschlager, A., Sovada, M., and members of the International Union for Conservation of Nature Species Survival Commission, Canid Specialist Group—North America Regional Section, 2013, *Vulpes velox*: International Union for Conservation of Nature Red List of Threatened Species (version 2013.2), accessed June 1, 2014, at http://www.iucnredlist.org/details/23059/0.

Olson, T.L., and Lindzey, F.G., 2002a, Swift fox (*Vulpes velox*) home-range dispersion patterns in southeastern Wyoming: Canadian Journal of Zoology, v. 80, p. 2024–2029.

Olson, T.L., and Lindzey, F.G., 2002b, Swift fox survival and production in southeastern Wyoming: Journal of Mammalogy, v. 83, p. 199–206.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100.

Russell, T.A., 2006, Habitat selection by swift foxes in Badlands National Park and the surrounding area in South Dakota: Brookings, S.D., South Dakota State University, M.Sc. thesis, 104 p.

Schwalm, D., Waits, L.P., and Ballard, W.B., 2014, Little fox of the prairie—Genetic structure and diversity throughout the distribution of a grassland carnivore in the United States: Conservation Genetics, v. 15, p. 1503−1514.

Soper, J.D., 1964, The mammals of Alberta: Edmonton, Canada, Hamly Press Ltd., 402 p.

Sovada, M.A., and Carbyn, L., eds., 2003, The swift fox—Ecology and conservation of swift foxes in a changing world: Regina, Saskatchewan, Canada, University of Regina Press, Canadian Plains Research Center Proceedings, v. 34, 250 p.

Sovada, M.A., Slivinski, C.C., Woodward, R.O., and Phillips, M.L., 2003, Home range, habitat use, litter size, and pup dispersal of swift foxes in two distinct landscapes of western Kansas, *in* Sovada, M.A., and Carbyn, L., eds., The swift fox—Ecology and conservation of swift foxes in a changing world: Regina, Saskatchewan, Canada, University of Regina Press, Canadian Plains Proceedings 34, p. 149–160.

Sovada, M.A., Woodward, R.O., and Igl, L.D., 2009, Historical range, current distribution, and conservation status of the swift fox, *Vulpes velox*, in North America: Canadian Field-Naturalist, v. 123, p. 346–367.

Stephens, R.M., and Anderson, S.H., 2005, Swift fox (*Vulpes velox*)—A technical conservation assessment: U.S. Department of Agriculture Forest Service, 45 p., accessed June 1, 2014, at http://www.fs.fed.us/r2/projects/scp/assessments/swiftfox.pdf.

Thompson, C.M., Augustine, D.J., and Mayers, D.M., 2008, Swift fox response to prescribed fire in shortgrass steppe: Western North American Naturalist, v. 68, p. 251–256.

Thompson, C.M., and Gese, E.M., 2007, Food webs and intraguild predation—Community interactions of a native mesocarnivore: Ecology, v. 88, p. 334–346.

Wood, J.A., Carter, S.K., Litshert, S.E., and Carr, N.B., 2017, Including broad-scale indicators in multiscale natural resource monitoring and assessment programs in the Bureau of Land Management, chap. 3 *of* Carter, S.K., Carr, N.B., Miller, K.H., and Wood, D.J.A., eds., Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, p. 15–37, accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Chapter 14. Mule Deer

Introduction

The mule deer (*Odocoileus hemionus*) is widely distributed throughout western North America. Their range is limited largely by environmental factors including prolonged cold winters, deep snow, and drought (Wallmo, 1981). In the SGP, mule deer are primarily distributed across the western portions of the ecoregion, although they occur elsewhere in suitable habitat. Across their range, mule deer use a variety of vegetation types provided there is ample forage, cover (thermal and escape), and water resources (Wallmo, 1981; Anderson and Wallmo, 1984; Mule Deer Working Group, 2004). In the SGP, vegetation types used by mule deer may include shrublands, savannas, and woodlands; riparian shrublands and woodlands; desert scrub; and hayfields, pasturelands, and small-grain fields (Relyea and others, 2000; Armstrong and others, 2011). Mule deer typically browse woody plants, but they may also include forbs and grasses in their diet (Gill and others, 1983).

A crucial factor controlling mule deer survivorship and fecundity is nutritional status, which depends on forage quantity and quality (Julander and others, 1961; Anderson and Wallmo, 1984; Bishop and others, 2009). Winter mortality resulting from starvation among fawns and older adults can be especially high (Bender and others, 2007). Throughout much of the West, mule deer populations are generally believed to be declining, in large part because of declining fawn-to-doe ratios (Carpenter, 1998). Declines are thought to be related to poor habitat conditions (Bishop and others, 2009; Bergman and others, 2014) because habitat quality has declined across much of their range, the result of altered fire regimes and associated plant successional changes, invasive vegetation, overgrazing, energy development, and direct habitat loss from urbanization (Watkins and others, 2007). Other causes for declining populations are thought to include density-dependent factors (White and Bartmann, 1998), fetal mortality or mortality of neonates at birth (Pojar and Bowden, 2004), disease, or predators (Bishop and others, 2009).

Mule deer movements can be influenced by variation in snow cover, rainfall, drought, and habitat productivity (Wallmo, 1981; Sawyer and others, 2009b). Snow inhibits mule deer movements, and deep snow can force them into other areas with less snow (Anderson and Wallmo, 1984). Similarly, even where winters are mild, seasonal drought and rainfall patterns influence migratory movements (Anderson and Wallmo, 1984).

Mule deer populations can be negatively affected by habitat loss and fragmentation resulting from development, altered fire regimes, introduced diseases, altered predator communities, and hunting pressure (Mule Deer Working Group, 2004). The effects of high densities of oil and gas development on mule deer have been extensively studied outside of the SGP in northwest Colorado and western Wyoming; both wintering and migrating mule deer are negatively affected by high levels of oil and gas development and associated disturbance, and the negative effects persist over time (Sawyer and others, 2009a, b; Lendrum and others, 2013; Northrup and others, 2016; Johnson and others, 2017; Lendrum and others, 2017; Sawyer and others, 2017). Residential development and roads also have negative effects on mule deer populations (Romin and Bissonette, 2013; Rost and Bailey, 2013; Simpson and others, 2016; Johnson and others, 2017). Another concern for mule deer is disturbance from off-highway vehicle traffic, which is increasing rapidly throughout the West (Ouren and others, 2007).

Agriculture can have both positive and negative consequences for mule deer. Conversion to croplands can lead to habitat loss, especially thermal cover, but mule deer may forage in agricultural lands (Garrott and others, 2013). Croplands and pastures can be a valuable food source in early spring for pregnant females and can also influence the timing of seasonal deer movements (Garrott and others, 2013). Chronic, heavy grazing by livestock and high densities of other wild ungulates have been reported to reduce forage and alter vegetation communities for mule deer in many parts of the species' range (Julander and others, 1961; Vavra and others, 2007; Clements and Young, 2013; Loft and others, 2013). Additional background information on mule deer can be found in the SGP pre-assessment report (Melcher, 2015).

Rapid Ecoregional Assessment Components Evaluated for the Mule Deer

The key ecological attributes and change agents addressed by core management questions for mule deer habitat include amount and distribution, landscape structure, and development (tables 14–1 and 14–2). Fire occurrence and climate change were evaluated for the entire SGP (see Reese and others, 2017, chap. 3). Overall landscape-level ranking variables are summarized in table 14–3. The core and integrated management questions are listed in table 14–4.

Mule deer. Photograph by Tom Koerner, U.S. Fish and Wildlife Service (Creative Commons Attribution 2.0 Generic).

Table 14–1. Key ecological attributes and associated indicators used to address core management questions for mule deer for the Southern Great Plains Rapid Ecoregional Assessment.

Attributes	Variables	Indicators[1]
Amount and distribution	Total area	Baseline habitat (year round)[2]
Landscape structure	Patch size	Patch sizes for baseline habitat
Landscape dynamics	Fire occurrence	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]Baseline habitat was estimated using data from Luce and others (2005). See chapter 2, "Methods Overview for Species," for methods and datasets used.

Table 14–2. Anthropogenic change agents and associated indicators used to address core management questions for mule deer for the Southern Great Plains Rapid Ecoregional Assessment.

[km, kilometer; mi, mile]

Attributes	Variables	Indicators[1]
Development	Terrestrial development index (TDI)	Percentage of baseline habitat in seven development classes based on a 2.5-km (1.55-mi) moving window
	Index of fragmentation	Patch sizes for relatively undeveloped[2] habitat
Climate change	Projected temperature and precipitation	See Reese and others (2017, chap. 3)

[1]See Reese and others (2017, chap. 2 and appendix A) for methods and datasets used to address core management questions.

[2]Terrestrial development index score less than or equal to 2 percent.

Table 14–3. Landscape-level variables used to address the integrated management question for mule deer. Ranks for landscape-level habitat area and development were combined into an overall landscape-level rank for the Southern Great Plains Rapid Ecoregional Assessment.

[>, greater than; km, kilometer; mi, mile]

Landscape-level variables[1]	Description	Relative rank[2]		
		Lowest	Medium	Highest
Area	Percentage of baseline habitat within a 5-km-radius (3.11-mi) moving window	>0–78.1	>78.1–93.7	>93.7
Development	Mean terrestrial development index (TDI) score for baseline habitat within a 5km-radius (3.11-mi) moving window	0–2	>2–10	>10

[1]See Reese and others (2017, chap. 2) for methods and datasets used to address integrated management questions.

[2]Ranking breakpoints for area of baseline habitat were determined from equal subsets of the data. Ranking breakpoints for terrestrial development index scores were standardized for all terrestrial conservation elements.

Table 14–4. Management questions addressed for mule deer for the Southern Great Plains Rapid Ecoregional Assessment.

Core management questions[1]	Results
What is the distribution of baseline habitat for mule deer?	Figure 14−1
Where does existing development pose the greatest threat to mule deer habitat, and where are the large, relatively undeveloped areas?	Figures 14−2 and 14−3
How has development fragmented mule deer habitat?	Figures 14−4 and 14−5
Integrated management question[2]	**Results**
Where is mule deer habitat with the highest overall landscape-level rank?	Figure 14−6

[1]See Reese and others (2017, chap. 11) for management questions that could not be addressed.

[2]See table 14−3.

Management Questions and Results

What is the distribution of baseline habitat for mule deer (fig. 14–1)?

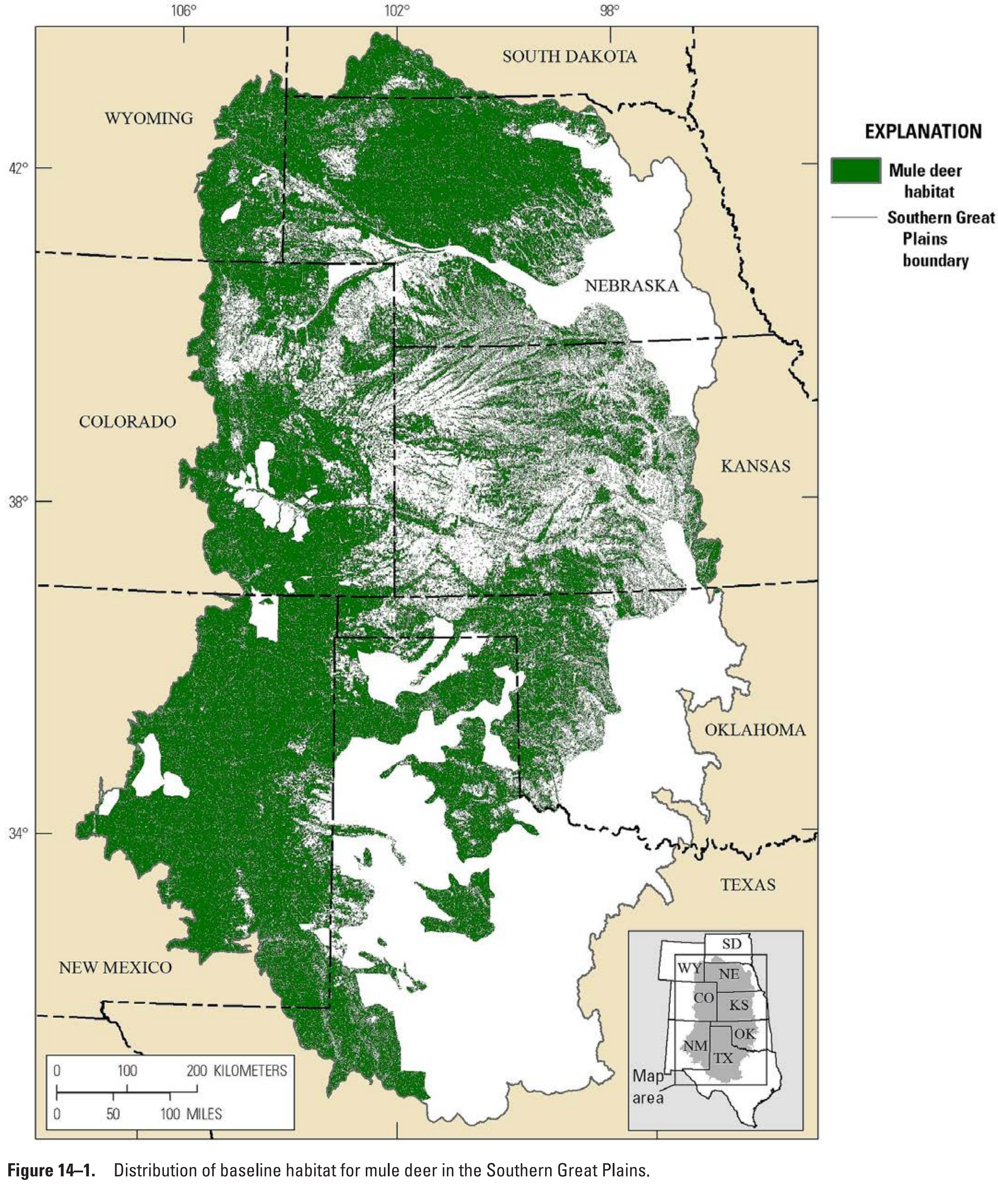


Figure 14–1. Distribution of baseline habitat for mule deer in the Southern Great Plains.

Where does existing development pose the greatest threat to mule deer habitat, and where are the large, relatively undeveloped areas (figs. 14–2 and 14–3)?

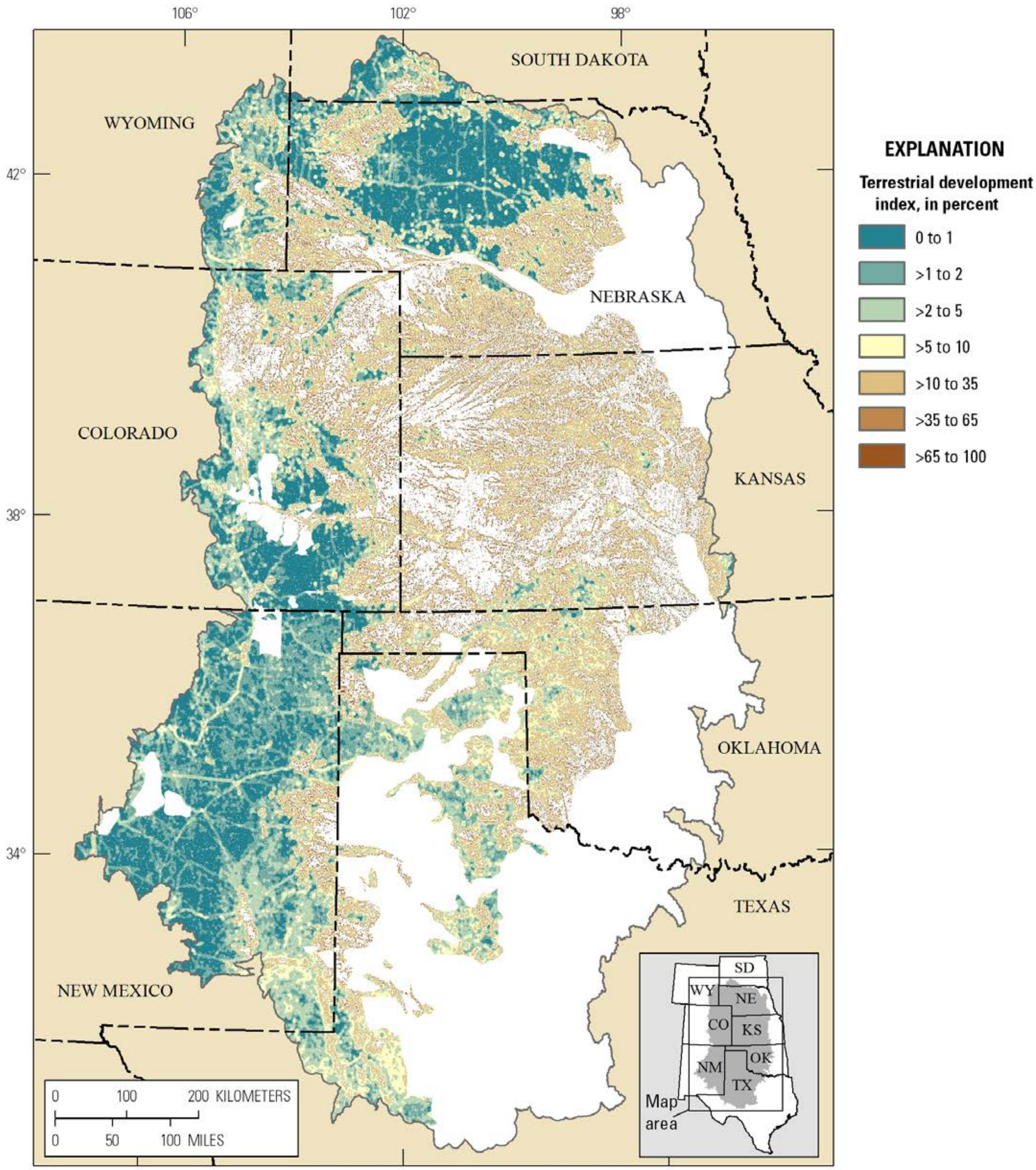


Figure 14–2. Terrestrial development index for mule deer baseline habitat in the Southern Great Plains.

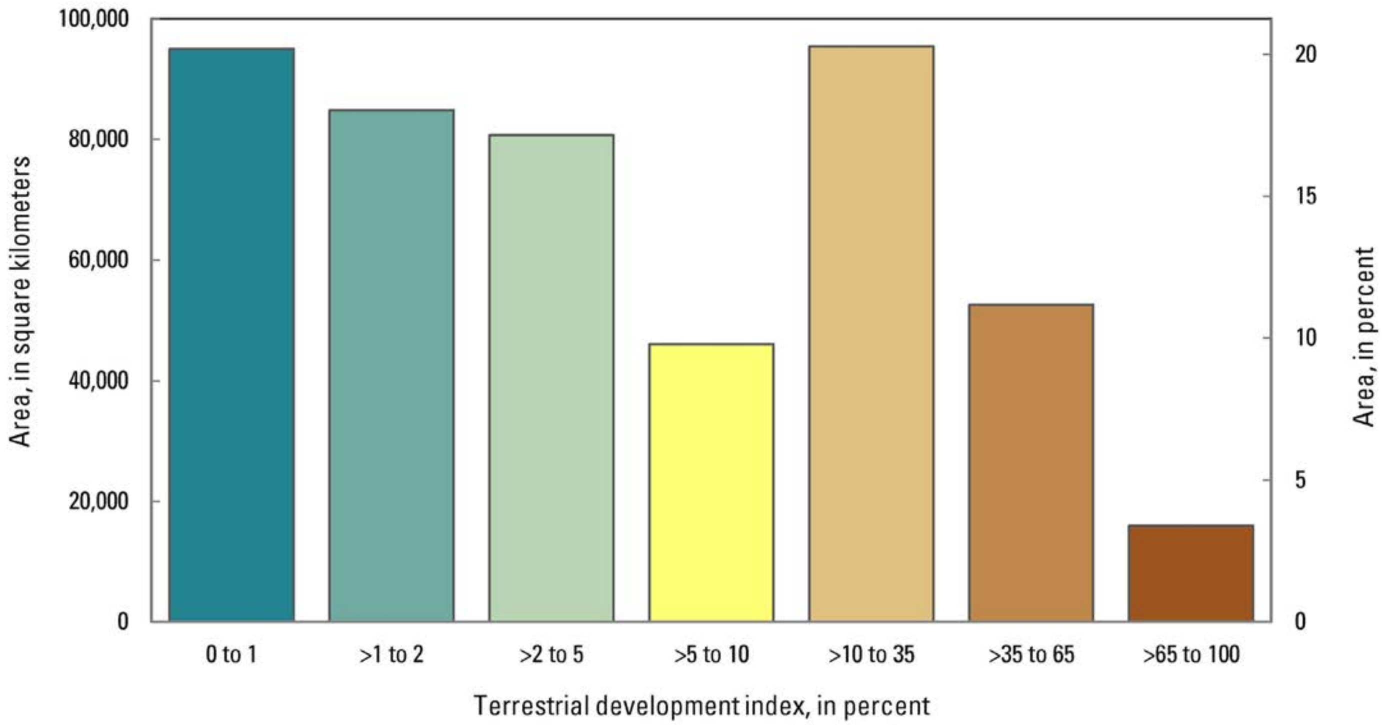


Figure 14–3. Area of mule deer baseline habitat by terrestrial development index class in the Southern Great Plains.

How has development fragmented mule deer habitat (figs. 14–4 and 14–5)?

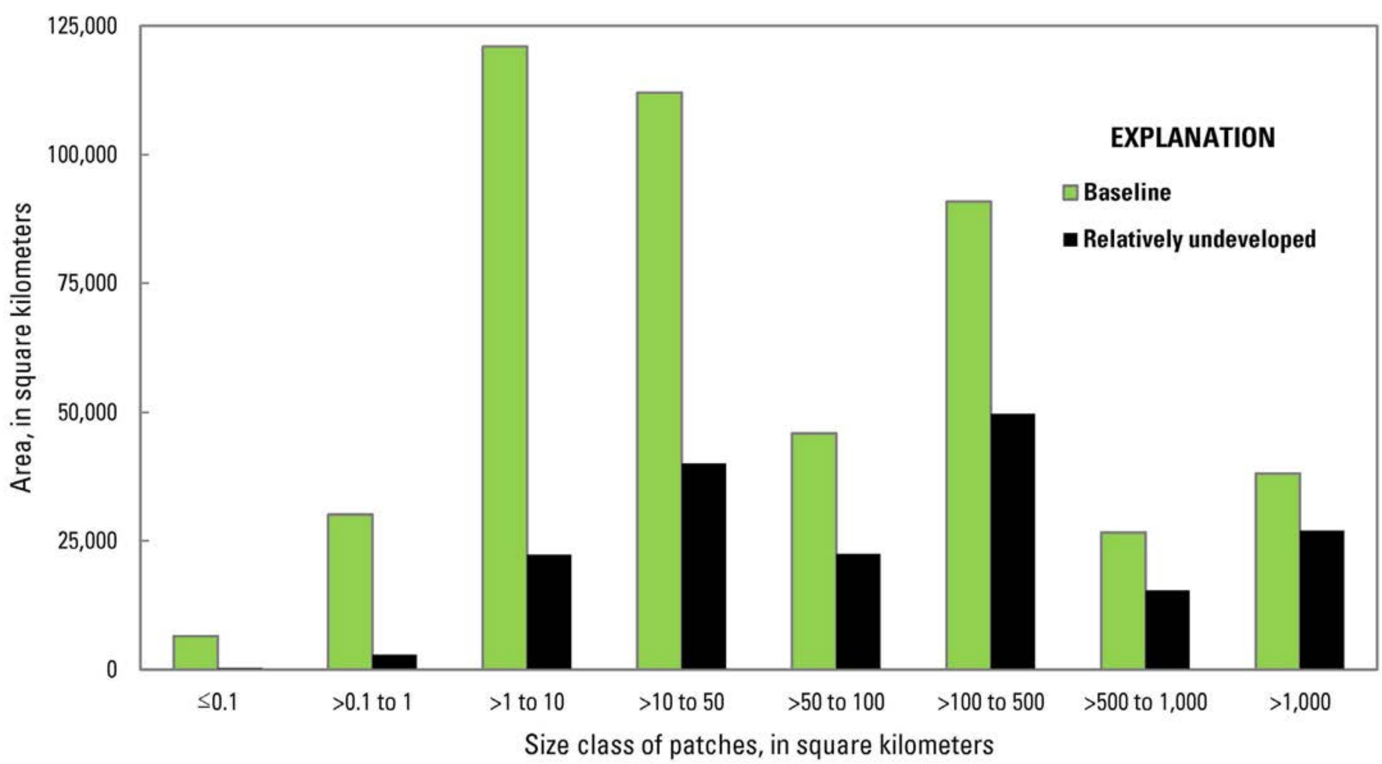


Figure 14–4. Area of mule deer habitat in the Southern Great Plains as a function of patch size for baseline and relatively undeveloped conditions (terrestrial development index score ≤2 percent).

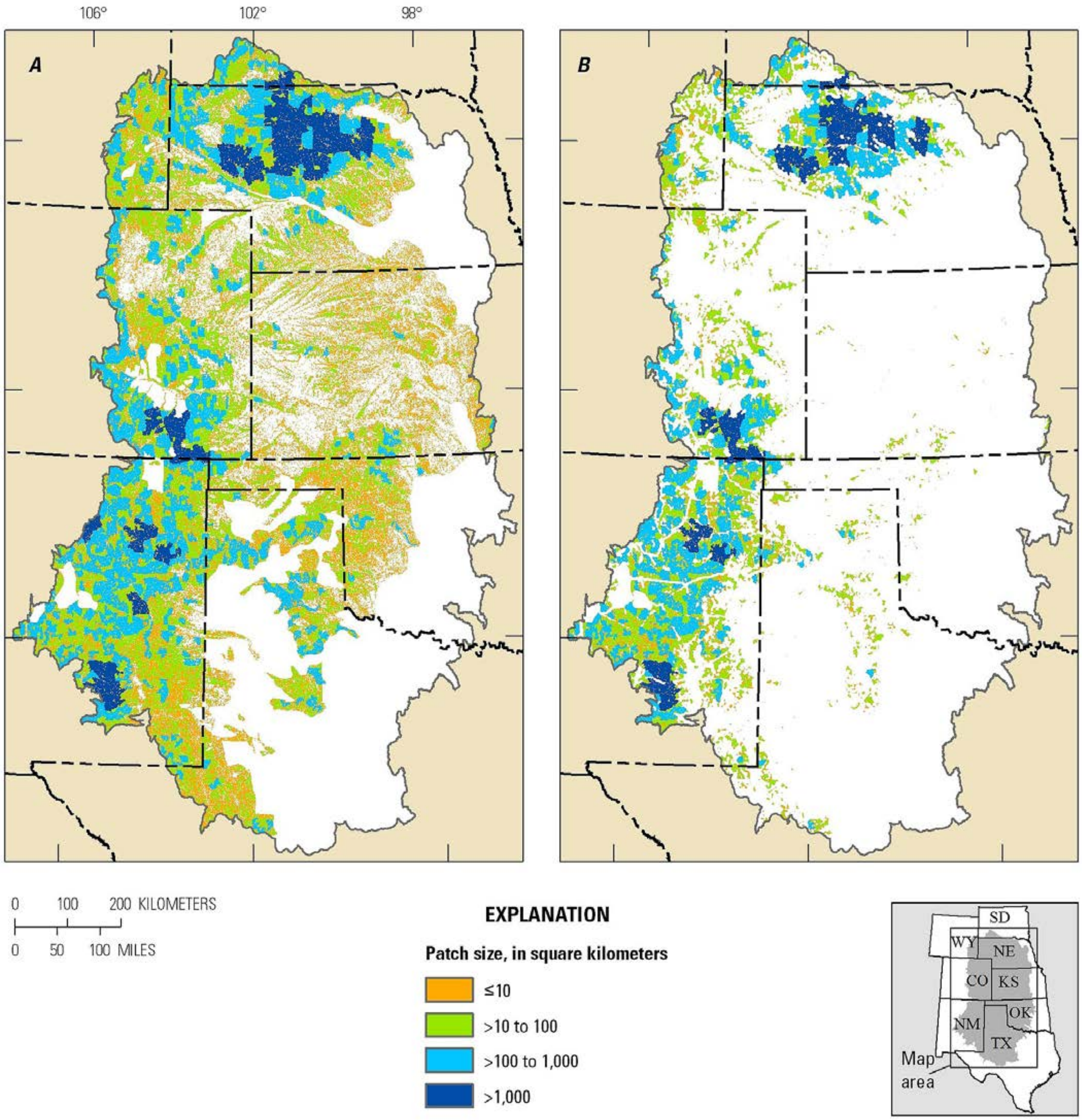


Figure 14–5. Patch size of mule deer habitat in the Southern Great Plains. *A,* Baseline habitat. *B,* Relatively undeveloped habitat (terrestrial development index score ≤2 percent).

Where is mule deer habitat with the highest overall landscape-level rank (fig. 14–6)?

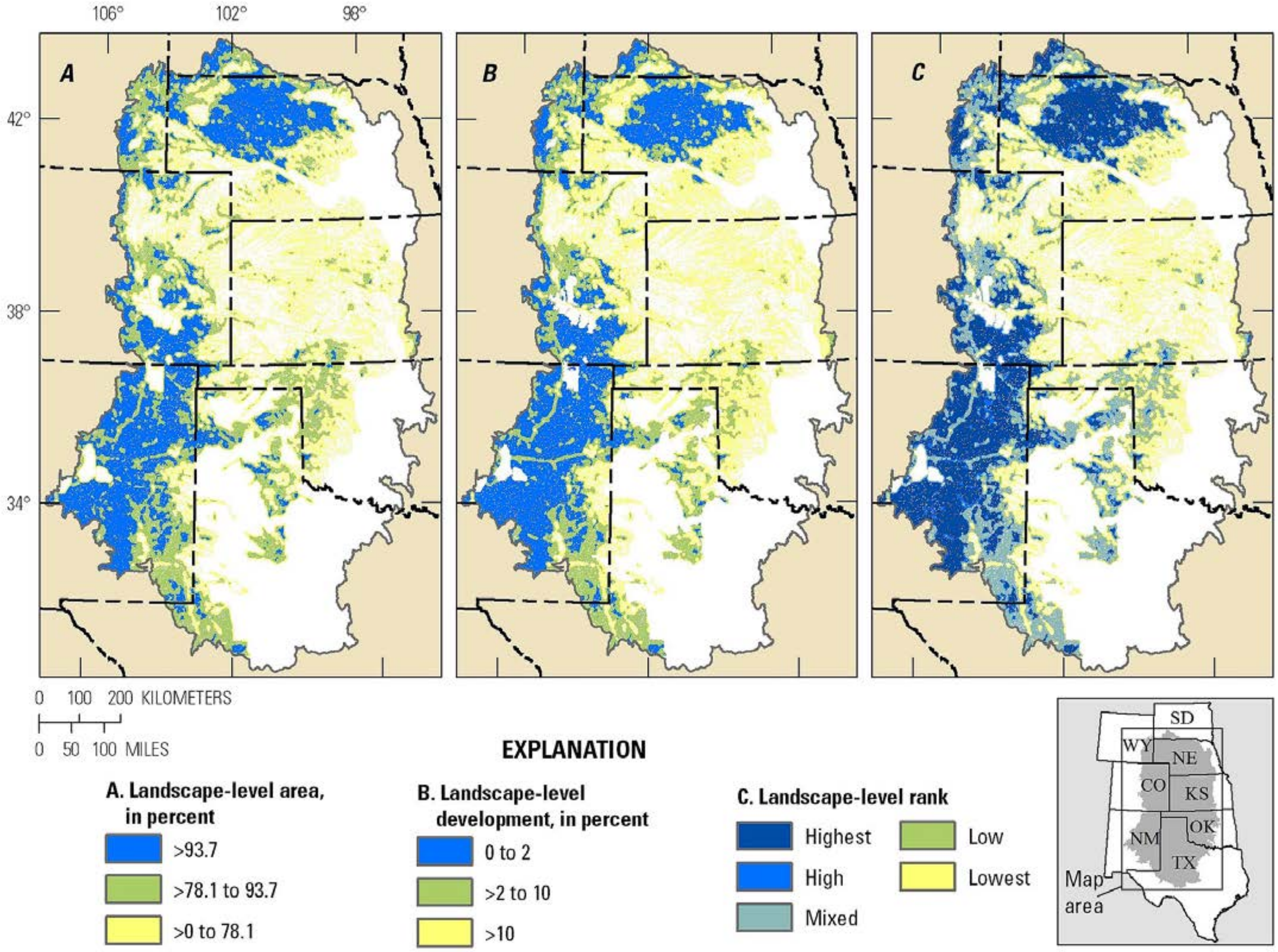


Figure 14–6. Landscape-level summaries for mule deer habitat in the Southern Great Plains. Overall landscape-level rank (*C*) is derived from (*A*) landscape-level habitat area and (*B*) landscape-level development, summarized by a 5-kilometer-radius (3.11-mile) moving window (see table 14–3). Highest overall landscape-level rank corresponds to the largest landscape-level area and the lowest landscape-level development. Lowest overall landscape-level rank corresponds to the smallest landscape-level area and highest landscape-level development. Landscape-level ranks are not intended as standalone summaries and are best interpreted in conjunction with the geospatial datasets used to address core management questions.

Summary

- Baseline habitat for mule deer is distributed widely throughout the SGP (fig. 14–1). Much of the habitat is used year round, except for isolated seasonal-use areas along the western side of the SGP, primarily within the project area buffer. There are approximately 471,000 km^2 (181,854 mi^2) of baseline habitat in the SGP.
- Mule deer habitat with the lowest development is located predominantly in the Nebraska Sand Hills, southeastern Colorado, and eastern New Mexico (fig. 14–2). Approximately 38 percent of its habitat is relatively undeveloped (TDI score ≤2 percent), and 17 percent has low development (TDI scores 2–5 percent) (fig. 14–3). Almost 15 percent of its habitat has very high development (TDI scores >35 percent).
- Fragmentation of mule deer habitat is the greatest in Kansas (figs. 14–4 and 14–5*A*). Nearly one-third of baseline habitat occurs in patches between 100 and 5,000 km^2 (39 and 1,931 mi^2) (fig. 14–4). Most of the largest relatively undeveloped habitat patches are in the Nebraska Sand Hills, eastern New Mexico, and southeastern Colorado (fig. 14–5*B*).
- The largest, most intact areas (the highest overall landscape-level rank) are in New Mexico and the Nebraska Sand Hills (fig. 14–6*C*).
- The broad-scale summaries provided by the REA are intended to be used in conjunction with local-level information on habitat conditions (Wood and others, 2017).

References Cited

Anderson, A.E., and Wallmo, O.C., 1984, Mule deer (*Odocoileus hemionus*): Mammalian Species, v. 219, p. 1–9.

Armstrong, D.M., Fitzgerald, J.P., and Meaney, C.A., 2011, Mammals of Colorado (2d ed.): Niwot, Colo., University Press of Colorado, 620 p.

Bender, L.C., Lomas, L.A., and Browning, J., 2007, Condition, survival, and cause-specific mortality of adult female mule deer in north-central New Mexico: Journal of Wildlife Management, v. 71, no. 4, p. 1118–1124.

Bergman, E.J., Bishop, C.J., Freddy, D.J., White, G.C., and Doherty, P.F., 2014, Habitat management influences overwinter survival of mule deer fawns in Colorado: Journal of Wildlife Management, v. 78, no. 3, p. 448–455.

Bishop, C.J., White, G.C., Freddy, D.J., Watkins, B.E., and Stephenson, T.R., 2009, Effect of enhanced nutrition on mule deer population rate of change: Wildlife Monographs, v. 172, no. 172, p. 1–28.

Carpenter, L.H., 1998, Deer in the West, *in* DeVos, J.C., ed., Proceedings of the 1997 deer/elk workshop, Rio Rico, Ariz., May 21–23, 1997: Phoenix, Ariz., Arizona Game and Fish Department, p. 1–10.

Clements, C.D., and Young, J.A., 2013, A viewpoint—Rangeland health and mule deer habitat: Journal of Range Management, v. 50, no. 2, p. 129–138.

Garrott, R.A., White, G.C., Bartmann, R.M., and Carpenter, L.H., 2013, Movements of female mule deer in northwest Colorado: Journal of Wildlife Management, v. 51, no. 3, p. 634–643.

Gill, R.B., Carpenter, L.H., Bartmann, R.M., Baker, D.L., and Schoonveld, G.G., 1983, Fecal analysis to estimate mule deer diets: Journal of Wildlife Management, v. 47, no. 4, p. 902–915.

Johnson, H.E., Sushinsky, J.R., Holland, A., Bergman, E.J., Balzer, T., Garner, J., and Reed, S.E., 2017, Increases in resiential and energy development are associated with reductions in recruitment for a large ungulate: Global Change Biology, v. 23, p. 578–591.

Julander, O., Robinette, W.L., and Jones, D.A., 1961, Relation of summer range condition to mule deer herd productivity: Journal of Wildlife Management, v. 25, no. 1, p. 54–60.

Lendrum, P.E., Anderson, C.R., Monteith, K.L., Jenks, J.A., and Bowyer, R.T., 2013, Migrating mule deer—Effects of anthropogenically altered landscapes: PLoS ONE, v. 8, no. 5, article e64548, 10 p., accessed July 15, 2014, at http://journals.plos.org/plosone/article?id=10.1371/journal.pone.0064548

Lendrum, P.E., Crooks, K.R., and Wittenmyer, G., 2017, Changes in circadian activity patterns of a wildlife community post high-intensity energy development: Journal of Mammology, v. 98, p. 1265–1271.

Loft, E.R., Menke, J.W., and Kie, J.G., 2013, Habitat shifts by mule deer—The influence of cattle grazing: Journal of Wildlife Management, v. 55, no. 1, p. 16–26.

Luce, A., Hurd, W., and Lowry, J.H., 2005., Mule Deer Mapping Project: Utah State University, Remote Sensing/GIS Laboratory database, accessed August 30, 2017, at http://www.gis.usu.edu/projects/mule-deer-mapping-project/.

Melcher, C.P., 2015, Mule deer, chap. 21 *of* Assal, T.J., Melcher, C.P., and Carr, N.B., eds., 2015, Southern Great Plains Rapid Ecoregional Assessment—Pre-assessment report: U.S. Geological Survey Open-File Report 2015–1003, p. 118–126, accessed July 13, 2017, at https://doi.org/10.3133/ofr20151003.

Mule Deer Working Group, 2004, North American mule deer conservation plan: Western Association of Fish and Wildlife Agencies, 17 p.

Northrup, J.M., Anderson, C.R., Jr., Hooten, M.B., and Wittemyer, G., 2016, Movement reveals scale dependence in habitat selection of a large ungulate: Ecological Applications, v. 26, p. 2746–2757.

Ouren, D.S., Haas, C., Melcher, C.P., Stewart, S.C., Ponds, P.D., Sexton, N.R., Burris, L., Fancher, T., and Bowen, Z.H., 2007, Effects of off-highway vehicles on Bureau of Land Management lands—A literature synthesis, annotated bibliographies, extensive bibliographies, and internet resources: U.S. Geological Survey Open-File Report 2007–1353, 225 p.

Pojar, T.M., and Bowden, D.C., 2004, Neonatal mule deer fawn survival in west-central Colorado: Journal of Wildlife Management, v. 68, no. 3, p. 550–560.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100.

Relyea, R.A., Lawrence, R.K., and Demarais, S., 2000, Home range of desert mule deer—Testing the body-size and habitat-productivity hypotheses: Journal of Wildlife Management, v. 64, no. 1, p. 146–153.

Romin, L.A., and Bissonette, J.A., 2013, Deer-vehicle collisions—Status of state monitoring activities and mitigation efforts: Wildlife Society Bulletin, v. 24, no. 2, p. 276–283.

Rost, G.R., and Bailey, J.A., 2013, Distribution of mule deer and elk in relation to roads: Journal of Wildlife Management, v. 43, no. 3, p. 634–641.

Sawyer, H., Kauffman, M.J., and Nielson, R.M., 2009a, Influence of well pad activity on winter habitat selection patterns of mule deer: Journal of Wildlife Management, v. 73, no. 7, p. 1052–1061.

Sawyer, H., Kauffman, M.J., Nielson, R.M., and Horne, J.S., 2009b, Identifying and prioritizing ungulate migration routes for landscape-level conservation: Ecological Applications, v. 19, p. 2016–2025.

Sawyer, H., Korfanta, N.M., Nielson, R.M., Monteith, K.L., and Strickland, D., 2017, Mule deer and energy development—Long-term trends of habituation and abundance: Global Change Biology, v. 23, p. 4521–4529.

Simpson, N.O., Stewart, K.M., Shroeder, C., Cox, M., Huebner, K., and Wasley, T., 2016, Overpasses and underpasses—Effectivenss of crossing structures for migratory ungulates: Jouranl of Wildlife Management, v. 80, p. 1370–1378.

Vavra, M., Parks, C.G., and Wisdom, M.J., 2007, Biodiversity, exotic plant species, and herbivory—The good, the bad, and the ungulate: Forest Ecology and Management, v. 246, no. 1, p. 66–72.

Wallmo, O.C., ed., 1981, Mule and black-tailed deer of North America: Lincoln, Nebr., University of Nebraska Press, 605 p.

Watkins, B.E., Bishop, C.J., Bergman, E.J., Hale, B., Wakeling, B.F., Bronson, A., Carpenter, L.H., and Lutz, D.W., 2007, Habitat guidelines for mule deer—Colorado Plateau shrubland and forest ecoregion: Cheyenne, Wyo., Western Association of Fish and Wildlife Agencies, Mule Deer Working Group, 75 p.

White, G.C., and Bartmann, R.M., 1998, Effect of density reduction on overwinter survival of free-ranging mule deer fawns: Journal of Wildlife Management, v. 62, no. 1, p. 214–225.

Wood, J.A., Carter, S.K., Litshert, S.E., and Carr, N.B., 2017, Including broad-scale indicators in multiscale natural resource monitoring and assessment programs in the Bureau of Land Management, chap. 3 *of* Carter, S.K., Carr, N.B., Miller, K.H., and Wood, D.J.A., eds., Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, p. 15–37, accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Chapter 15. Synthesis—Ecological Communities and Species

Introduction

This chapter summarizes the information presented in the Southern Great Plains Rapid Ecoregional Assessment (REA) for both ecological communities (volume I; Reese and others, 2017) and species (this volume). An overall goal of the REA is to identify relatively intact areas in the Southern Great Plains (SGP) for communities and species evaluated as a conservation elements (Assal and others, 2015; Reese and others, 2017). For the REA, we defined landscape intactness as a quantifiable estimate of naturalness measured on a gradient of anthropogenic influence and evaluated across large landscapes or ecoregions (Carter and others, 2017). At the spatial extent of ecoregions, large, relatively undeveloped areas may represent areas with high landscape intactness.

The terrestrial development index (TDI) and the aquatic development index (ADI) were used to quantify the gradient of anthropogenic influence for the Southern Great Plains REA (Reese and others, 2017, figs. 3–1 and 3–5). In this chapter, relatively undeveloped areas of terrestrial systems are defined by TDI scores ≤2 percent (fig. 15–1), and relatively undeveloped sixth-level watersheds are defined by ADI scores ≤20 (fig.15–2). Because of the differences among methodologies for TDI and ADI, the thresholds used to define relatively undeveloped areas are not directly comparable across terrestrial and aquatic systems.

In each conservation element chapter, the TDI and ADI scores were mapped and their proportional areas summarized. The proportional area of TDI and ADI scores for the Southern Great Plains overall was also summarized (Reese and others, 2017, figs. 3–2 and 3–6). To facilitate comparison among conservation elements, we compiled the distribution of TDI and ADI scores for terrestrial and aquatic systems overall and for each conservation element in this chapter (figs. 15–3 and 15–4). Development levels among terrestrial and aquatic conservation elements were compared separately because the TDI and ADI are not directly comparable to each other.

Table 15–1. Overall management questions addressed for the Southern Great Plains Rapid Ecoregional Assessment.

Core management questions[1]	Results
Where are the relatively undeveloped terrestrial areas?[2]	Figure 15−1
What is the land ownership of relatively undeveloped terrestrial areas?	Table 15−2
Where are the relatively undeveloped sixth-level watersheds?[3]	Figure 15−2
How do development levels (based on the terrestrial development index) vary by terrestrial conservation element and for the Southern Great Plains overall?	Figure 15−3
How do development levels (based on the aquatic development index) vary by aquatic conservation element and for the Southern Great Plains overall?	Figure 15−4

[1]See Reese and others (2017, chap. 2 and appendix A) for methodological details on the terrestrial and aquatic development indexes.

[2]Relatively undeveloped terrestrial areas were defined as having terrestrial development index scores less than or equal to 2 percent.

[3]Relatively undeveloped sixth-level watersheds were defined as having aquatic development index scores less than or equal to 20.

Management Questions and Results

Where are the relatively undeveloped terrestrial areas (fig. 15–1)?

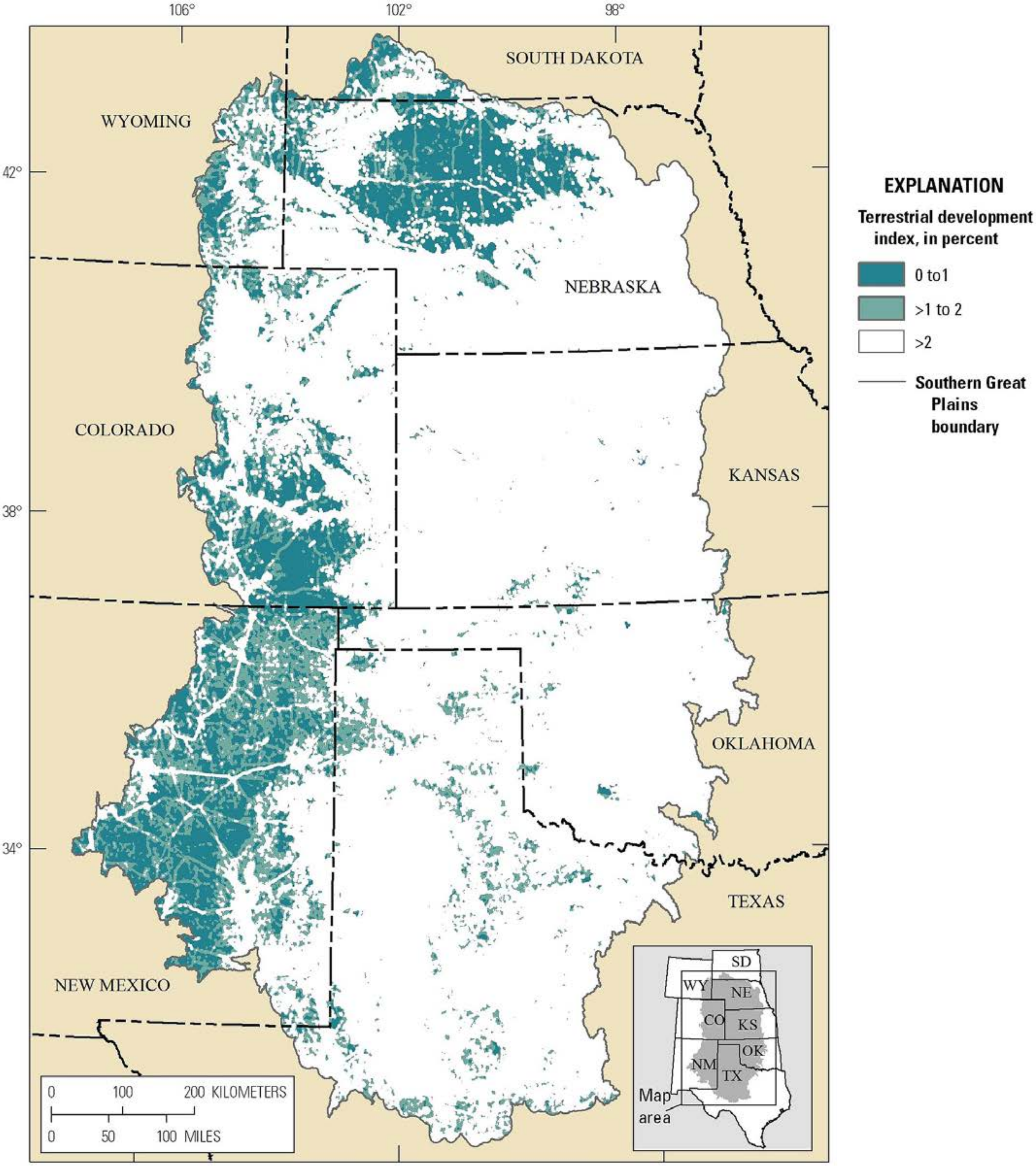


Figure 15–1. Relatively undeveloped terrestrial areas (terrestrial development index [TDI] scores ≤2 percent) in the Southern Great Plains.

What is the land ownership of relatively undeveloped terrestrial areas (table 15–2)?

Table 15–2. Area and percent of relatively undeveloped terrestrial areas by land ownership or jurisdiction in the Southern Great Plains. Relatively undeveloped areas are defined by terrestrial development index scores less than or equal to 2 percent.

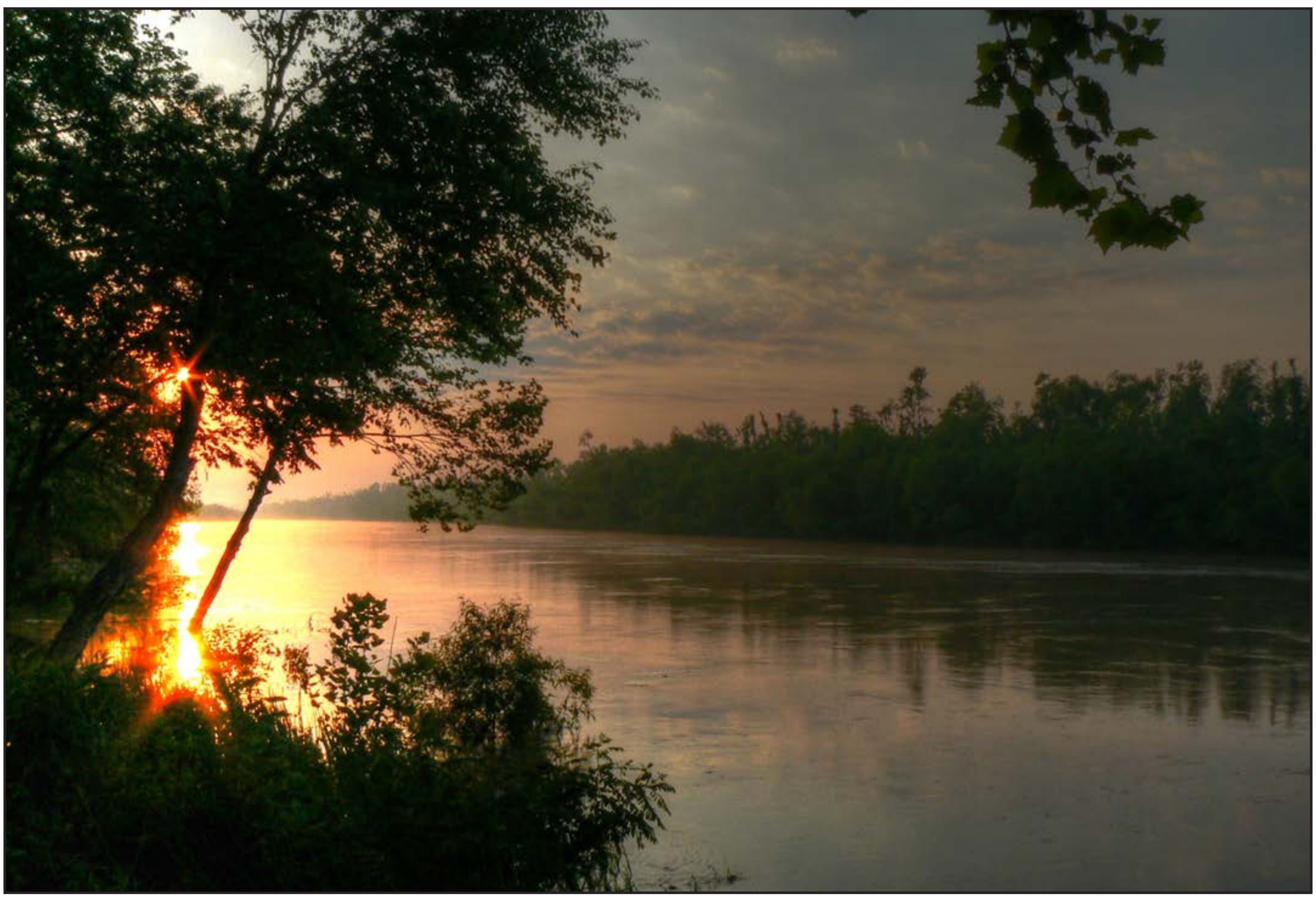
[km^2, square kilometer]

Ownership or jurisdiction[1]	Area (km^2)	Percent of relatively undeveloped areas in the Southern Great Plains
Private	153,378	75.8
State/county	18,733	9.3
Forest Service[2]	10,721	5.3
Bureau of Land Management	9,898	4.9
Tribal	4,833	2.4
Other Federal[3]	3,173	1.6
Private conservation	1,502	0.7

[1]Jurisdiction refers to lands administered by Federal, State, or county agencies.

[2]U.S. Department of Agriculture Forest Service.

[3]Federal agencies managing less than 1 percent of the total relatively undeveloped terrestrial areas: Bureau of Reclamation, Department of Agriculture, Department of Defense, Department of Energy, National Guard, National Park Service, U.S. Air Force, U.S. Army, U.S. Army Corps of Engineers, and U.S. Fish and Wildlife Service.



North Canadian River, Oklahoma. The Canadian River is the last remaining stronghold for the Arkansas River shiner. Photograph by Thomas and Dianne Jones (Creative Commons Attribution 2.0 Generic).

Where are the relatively undeveloped sixth-level watersheds (fig. 15–2)?

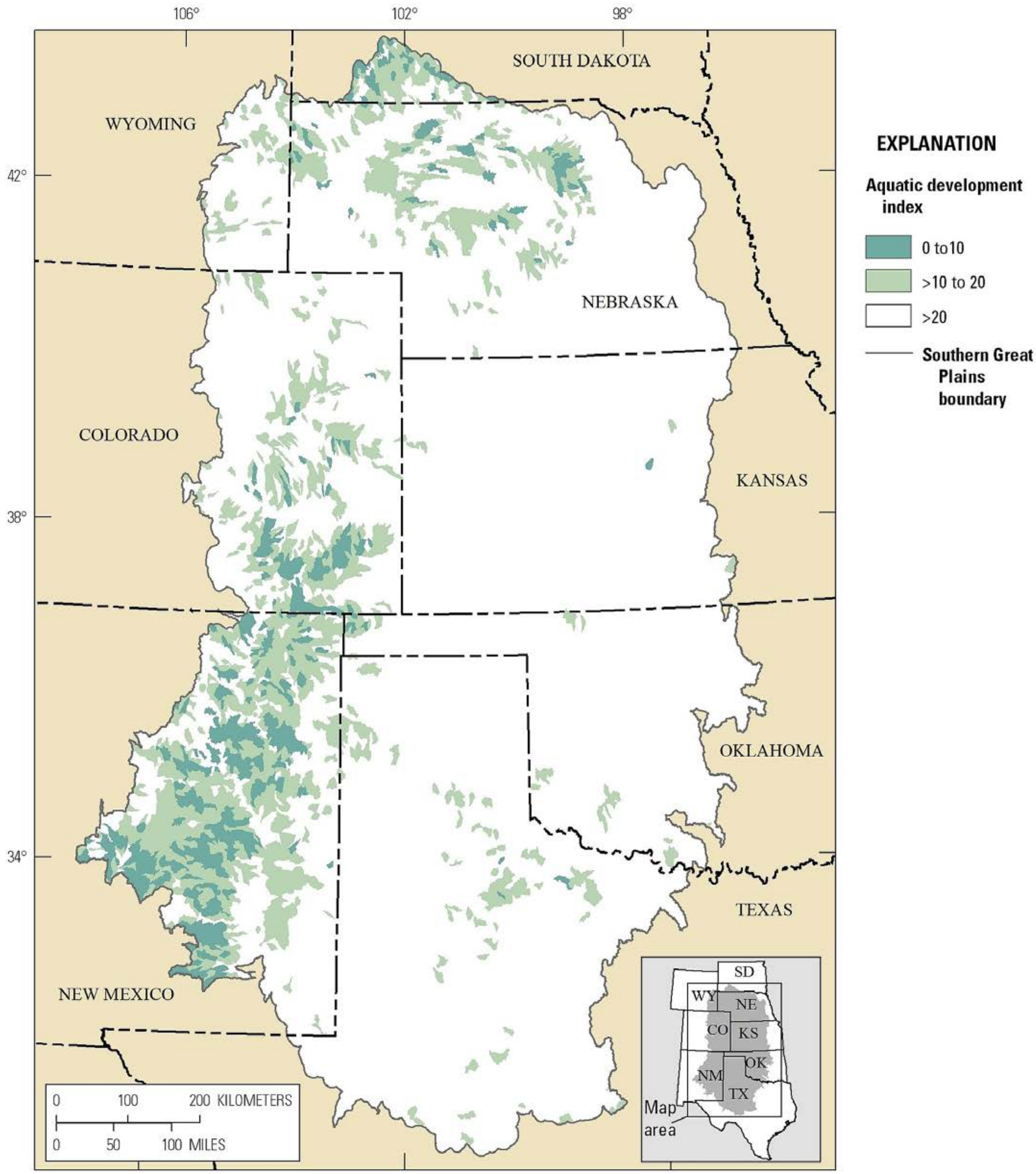


Figure 15–2. Relatively undeveloped aquatic areas (aquatic development index scores ≤20), summarized by sixth-level watershed, in the Southern Great Plains.

How do development levels vary by terrestrial conservation element and for the Southern Great Plains overall (fig. 15–3)?

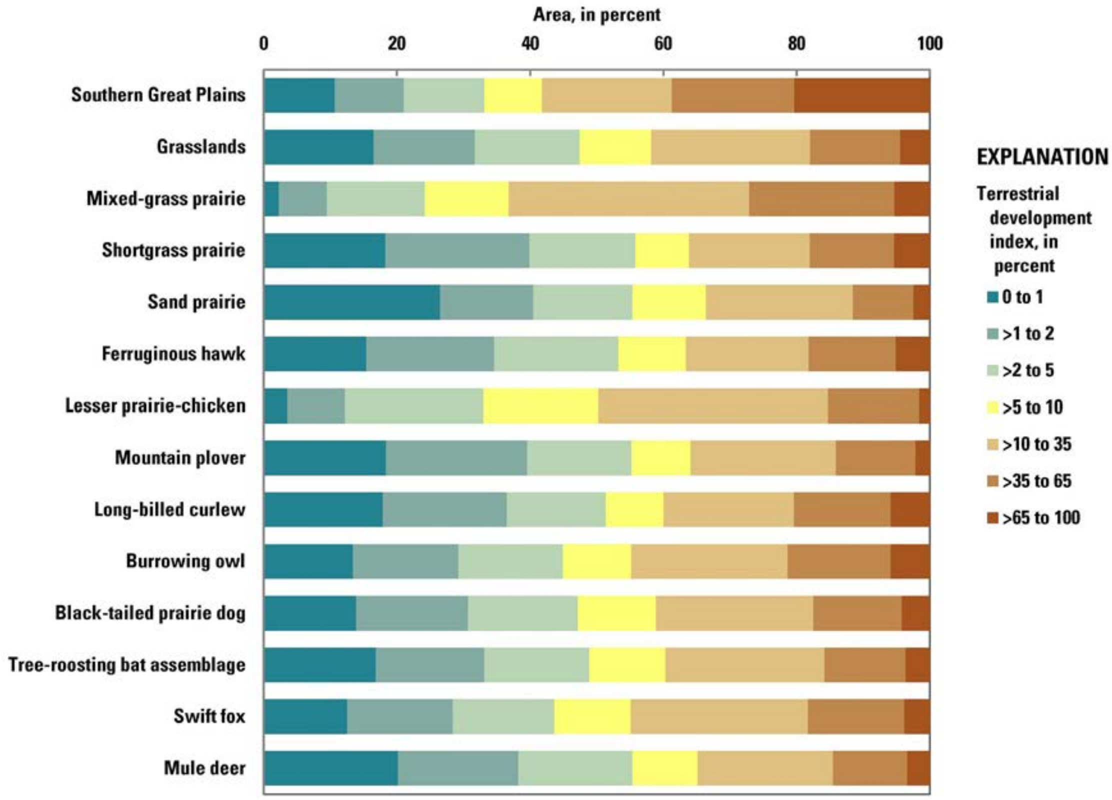


Figure 15–3. Percentage of area by terrestrial development index class for each conservation element and for the Southern Great Plains overall.

How do development levels vary by aquatic conservation element and for the Southern Great Plains overall (fig. 15–4)?

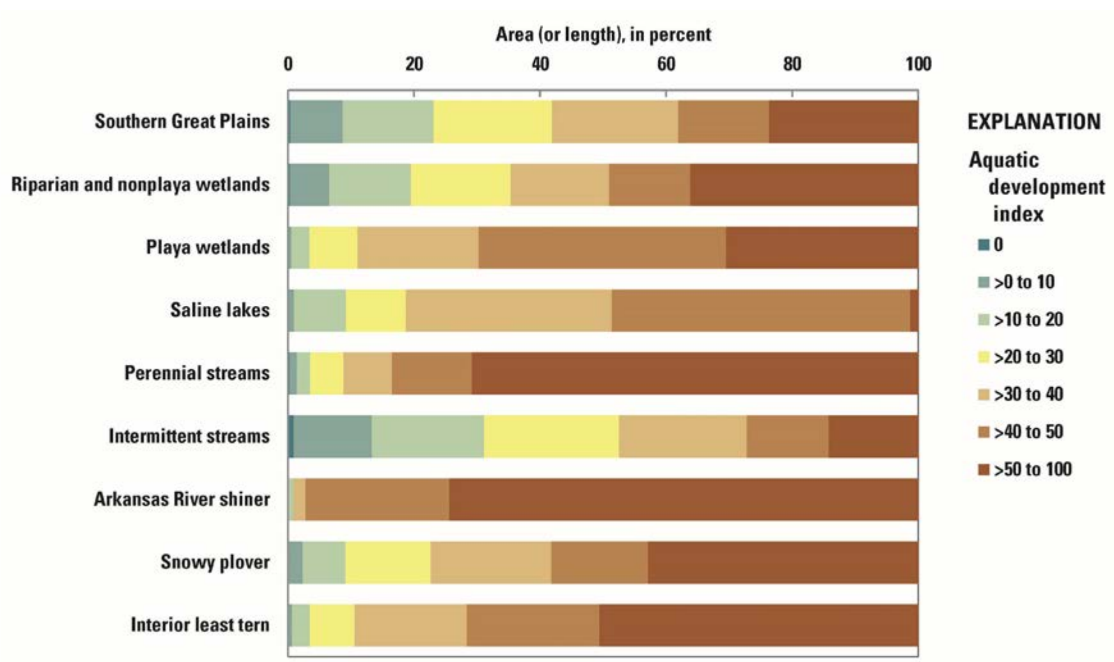


Figure 15–4. Percentage of area by aquatic development index class for each conservation element and for the Southern Great Plains overall. Area applies to most conservation elements; length applies to perennial streams, intermittent streams, and the Arkansas River shiner.

Summary

- Large, relatively undeveloped terrestrial areas (TDI scores ≤2) with potentially high landscape intactness occur mostly in the western and north-central portions of the SGP in areas that are predominantly sand or shortgrass prairie (fig. 15–1). Approximately 40 percent of sand and shortgrass prairies is relatively undeveloped, but less than 10 percent of mixed-grass prairie is relatively undeveloped (fig. 15–3).
- Relatively undeveloped terrestrial areas cover 21 percent of the project area (fig. 15–3), 82 percent of which occurs within patches greater than 1,000 square kilometers (386 square miles).
- Federal lands, primarily those managed by the U.S. Forest Service and the Bureau of Land Management, account for nearly 14 percent of relatively undeveloped terrestrial areas within the SGP. Approximately 76 percent of relatively undeveloped lands are in private ownership (table 15–2).
- Most terrestrial species have approximately one-third of their distribution classified as relatively undeveloped; for the lesser prairie-chicken, however, only 12 percent of its remaining habitat is relatively undeveloped (fig. 15–3).
- Relatively undeveloped watersheds are concentrated in the southwestern and northern portions of the SGP (fig. 15–2), where intermittent streams are more prevalent (see Reese and others, 2017, figs. 10–1 and 10–4).
- Development levels are very high for all species associated with aquatic environments. In particular, habitat for the Arkansas River shiner has very high development, with less than 1 percent relatively undeveloped. Relatively undeveloped habitat for the snowy plover and the interior least tern accounts for less than 10 percent of their habitat in the SGP (fig. 15–4).
- Ecoregion-level summaries of landscape intactness, such as these, can provide a broad-scale context for the evaluation of the condition of species habitats and ecological communities based on local information.

References Cited

Assal, T.J., Melcher, C.P., and Carr, N.B., eds., 2015, Southern Great Plains Rapid Ecoregional Assessment—Pre-assessment report: U.S. Geological Survey Open-File Report 2015–1003, 284 p., accessed September 2015 at https://doi.org/10.3133/ofr20151003.

Carter, S.K., Carr, N.B., Miller, K.H., and Wood, D.J.A., eds., 2017, Multiscale guidance and tools for implementing a landscape approach to resource management in the Bureau of Land Management: U.S. Geological Survey Open-File Report 2016–1207, 79 p., accessed January 2017 at https://doi.org/10.3133/ofr20161207.

Reese, G.C., Burris, L., Carr, N.B., Leinwand, I.I.F., and Melcher, C.P., 2017, Southern Great Plains Rapid Ecoregional Assessment—Volume I. Ecological communities: U.S. Geological Survey Open-File Report 2017–1100, 126 p., https://doi.org/10.3133/ofr20171100.

USGS

Reese and others—**Southern Great Plains Rapid Ecoregional Assessment—Volume II. Species and Assemblages**—Open-File Report 2018–1109

ISSN 2331-1258 (online)
https://doi.org/10.3133/ofr20181109

U.S. Department of the Interior
Bureau of Land Management

Data Request Method

Rapid Ecoregional Assessments (REAs)—National Operations Center, CO

Individual REA data layers and some other products are still available but are no longer being published.

If you would like to obtain more information, including data and model zip files* (containing Esri ModelBuilder files for ArcGIS 10.x and relevant Python scripts), please email **BLM_OC_REA_Data_Portal_Feedback_Team@blm.gov**.
*Note that a few models require software that BLM does not provide such as R, Maxent, and TauDEM.

Models associated with individual REAs may require data links to be updated to function properly. REA reports, technical appendices, and model overviews (for some REAs) contain detailed information to determine what products are available and what datasets are necessary to run a certain model.

Please include the report name and any specific data information that you can provide with your request.

Other BLM data can be found on the Geospatial Business Platform Hub (https://gbp-blm-egis.hub.arcgis.com).