In cooperation with Bureau of Land Management

Southern Great Plains Rapid Ecoregional Assessment—Pre-Assessment Report

Open-File Report 2015–1003

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

Front cover photographs: Shortgrass prairie in Comanche National Grassland, Colorado. (Photo courtesy of the U.S Forest Service) (Inset) Swift fox. (Photo courtesy of the U.S. Fish and Wildlife Service) Back cover photograph: Black-tailed prairie dogs. (Photo by Dean Biggins, U.S. Geological Survey)

Southern Great Plains Rapid Ecoregional Assessment—Pre-Assessment Report

Edited by Timothy J. Assal, Cynthia P. Melcher, and Natasha B. Carr

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Conversion Factors

SI to Inch/Pound

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)
millimeter (mm)	0.03937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
meter (m)	1.094	yard (yd)
	Area	
square meter (m^2)	0.0002471	acre
hectare (ha)	2.471	acre
square kilometer (km^2)	247.1	acre
square meter (m^2)	10.76	square foot (ft^2)
hectare (ha)	0.003861	square mile (mi^2)
square kilometer (km^2)	0.3861	square mile (mi^2)
	Flow rate	
cubic meter per second (m^3/s)	70.07	acre-foot per day (acre-ft/d)
	Mass	
kilogram (kg)	2.205	pound avoirdupois (lb)

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

°F=(1.8×°C)+32.

Abbreviations

AMT	Assessment Management Team
ARS	Arkansas River shiner
BLM	Bureau of Land Management
BTPD	black-tailed prairie dog
CA	Change Agent
CE	Conservation Element
CRP	Conservation Reserve Program
GPLCC	Great Plains Landscape Conservation Cooperative
ILT	interior least tern
LBC	long-billed curlew
LPC	lesser prairie-chicken
MQ	Management Question
NWR	National Wildlife Refuge
REA	Rapid Ecoregional Assessment
SGAR	second-generation anticoagulant rodenticide
SGP	Southern Great Plains
SMGP	southern mixed-grass prairie
USGS	U.S. Geological Survey
WNS	white-nose syndrome
WNv	West Nile virus

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Executive Summary

By Timothy J. Assal and Natasha B. Carr

The purpose of the Pre-Assessment Report for the Southern Great Plains Rapid Ecoregional Assessment (REA) is to document the selection process for and final list of Conservation Elements, Change Agents, and Management Questions developed during Phase I. The overall goal of the REAs being conducted for the Bureau of Land Management (BLM) is to provide information that supports regional planning and analysis for the management of ecological resources. The REA provides an assessment of baseline ecological conditions, an evaluation of current risks from drivers of ecosystem change, and a predictive capacity for evaluating future risks. The REA also may be used for identifying priority areas for conservation or restoration and for assessing the cumulative effects of a variety of land uses. There are several components of the REAs. Management Questions, developed by the BLM and partners for the ecoregion, identify the information needed for addressing land-management responsibilities. Conservation Elements represent regionally significant terrestrial and aquatic species and communities that are to be conserved and (or) restored. For each Conservation Element, key ecological attributes will be evaluated to determine the status of each species and community. The REA also will evaluate major drivers of ecosystem change, or Change Agents, currently affecting or likely to affect the status of Conservation Elements in the future. The relationships between Change Agents and key ecological attributes will be summarized using conceptual models. The REA process is a two-phase process. Phase I (pre-assessment) includes developing and finalizing the lists of priority Management Questions, Conservation Elements, and Change Agents, culminating in the REA Pre-Assessment Report.

Chapter 1 provides an overview of the REA process. Chapter 2 describes the biophysical and anthropogenic features of the Southern Great Plains, and Chapter 3 explains the process used to identify Conservation Elements, Change Agents and Management Questionss. The remaining chapters each feature one of 19 Conservation Elements—6 ecological communities and 13 species (including 2 species assemblages)—to be addressed in Phase II. For each Conservation Element, we will address the four primary Change Agents—development, fire, invasive species, and climate change—required for the REA. In addition, we will evaluate insect pests and disease for particular Conservation Elements. Development includes effects related to energy and infrastructure, agricultural activities, and other human activities, including urbanization and recreation.

An overview on the ecology and management issues for each Conservation Element is provided, including distribution and ecology, landscape structure and dynamics, and associated species of management concern affiliated with each Conservation Element. For each Conservation Element, effects of the Change Agents are described. An overview of potential key ecological attributes and potential Change Agents are summarized by conceptual models and tables. The tables provide an organizational framework and background information for evaluating the key ecological attributes and Change Agents in Phase II.

Section I. Southern Great Plains Ecoregional Assessment—Background and Overview

Chapter 1. Introduction—Bureau of Land Management Rapid Ecoregional Assessments

By Natasha B. Carr, Cynthia P. Melcher, and Zachary H. Bowen

Purpose of the Rapid Ecoregional Assessment

The overall goal of the Rapid Ecoregional Assessments (REAs) undertaken by the Bureau of Land Management (BLM) is to provide information that facilitates development of ecoregion-based conservation strategies across jurisdictional boundaries and to facilitate planning and analysis for the management of ecological resources. The REA provides an assessment of ecological conditions, an evaluation of risk from Change Agents (CAs), a predictive capacity for evaluating future risks from CAs, baseline information for long-term monitoring of ecoregional conditions, and guidance for adaptation and mitigation planning in response to climate change. The REA also may be used for identifying and prioritizing potential conservation areas or native plant and animal communities that need restoration, for assessing cumulative effects as required by the National Environmental Policy Act, and for informing landscape-scale planning and decision-making for all resources and uses of public lands. Overall, the REA provides a vehicle for creating stronger and more effective, efficient collaboration and cooperation among all parties interested in regional land and resource management.

The BLM established the overall process and required components for the REA, which includes Management Questions (MQs), Conservation Elements (CEs), which are ecological resources of concern, and CAs for each ecoregion. Within these overall guidelines, however, there is flexibility to tailor the REA to the specific information priorities for a given ecoregion. We first provide an overview of the general REA guidelines. Sections that follow discuss specific details pertaining to the Wyoming Basin REA and which we adapted for the Southern Great Plains (SGP) REA.

Overview of the Rapid Ecoregional Assessment Process

An REA entails a two-phase process (table 1). In Phase I (pre-assessment), the lists of priority MQs, CEs, and CAs are developed and finalized. Phase I also includes the development of key ecological attributes and CAs. Phase II (assessment) comprises three tasks related to the compilation, analyses, and documentation of datasets to address MQs and complete the ecoregional assessment.

Table 1. Overview of Phase I (pre-assessment) and Phase II (assessment) tasks for the Rapid Ecoregional Assessment (REA).

Phase	Task	Task description
I. Pre-assessment	1	Refine Management Questions; select Conservation Elements and Change Agents; develop overall ecosystem conceptual model.
	2	Identify key ecological attributes and indicators for Conservation Elements that will establish the framework for Phase II analysis.
	3	Develop conceptual models for Conservation Elements.
	4	Prepare Pre-Assessment Report.
II. Assessment	5	Compile and generate source datasets.
	6	Conduct analyses and generate findings.
	7	Prepare final REA report and documents.

Rapid Ecoregional Assessment Products

The purpose of the Pre-Assessment Report is to document the selection process for, and final list of, MQs, CEs, and CAs developed during Phase I (although the lists may be updated based on analysis conducted during Phase II). The report also presents the overall assessment framework that will be used to determine the status of CEs and answer MQs. The assessment framework describes the overall approach and includes templates for conceptual models and formats for organizing and documenting the data sources and methods, which will be refined and applied in Phase II.

At the end of Phase II, a final REA report is published. The final REA report expands upon the Pre-Assessment Report to document specific methods, results, and conclusions relating to MQs, CEs, and CAs. All source and derived datasets are to be provided to the BLM following the guidelines established in the REA Statement of Work.

Components of the Rapid Ecoregional Assessment

The major components of an REA include MQs, CEs, and CAs (table 2). Also crucial to the REA process is the development of conceptual models, which guide the selection of key ecological attributes to be evaluated in the REA. These components are described in detail in the sections that follow.

Management Questions

Management Questions (table 2) identify the information needed for addressing land-management responsibilities, including land-use planning, developing best-management practices, authorizing uses, and establishing priorities for conservation and restoration. The MQs help to focus the REA process and ensure that the most relevant datasets are compiled, analyzed, and summarized. The MQs may pertain to ecological resources and CAs. Ecological resources include native terrestrial and aquatic species and communities of regional significance. The CAs are ecological processes or human activities that influence the current status of resources and may pose future risks to those resources.

Table 2. Major components of the Rapid Ecoregional Assessment.

[From Carr and others (2013) and adapted from Parrish and others (2003)]

Term	Definition or description
Management Questions	Priority information needs regarding ecological resources and Change Agents. Management Questions address land-management responsibilities and will guide the assessment process and ensure that the most relevant datasets are compiled, analyzed, and summarized.
Conservation Elements	A limited number of species, species assemblages, and ecological communities or ecosystems that represent critical components of ecosystems. *Terrestrial and aquatic ecological communities* that comprise most of the ecoregion and are presumed to represent the habitat requirements of most plant and animal species of the ecoregion. *Regionally significant species or species assemblages*, including sensitive or specialized species, not represented adequately by the ecological communities in which they occur.
Key ecological attributes	Characteristics of Conservation Elements that are especially crucial and affect long-term persistence or viability of the Conservation Element or associated species.
Change Agents	Primary factors currently affecting or likely to affect the status of Conservation Elements.
Indicators or metrics	Measurable variables used to assess the status and condition of key ecological attributes.
Index of ecological integrity	A complementary, integrated suite of Conservation Elements that collectively represent important ecological components of an ecosystem.

Conservation Elements

The CEs (table 2) represent regionally significant terrestrial and aquatic species and communities that are of management concern. Potential CEs will be identified, prioritized, and finalized to form a limited suite of CEs for which current status and potential for change will be assessed. There are two CE categories.

Terrestrial and Aquatic Ecological Communities

Conservation of these CEs is based on the premise that intact and functioning systems are more resistant and resilient to stressors (Noss, 1987; Poiani and others, 2000). Because it is not feasible to manage or monitor all species individually, we assume that protection of intact and functioning systems will serve as a safety net for most species.

Species and Species Assemblages

These CEs include plants, animals, and other organisms. They may be single species, assemblages of taxonomically similar species (for example, bat assemblage), or species that use similar resources (for example, freshwater mussel assemblage). These CEs highlight rare or specialized species that likely would not be assessed adequately by the ecological communities (Poiani and others, 2000) because they require localized habitats or because they are already at risk and require active, targeted management to prevent further population declines. Typically, these CEs are species with special status, including declining, endemic, rare, sensitive, or area-sensitive species (table 2).

Key Ecological Attributes

For each CE, we will identify key ecological attributes (such as landscape structure) to be evaluated as part of the REA. The process of selecting the key ecological attributes will be informed by conceptual models, data availability, and relevance to the MQs. Indicator variables and metrics will be developed to quantify the key ecological attributes for use in evaluating CE status.

Change Agents

The CAs are major drivers of ecosystem change. The REA will identify and assess primary factors (CAs) that currently affect or are likely to affect the status of CEs for two future points in time (2025 and 2060). Additional time frames may be included in the assessment, as appropriate. Criteria for including CAs in the REA are as follows.

For each CE, a limited suite of the most pertinent drivers of change will be identified and evaluated. The CAs can be either anthropogenic (such as energy development or invasive species) or natural drivers that can be altered directly or indirectly by human activities (such as climate, fire, or insect outbreaks). Existing or derived data are sufficient to quantify CAs for the entire REA.

The CAs to be evaluated for the entire ecoregion minimally should include

- development (for example, urban, energy, roads, dams, agricultural activities),
- wildland fire,
- invasive species, and
- climate change.

Conceptual Models

Conceptual models are useful for describing and visualizing ecosystem components and their interactions based on the current understanding of cause and effect relationships (Manley and others, 2000). The conceptual models will be used to highlight the key ecological attributes and CAs addressed by the REA, as well as the dominant pathways and major interactions of CA influence. Although generally hypothetical, conceptual models can help to organize thinking about ecosystem integrity and to develop approaches for studying, monitoring, and managing ecosystem functions. Another important purpose of conceptual models is to make transparent the assumptions that are made when assessing potential effects of CAs on CEs. To keep conceptual models useful, we do not include all possible interactions among drivers.

Assessment Management Team

The Assessment Management Team (AMT) provides overall guidance for the development of the REA, ensures that procedures and products are consistent with project objectives, ensures a collaborative, interagency approach, and provides policy and workload guidance to the Technical Team. The AMT is composed of Federal, tribal, State, and local land management agencies.

Technical Team

The Technical Team provides technical and ecological guidance, direction, review, and recommendations for the development of the REA. The Technical Team is tasked with providing specific information and technical knowledge about the ecoregion to the Assessment Management Team to assist with developing management questions, evaluating conceptual models, reviewing process models, and interpreting results of the assessment. The Technical Team is composed of technical experts from participating Federal, tribal, State, and local land management agencies.

Process for Identifying and Selecting Priority Conservation Elements and Management Questions

Webinars

Webinars, as well as conference calls and workshops, are conducted prior to developing an REA to develop, present, discuss, and finalize the list of candidate CEs and associated MQs. These meetings facilitate extensive and in-depth input and feedback from the Technical Team and provide recommendations to the Technical Team on the scientific approaches that would best achieve the objectives and constraints of the REA. To facilitate this level of input, conference calls, webinars and workshops are approached as brainstorming sessions. Example MQs and CEs are presented to help initiate the discussions. After obtaining input on the candidate list of CEs, additional CEs may be proposed. The input received is carefully evaluated and documented. Input includes

- the objectives and sideboards for including CEs and MQs based on the REA scope of work;
- the criteria for CE and MQ selection;
- identifying participants' abilities and willingness to contribute further to the process, such as providing useful data or expert review of products; and
- organizing and synthesizing the information discussed during meetings for additional feedback and review by the AMT and Technical Team.

Compiling and Reviewing Candidate Lists of Conservation Elements and Management Questions

All candidate MQs and CEs, as derived from Technical Team and AMT input, are screened according to the selection criteria (see next two sections, "Conservation Element Selection Criteria" and Management Question Selection Criteria"), which were refined from the REA scope of work and from Carr and others (2013). All suggested candidates are tracked, and candidates that do not meet the criteria are documented accordingly. The CEs and MQs that meet the criteria are reviewed and finalized.

Conservation Element Selection Criteria

Approaches used to identify potential CEs include identifying both a candidate's regional significance and its unique or special attributes. A regionally significant CE has attributes that give it more than local significance, especially in comparison to similar ecological resources.

Attributes associated with a CE can include special worth, irreplaceable functions, exemplary or unique populations, or species otherwise considered fragile, sensitive, rare, or vulnerable to adverse change and that require management/conservation beyond the local scale (such as a BLM field office). Candidate CEs are then prioritized, as determined by the Technical Team and approved by the AMT. The final CE list is a limited suite of regionally significant communities, species, and species assemblages that collectively represent an important suite of ecological functions and natural resources for the ecoregion.

Management Question Selection Criteria

Once the CE list is finalized, the Technical Team identifies the priority MQs for each CE. Additional MQs may be drawn from across disciplines and from interested parties in the ecoregion, literature searches, prior assessments, Resource Management Plans (RMPs) and other land-use plans, and other sources identified through Phase I, Task 1. Preliminary MQs are evaluated against established criteria to prioritize and select a limited set of MQs. Potential criteria for evaluating MQs are as follows.

- Does the MQ address large-scale, ecoregion-level issues (that is, does the effect of decisions cross field office boundaries, connected phenomena, and so on)?
- Can the MQ be answered by available geospatial information, remote sensing, or acceptable surrogates at the landscape scale?
- Does the MQ identify the potential subsequent decision process and (or) action associated with the answer to the question?
- Does the MQ relate to the key processes, attributes, and indicators for the ecoregion model?
- Has the MQ been answered in another recently completed ecoregion assessment, and is there additional information that warrants reexamining this issue?

Review and Finalize Components

The CEs and MQs that have met the established REA criteria above are documented accordingly and presented to the AMT as final draft candidates. If additional consideration is needed, a supplemental draft list is distributed for review by the AMT; the AMT provides its final recommendations to the Contracting Officer’s Representative, who provides the final list to the contracting team. The analyses and products generated for each CE serve to answer relevant priority MQs.

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Chapter 2. Southern Great Plains Rapid Ecoregional Assessment Overview

By Timothy J. Assal and Natasha B. Carr

Background on the Southern Great Plains Region—Ecological Setting

The SGP REA area encompasses more than 72.8 million hectares (ha) in the south-central United States. The area includes three ecoregions as defined by Omernik (1987): the Central Great Plains, the High Plains, and the Southwestern Tablelands. The REA project boundary includes all 5th-level hydrologic unit class watersheds that intersect the project perimeter (fig. 1). The project area covers portions of five states: the northernmost Panhandle of Texas, western Oklahoma, eastern New Mexico, eastern Colorado, and westernmost Kansas.

Agriculture is the dominant land use in the region. The elevated Southwestern Tablelands contain little irrigated agriculture and are largely used for livestock. Higher in elevation and drier than the Central Great Plains to the east, much of the western High Plains is used for cropland. The Central Great Plains are slightly lower in elevation and receive more precipitation than the plains to the west. Once a vast grassland with scattered low trees and shrubs in the south, much of this ecological region is now cropland.

Biophysical Controls

Many of the biophysical controls of the SGP project area reflect an east-west gradient from the Rocky Mountain Front in the west to the low plains in the eastern portion of the region. The highest point in the SGP REA project area is 2,708 meters (m) at Cerro Pelon, New Mexico. The lowest point is found where the Red River exits the project area (222 m) along the Oklahoma-Texas border near the town of Spanish Fort, Texas. The elevation range of the project area is 2,486 m.

The surficial geology is quite varied across the region (Soller and others, 2009). The Front Range of New Mexico and Colorado are dominated by residual materials of igneous and metamorphic rock. Isolated areas of volcanic rock (basaltic and andesitic) dot the landscape of northeastern New Mexico. The uplands in New Mexico and Colorado are dominated by residual materials of sedimentary rocks. The major river valleys of eastern Colorado are dominated by thin alluviums and delineated from the uplands by bands of eolian loess and sand dunes. The alluvial deposits delineate major stream valleys that cross the southern Great Plains from west to east. The Oklahoma and Texas panhandles and areas in central Kansas are composed of eolian sediments with discontinuous colluvial material. Large areas of colluvial sediments and loess are found in northwestern Kansas. The eastern portion of the project area in central Texas and Oklahoma is dominated by bands of residual materials developed in sedimentary rock and bedrock (Soller and others, 2009).

Soil development is governed by geology, climate, topography, biology, and time (Hess, 2011). Dry mollisols are the dominant soil type in central Texas, central Oklahoma, and central Kansas. Alfisols are found in the eastern portion of the study area in Oklahoma and Texas, as well as parts of the Texas panhandle. Aridisols and entisols dominate in southeastern Colorado, and alfisols and aridisols are found in northeastern New Mexico (Hess, 2011). Prairie soils are typically poor in nitrogen and carbon (National Park Service, 2008). Climate, vegetation, biota, topography, and parent material contribute to the rate of soil formation, which is typically 1 inch

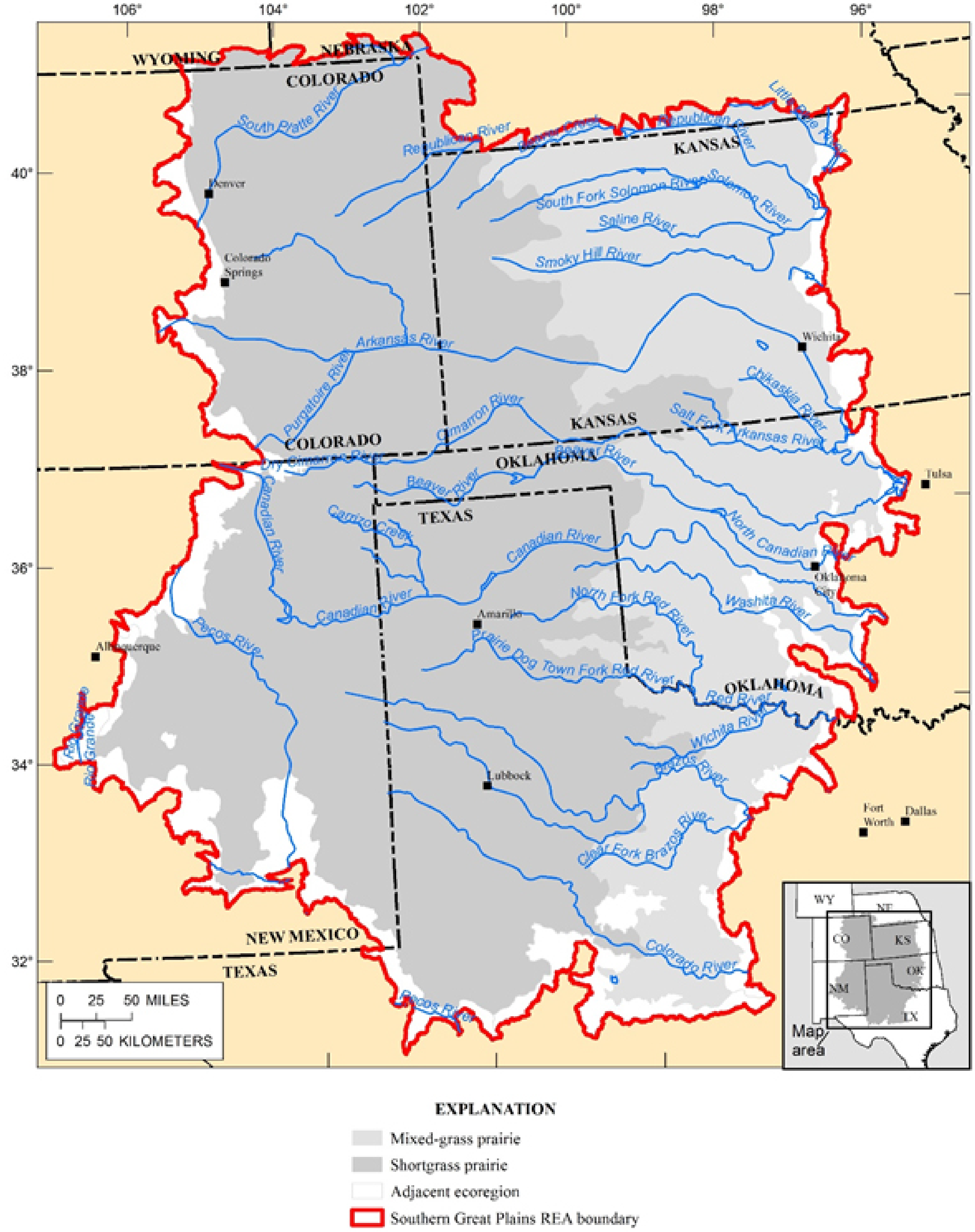

Figure 1. Overview map of the Southern Great Plains Rapid Ecoregional Assessment (REA) project area. (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.)

of topsoil every 100–1,000 years. The vast majority of native prairie vegetative biomass is found below ground. Cultivation of prairie soils removes extensive root systems and organic carbon, and it reduces soil stability and overall soil productivity (National Park Service, 2008).

The assemblage of ecosystems in the SGP are governed by two climatic gradients: precipitation (east-west) and temperature (north-south) (National Park Service, 2008). Annual precipitation ranges from 31 centimeters (cm) on the western plains to 97 cm in south-central Oklahoma. Approximately two-thirds of the rainfall occurs between April and September, and frequent droughts, the result of reduced precipitation and increased evapotranspiration, occur regularly. Average maximum daily temperatures peak at 26 °C (south-central Texas), and average minimum daily temperatures hover at –0.5 °C in the western part of the region (National Park Service, 2008).

Major Ecosystems and Communities

The SGP region consists mostly of mixed-grass and shortgrass terrestrial ecosystems. Blue grama (*Bouteloua gracilis*) and buffalograss (*B. dactyloides* [formerly in the genus *Buchloe*]) dominate the western grasslands, where big bluestem (*Andropogon gerardii*) and little bluestem (*Schizachyrium scoparium*), switchgrass (*Panicum virgatum*), and Indiangrass (*Sorghastrum nutans*) become more prevalent in the eastern portion of the region (National Park Service, 2008). Cottonwood trees (*Populus deltoides*) are found along riparian areas in the west, and American elm (*Ulmus americana*), sugarberry (*Celtis laevigata*), bald cypress (*Taxodium distichum*), and green ash (*Fraxinus pennslyvanica*) may be found along streams with increasing precipitation moving east (National Park Service, 2008). Today, only small fragments of native ecosystem remain in the Great Plains. Much of the Great Plains region, including this project area, has been converted to farmland or grazed intensively by domestic livestock (Ricketts and others, 1999).

Grazing by bison and prairie dogs was a primary ecological driver in the Great Plains (National Park Service, 2008). In the middle of the 19th century, large populations of grazing ungulates existed on the Great Plains, including American bison (*Bison bison*), pronghorn (*Antilocapra americana*), and elk (*Cervus elaphus*). These populations formed the prey base for grizzly bears (*Ursus arctos*) and gray wolves (*Canis lupus*) (Ricketts and others, 1999). Conspicuous herbivores such as black-tailed prairie dogs (*Cynomys ludovicianus*) occupied expansive areas of open grasslands (Proctor and others, 2006). Since that time, the large natural predators have been hunted to extinction, bison have been extirpated, and prairie dog populations have been greatly diminished from their historical levels. Grasslands remain an important habitat for small mammals and numerous grassland bird species. Shrublands found in the grassland mosaic contribute to habitat for the lesser prairie-chicken (*Tympanuchus pallidicinctus*) and the dunes sagebrush lizard (*Sceloporus arenicolus*).

Although grasslands are the dominant ecosystems, numerous aquatic habitats are found in the semiarid landscape, providing habitat for prairie birds, migrating waterfowl, and breeding and migratory shorebirds. Large streams throughout the SGP are characterized by moderate flow, dynamic channels, and high turbidity and high evaporation rates. Small streams in the study area are characterized by irregular flow and a distinct wet-dry cycle. Streams in the region provide habitat for pelagic spawning fishes (including the Arkansas River shiner [*Notropis girardi*]) and numerous species of freshwater mussels. Riparian areas and wetlands bridge aquatic environments with upland terrestrial ecosystems. Playas form in shallow depressions in relatively flat terrain and provide important wildlife habitat, particularly for waterbirds (Mitchusson, 2000). Manmade impoundments have altered stream courses and created large reservoirs in the region.

Ecosystem Processes

Climate and disturbance are major drivers of vegetation community composition and structure. The climatic gradient in the region influences several major ecosystem processes. Regional patterns of aboveground net primary production in grasslands reflect the east-west precipitation gradient, with lowest values of aboveground net primary production observed in the west and highest values in the east (Sala and others, 1988). Burke and others (1989) found that soil organic carbon of grassland soils increases with precipitation and clay content and decreases with temperature.

Grasses in the region evolved under frequent disturbance from large mammalian grazers, trampling, and fire. Perennial grasses reproduce via seed and vegetatively through underground rhizomes that are protected from ungulates (Ricketts and others, 1999). Climate influences the physiological distribution of grasses, with C_3 grass being prevalent in the north and C_4 grass being dominant in the south (Sage and others, 1999). The C_3 grasses are most efficient at photosynthesizing in cooler temperatures, while warm-season grasses have a photosynthetic pathway adapted to hot climates (Barbour and others, 1999).

Vegetation of natural grasslands favors frequent fire because the dry leaves and stems accumulate rapidly, producing extensive amounts of fine fuels easily ignited during convective storms (Knight, 1994; Barbour and others, 1999). Prior to Anglo-European settlement, the continuity of flammable vegetation on the prairie allowed fires to burn large areas (Barbour and others, 1999). Fires could burn for weeks or months until fuels were consumed, a barrier was reached (for example, a watercourse), or the weather changed (Knight, 1994). Grassland plants are adapted to frequent, low-severity fire and may quickly resprout from intact roots or increased reproduction from seed (Barbour and others, 1999). Fire plays a key role in nutrient cycling and subsequently influences primary productivity of grasslands. Combustion converts organic matter into the inorganic nutrients needed for plant growth, and primary production often increases after fire (Knight, 1994).

Drought occurs regularly in the region. Drought reduces grassland productivity, limits plant population densities, and increases mortality of annual forbs and grasses within stands of perennial species (Tilman and El Haddi, 1992). The most severe drought in the modern record, the Dust Bowl of the mid-1930s, was largely confined to the shortgrass prairie region of the study area (Hart, 2008). Multidecadal droughts in the shortgrass prairie have been documented in the tree ring record dating from the 16th century (Stahle and others, 2000).

Overall Grassland Ecosystem Model

Conceptual models are a useful tool to convey dynamics and highlight important processes within an ecosystem. Furthermore, they illustrate connections between ecological drivers and anthropogenic stressors. Drivers refer to natural agents responsible for temporal changes or variability in quantitative measures of structural and functional attributes of ecosystems. Anthropogenic stressors are physical, chemical, or biological perturbations to a system that cause substantial changes in the ecological components, patterns, and processes in natural systems (Barrett and others, 1976). We used a conceptual model developed through the National Park Service Inventory and Monitoring program’s Southern Plains Network to illustrate grassland dynamics in the region. The Southern Plains Network is composed of 11 National Park Service units located in the five states included in the SGP REA. The model illustrates important overall ecosystem components, drivers, stressors, and the interaction of each in the SGP ecoregion (fig. 2). The model also identifies external stressors that are characteristic of private agricultural lands. The National Park Service units offer some of the most intact landscapes of the SGP and can provide the closest reference conditions because there are fewer anthropogenic influences within the park units.

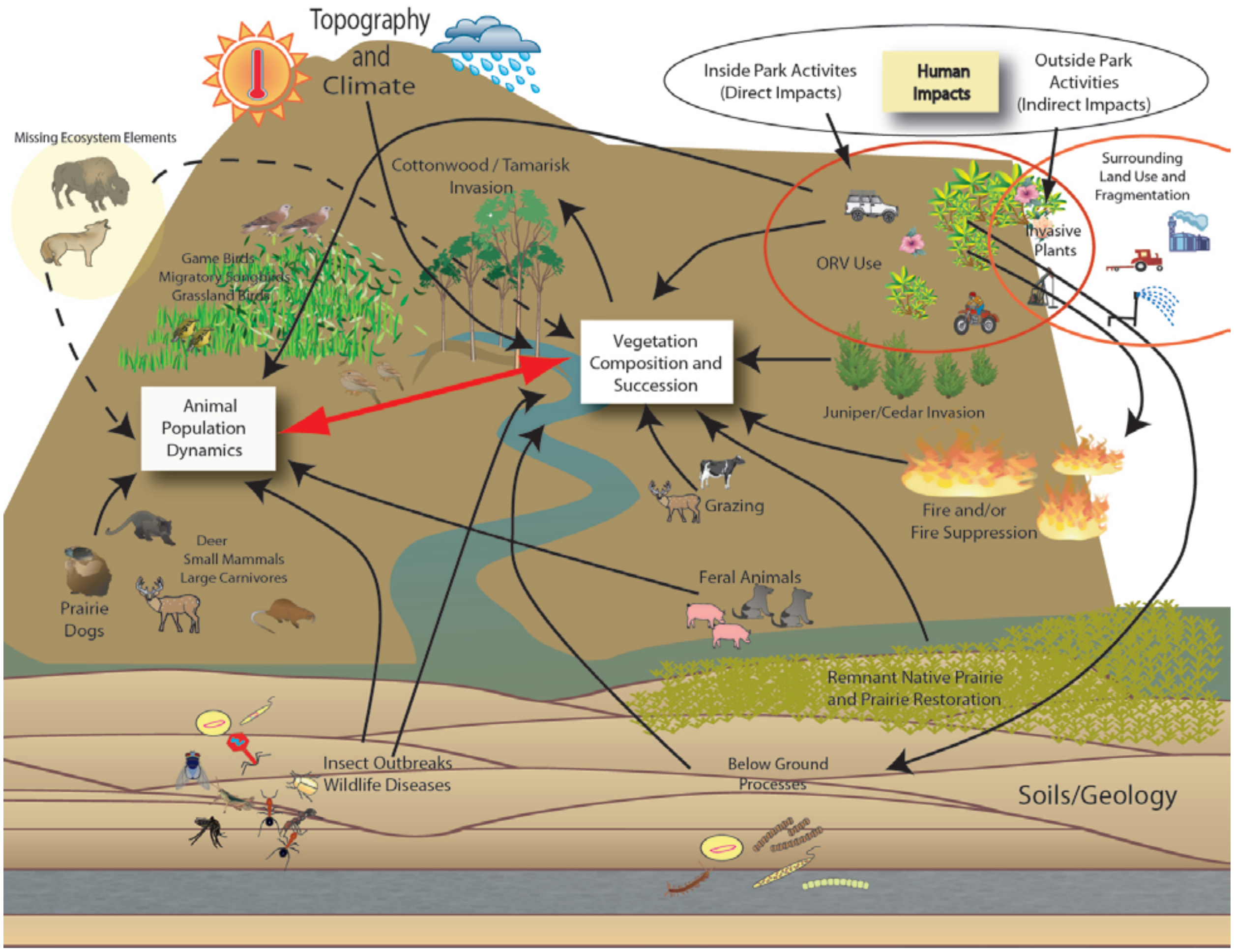

Figure 2. Conceptual model developed by the National Park Service's Southern Plains Network Vital Signs Monitoring Plan (National Park Service, 2008). The model identifies the major ecological drivers and anthropogenic stressors in the region, along with core vital signs (red arrow) monitored at National Park Service units. (Courtesy of the National Park Service) (ORV, off-road vehicle)

Background on the Southern Great Plains Region—Human Context

Demographic Overview

The population of the Great Plains began to increase after the Civil War as the population migrated westward from more densely populated areas in the east (Gutman and others, 2005). The shift was encouraged by the Homestead Act of 1862 under which 1.5 million people settled in the region. Settlers found potential for arable crops in the eastern portion of the Great Plains and pasture land in the west (Gutman and others, 2005). Large tracts of native grassland were subsequently converted to row-crop agriculture or pasture (Samson and others, 2004). Although the region's rural population has been declining since the 1930s (Gutman and others, 2005), the total population has steadily increased from 1950 to the present (Wilson, 2009), signifying a shift to urban areas. Population trends differ by region of the Great Plains, but overall the population growth lagged slightly behind that pace of national population growth (Archer and Lonsdale, 2003). Today the major population centers are found on the eastern and western edges of the region. In the eastern portion of the SGP, the major population center, which includes Wichita Falls, Oklahoma City, and Wichita, stretches along the Interstate 44/Interstate 35 corridor. The corridor along the Colorado Front Range and Interstate 25, which includes Pueblo, Colorado Springs, Denver, and Fort Collins, is also a densely populated landscape. Smaller communities are found along Interstate 70 in the north and Interstate 40 in the south.

Land Ownership

The total land area of the SGP is more than 72 million ha. Most of the surficial land in the project is privately owned (fig. 3, table 3). Collectively, State, Federal, and tribal lands compose less than 10 percent of the project area. Most Bureau of Land Management (BLM) lands within the SGP are located in New Mexico (table 3), but the SGP overlaps eight BLM Field Office jurisdictions: the Royal Gorge Field Office in Colorado; the Taos, Roswell, Socorro, Rio Puerco and Carlsbad Field Offices in New Mexico; and all BLM lands in Texas, Oklahoma, and Kansas encompassed by the Oklahoma Field Office.

Land Use

The region experienced social and economic disruptions during the Dust Bowl of the 1930s, yet land use changed little during this time. The amount of land conversion to cropland peaked in 1940 and remained stable during the 20th century, despite the depopulation in rural areas (Gutman and others, 2005). By the 1940s, advanced pumping technology and the availability of rural electric power encouraged the use of groundwater for crop irrigation (Hart, 2008). The largest source of groundwater in the region is the Ogallala aquifer, which has been severely depleted by extensive pumping for irrigation (Nativ and Smith, 1987).

Agriculture remains the most important land use in the SGP ecoregion, with farming and ranching dominating the eastern and western regions, respectively. Livestock grazing is the main agricultural activity of the shortgrass prairie, whereas cultivated wheat is the dominant land use of the mixed-grass prairie (Hart, 2008; National Park Service, 2008). Many small family farms have been consolidated into larger, corporation-owned farms. Farmland has been converted primarily through either farm abandonment or urban sprawl (Parton and others, 2003). Mineral and energy extraction are important economic drivers in the region (National Park Service, 2008), and wind development has accelerated in recent years. Population dynamics in the region have implications on current and future land use, including conventional and wind energy development. This subject will be explored further in Phase II.

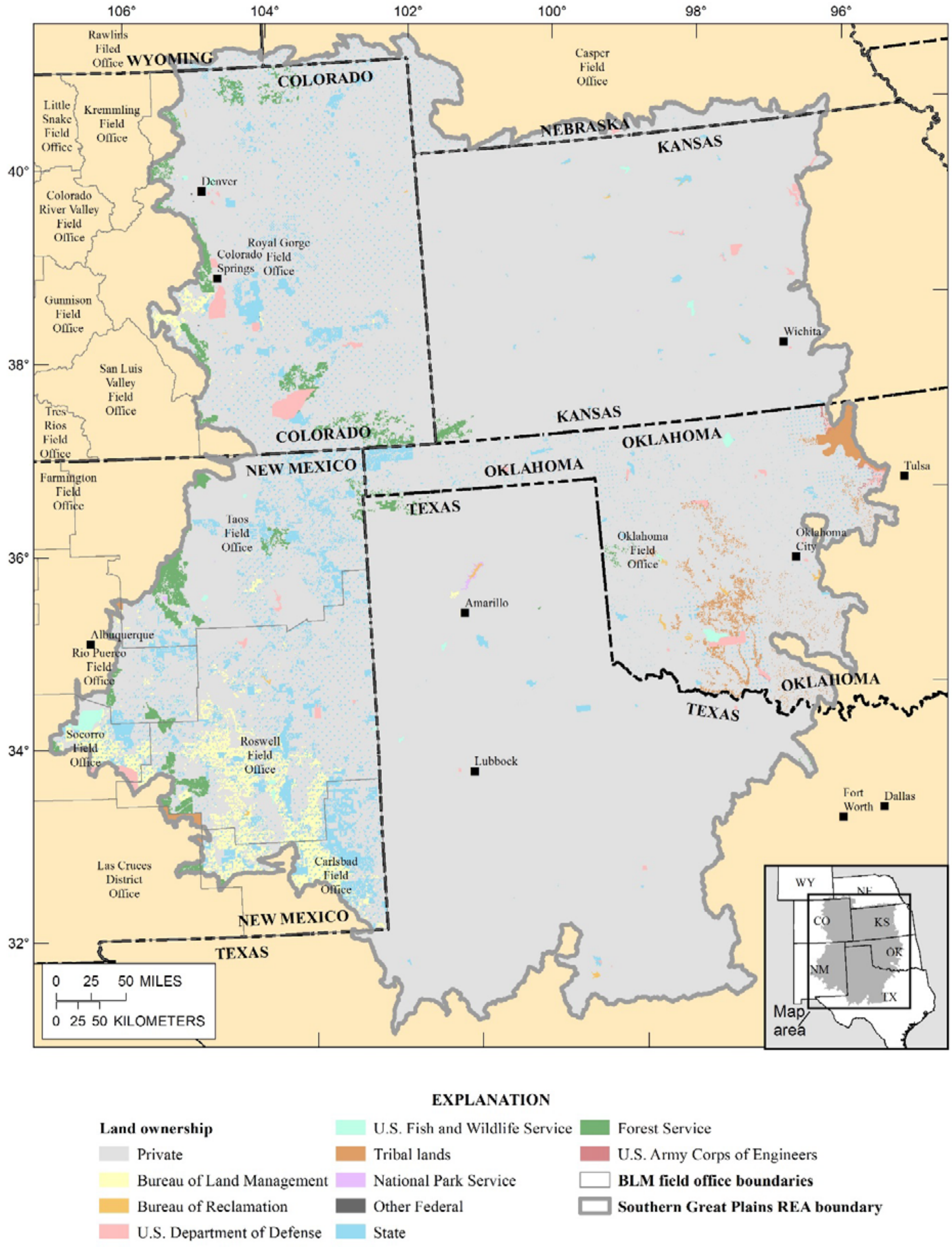

Figure 3. The Southern Great Plains Rapid Ecoregional Assessment (REA) project area and jurisdictions, including Bureau of Land Management (BLM) Field Office boundaries.

Table 3. Area and percentage of land managed or owned by different entities within each state in the Southern Great Plains Rapid Ecoregional Assessment project area (based on fig. 3).

Jurisdiction	Total area (acres)	Percent	Area by state (acres)						
			Colorado	Kansas	Nebraska	New Mexico	Oklahoma1	Texas1	Wyoming
Private	167,085,778	90.6	26,409,240	37,648,948	3,583,109	22,846,385	23,361,006	52,873,511	363,579
States	8,046,891	4.4	2,031,230	139,057	127,385	4,778,894	832,803	102,880	34,642
Bureau of Land Management	3,441,328	1.9	283,199	0	18	3,142,877	3,275	11,958	-
Forest Service	2,782,275	1.5	1,224,479	108,628	6	1,314,638	46,439	88,066	19
Tribal lands	1,446,260	0.8	-	-	-	165,796	1,279,419	1,045	-
U.S. Department of Defense	1,016,479	0.6	449,644	128,465	13,862	190,399	217,416	16,693	-
U.S. Fish and Wildlife Service	376,165	0.2	26,307	33,327	-	197,281	104,696	14,554	-
Bureau of Reclamation	107,213	<0.1	6,947	11,246	-	11,752	50,877	26,391	-
U.S. Army Corps of Engineers	50,982	<0.1	-	-	-	-	50,982	-	-
National Park Service	37,684	<0.1	734	664	-	7,817	-	28,469	-
Other Federal lands	1,923	<0.1	1,923	-	-	-	-	-	-
TOTALS	184,392,978		30,433,703	38,070,336	3,724,380	32,655,839	25,946,913	53,163,567	398,240

[1] Acreage totals for Bureau of Land Management land in Texas and Oklahoma do not include an estimate for the Red River public domain lands.

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Chapter 3. Southern Great Plains Rapid Ecoregional Assessment

By Timothy J. Assal and Natasha B. Carr

Conservation Elements and Primary Management Issues

The CEs represent regionally significant terrestrial and aquatic species and ecological communities that are to be conserved and (or) restored. The initial proposed set of CEs was reduced to a limited suite of CEs for which current status and potential for change will be assessed. There are two CE categories: (1) ecological communities and (2) species or species assemblages. Focus on communities is based on the premise that intact and functioning terrestrial and aquatic ecosystems are more resistant and resilient to stressors, and it is assumed that intact and functioning systems can serve as a safety net for most species. Species level CEs include single species or assemblages of taxonomically similar species of plants, animals, and other organisms. Species assemblages may also be considered for species that use similar resources. Species CEs may include those with special status, including endemic, rare, sensitive, or area-sensitive species. These CEs highlight rare or specialized species that likely would not be assessed adequately by the ecological communities, because they require localized habitats or because they are already at risk and require active, targeted management to prevent further population declines.

Identification of Conservation Elements

Identification of CEs was a primary task in Phase I of the Rapid Ecoregional Assessment (REA). A series of workshops and webinars were used to identify primary conservation and management issues in the SGP. Initially, two workshops were held with the Assessment Management Team (AMT) and Technical Team in Arcadia, Oklahoma, in April 2013. The purpose of these workshops was to introduce the REA concept and potential products to members of the AMT and Technical Team, respectively. In addition we sought to begin the discussion on the management questions and CEs specific to each member's agency. The groups held lengthy discussions of current regional initiatives that could be utilized in this REA. Early on, we recognized the identification of CEs would be an iterative process between scientists and resource managers. Criteria used to select CEs, in all cases as recommended by the Technical Team and the U.S. Geological Survey (USGS) and approved by the AMT, were as follows.

- Six to eight regionally significant terrestrial and aquatic communities of the major ecological systems and functions of the SGP.
- Ten to twelve regionally significant species or communities that occur throughout the jurisdiction of at least two states in the SGP project area, with an emphasis on widely distributed species; this criterion was developed to help meet the REA goal of ensuring that the REA is relevant to regional priority management issues (other management issues may be addressed by specific MQs).
- Commodity species (game or furbearer species).
- Species or species assemblages of conservation concern.
- Species directly tied to management priorities and issues.

- Species not addressed adequately by an assessment of the ecological communities in which those species occur.

Identification of Ecological Community Conservation Elements

The Great Plains Landscape Conservation Cooperative (GPLCC) list of high-priority habitats (http://www.greatplainslcc.org/about/priorities/) was used to identify a preliminary set of priority communities. The GPLCC is a partnership that provides applied science and decision support tools to assist natural resource managers in conservation of species and communities in the SGP. Members of the Technical Team provided additional priority ecological communities representative of the SGP to be considered as CEs. The AMT stipulated that all of the major ecological terrestrial and aquatic ecological communities within the REA should be represented (there should be no gaps).

Shortgrass and mixed-grass prairies are the dominant terrestrial communities of the project area. There is no distinct boundary between these two communities, but rather an ecological gradient. The variation in the distribution and dominance of species within this gradient clearly points to site conditions and history, including management, fire, and grazing, as having important determinant effects on community structure and composition. Available data describing subordinate information within each terrestrial community will be evaluated in Phase II. The Natural Resources Conservation Service provides a framework for classifying and describing rangeland soils and vegetation through the Major Land Resource Area and Ecological Site Descriptions. There are 38 unique Major Land Resource Area units in the SGP project area. This information can be used to describe local site potential. In Phase II, we will investigate if the framework is feasible to use in a landscape-scale rapid ecoregional assessment. Communities that only occur in the buffer of the project area were not included because they are best addressed in adjacent ecoregions.

The Technical Team identified two shrub species, shinnery oak (*Quercus havardii*; also known by some authorities as Harvard oak) and sand sagebrush, that provide crucial habitats within the grassland matrix. These shrublands are intermixed in the shortgrass and mixed-grass prairie mosaic of the SGP. We determined these would be best addressed as key ecological attributes for the shortgrass and mixed-grass prairie communities. These species dominate habitats for several endemic and regionally important species, indicating a need to manage for healthy oak and sand sagebrush (*Artemisia filifolia*) communities to conserve habitats within the prairie landscape. The shrub species were given special consideration as part of the mixed-grass prairie CE, although they do occur in the shortgrass prairie as well.

The aquatic ecological communities selected include riparian and wetlands, playas and saline lakes, lakes and reservoirs, and prairie rivers and streams. There are six ecological communities to be considered in Phase II (fig. 4).

Identification of Species and Species Assemblage Conservation Elements

The USGS proposed a suite of selection criteria to provide an objective and transparent process to reduce the initial list of species to a final list of 10−12, as proposed in the scope of work. The Technical Team provided input on the selection criteria, and the AMT approved the final criteria below.

I. Priority species or assemblages of conservation concern as determined by the Bureau of Land Management and other Federal and State agencies.

II. Directly tied to management priorities and issues of representatives on the technical team.
III. Regionally significant species or communities: occurrence throughout at least two SGP REA States, with an emphasis on widely distributed species.
IV. Commodity species (game or furbearer species).
V. Species not adequately addressed by ecological community CEs or other species CEs.

The GPLCC developed a list of priority wildlife species (http://www.greatplainslcc.org/about/priorities/) from information in the Wildlife Action Plans of the State partners encompassed within the GPLCC boundary, as well as regional planning documents of the U.S. Fish and Wildlife Service and The Nature Conservancy. We used the GLPCC list as a preliminary list of priority species to identify species that met criterion I. In addition, Technical Team members proposed candidate species from their respective State Wildlife Action Plans. We identified 131 species for the initial list.

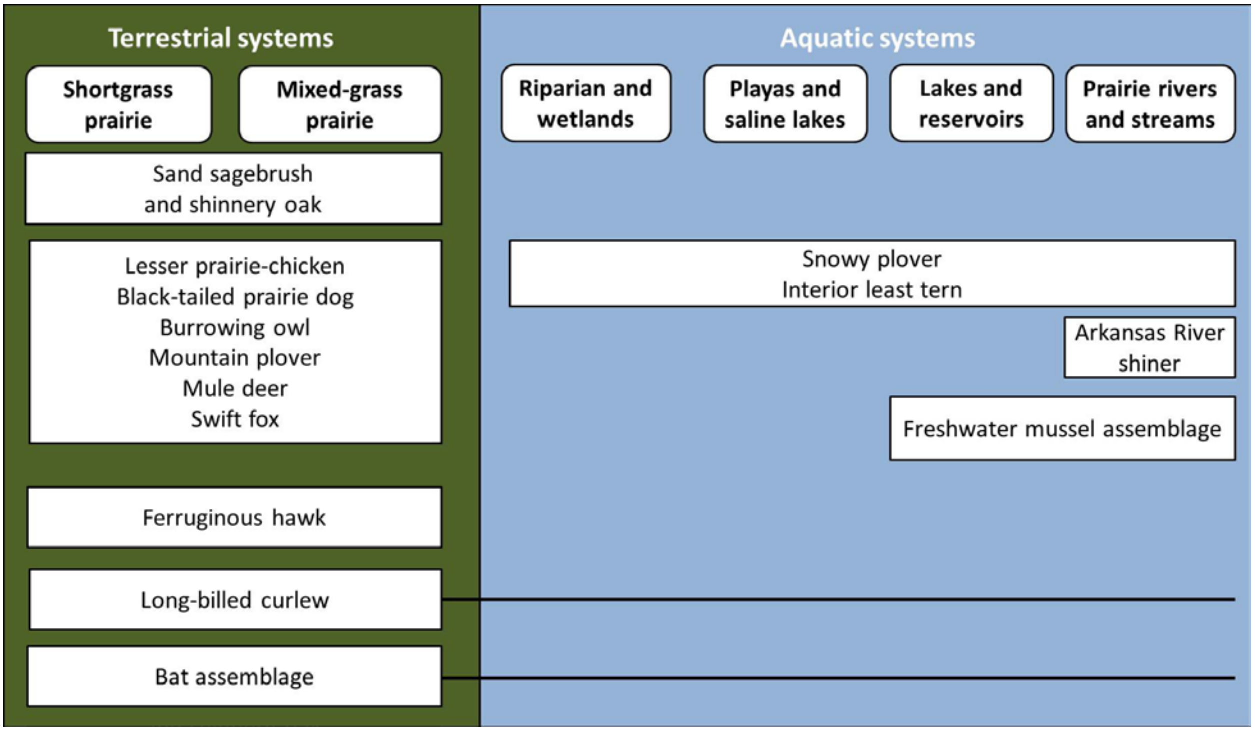

Figure 4. Ecological communities, species, and species assemblages identified as Conservation Elements (CEs) for terrestrial (green) and aquatic (blue) systems. The ecological community CEs are shown in bold text in the top row of white boxes. Species and species assemblage CEs are shown in boxes below the primary community (or communities) where they are found. For wildlife species that are strongly tied to terrestrial systems but also use adjacent aquatic habitats, horizontal lines are placed under the aquatic systems also used by that species. (Harvard oak [*Quercus harvardii*] is commonly known as shinnery oak.)

To evaluate criterion II, each member of the Technical Team provided a management-priority rank for each candidate species in the preliminary list. The Technical Team identified three species assemblages: fish, freshwater mussels, and bats. Candidate species of management concern that did not meet the regionally significant criterion could still warrant consideration as part of a species assemblage. Technical team members subsequently ranked each species or species assemblage based on their agencies' priorities. Technical team members also identified management issues associated with a species, particularly if the species was ranked as a high or medium priority. The USGS collated all responses and assigned a score based on the priority level (high = 3, medium = 2, low = 1, not a priority = 0). The scores were summed across all five states, and the priority score was used to rank the candidate species level CEs.

The species ranked in the top 30 were evaluated for additional criteria. To evaluate regional significance (criterion III), the BLM developed range maps of the top 30 species. The technical team identified game species (criterion IV) and engaged in discussion regarding criterion V for the remaining species on the list.

The AMT approved the final list of 13 species as CEs. The list includes seven bird species, four mammals (including one species assemblage), one fish, and one invertebrate species assemblage (fig. 4, table 4). Candidate species that did not make the final list are included in appendix A. A total of 19 ecological communities and species CEs will be assessed in Phase II. Data availability for each CE will be evaluated at the start of Phase II. It is possible that not all CEs can be fully assessed in Phase II if adequate data are not available.

Table 4. Final list of species Conservation Elements.

[NM, New Mexico; TX, Texas; CO, Colorado; OK, Oklahoma; KS, Kansas. Management priority: H, high; M, medium; L, low]

	Selection Criteria[1]								
	I					II	III	IV	V
	Management priority								
Species	**NM**	**TX**	**CO**	**OK**	**KS**	**Regional priority score**	**Regionally significant**	**Commodity species**	**Addressed by community conservation element**
Lesser prairie-chicken	H	H	H	H	H	15	Yes	Yes	No
Black-tailed prairie dog	H	H	M	L	H	12	Yes	Yes	No
Interior least tern	H	M	L	M	M	10	Yes	No	No
Burrowing owl	H	M	M	L	L	9	Yes	No	No
Mountain plover	H	M	M	L	L	9	Yes	No	No
Long-billed curlew	H	M	L	L	L	8	Yes	No	No
Freshwater mussel assemblage	H	M	-	-	H	8	Yes	No	No
Ferruginous hawk	M	M	L	L	L	7	Yes	No	No
Mule deer	H	L	-	-	H	7	Yes	Yes	No
Snowy plover	H	H	-	L		7	Yes	No	No
Bat assemblage	M	M	-	-	H	7	Yes	No	No
Swift fox	M	H	L	-	L	7	Yes	Yes	No
Arkansas River shiner	H	-	-	M	M	7	Yes	No	No

Conservation Element Overview

An overview on the ecology and management issues for each CE is provided in Section II (Chapters 4–22). The USGS assembled a team of 12 subject matter specialists composed of Federal and university scientists. Each CE chapter consists of three sections; key ecological attributes, CAs, and REA components. The key ecological attribute section details the distribution and ecology, landscape structure and dynamics, and associated species of management concern affiliated with each CE. The landscape structure and dynamics section describes the historic patterns and processes (for example, fire, climate, drought) of ecological communities and the response of species to the structure and dynamics of their habitat. The Change Agents section details how CAs disrupt the historic or natural landscape structure and dynamics.

An overview of potential key ecological attributes and CAs are summarized by conceptual models and tables. A basic conceptual model (fig. 5) was developed for each CE to highlight dominant ecological processes, primary interactions, and potential effects of major drivers and stressors. The key ecological attributes table (table 5) summarizes variables that may be assessed in Phase II. Table 6 summarizes the potential effects of each CA. The tables provide an organizational framework and background information for evaluating key ecological attributes and CAs in Phase II.

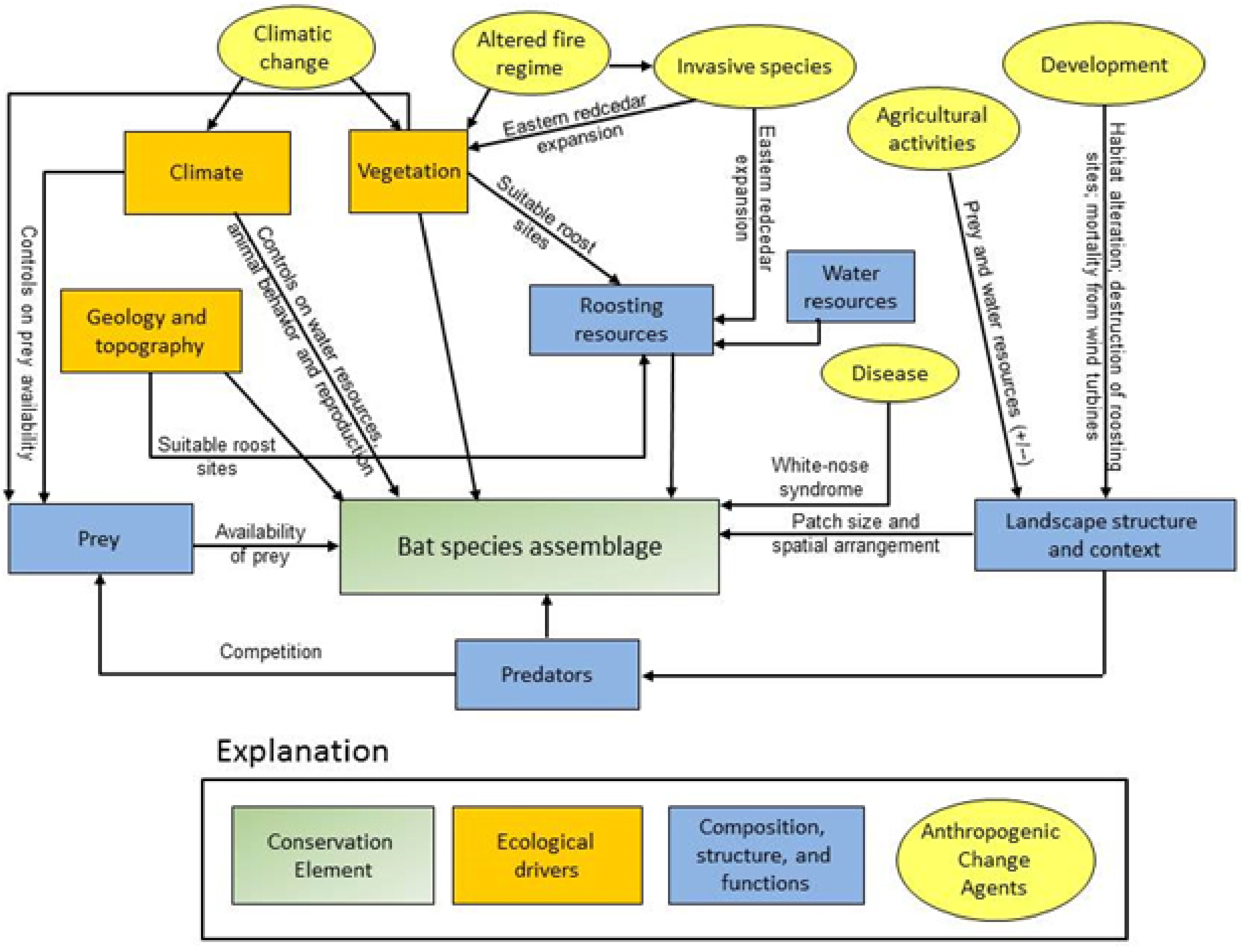

Figure 5. Example of generalized conceptual model using the bat species assemblage.

Table 5. Example of key ecological attributes table using the bat species assemblage.

Attribute	Variables
Amount and distribution	Bat species distribution (occurrence records).
Landscape structure	Size and spatial distribution of habitat; availability and type of roost sites.
Landscape dynamics	Habitat productivity (availability of prey food resources), drought (effects on water resources, prey availability), roosting resources, predator dynamics.

Table 6. Example of Change Agent table using the bat species assemblage.

Attribute	Variables
Development (energy and infrastructure)	Habitat alteration (disturbance/destruction of roosting resources; noise pollution), contamination/loss of water resources, contamination of food resources, mortality (wind turbines).
Development (agricultural activities)	Habitat alteration (fluctuation in availability of prey [±]; effects on water resources [±]).
Altered fire regime	Fire suppression results in habitat alteration (expansion of forested areas may increase roosting structures; effects on prey availability).
Invasive species	Habitat alteration (expansion of eastern redcedar—increase in forage and roosting [+], reduction in water resources [-]).
Insects and disease	Presence and effect of white-nose syndrome on population.
Climate change	Potential changes to habitat condition (distribution and abundance of vegetation, prey, and water resources; roost microclimate); potential changes to behavior (timing and success of reproduction and hibernation).

Change Agents

The specific influence of each CA varies by CE and will be discussed in detail in each CE chapter. Below we summarize the overall patterns and general effects of each CA.

The grasslands of the Great Plains are considered one of the most endangered ecosystems in North America. Human settlement and agriculture have converted open grasslands into a mosaic of cultivated croplands, prairie remnants, and expanding woodlands (Samson and Knopf, 1994). Approximately 40–90 percent of mixed-grass and 50–75 percent of shortgrass prairie has been under cultivation at some point in time since European settlement (Bragg and Steuter, 1996; Weaver and others, 1996). Areas of the prairie that have not been cultivated are intensely grazed by domestic livestock that has undoubtedly influenced ecosystems in the southern Great Plains (Ricketts and others, 1999). The effects of grazing and grazing management on plant and animal communities are variable and can be both direct and indirect. Effects include loss of riparian vegetation, removal of vegetative cover, and dispersal of seeds from invasive plant species (Chaney and other, 1990; Vavra and others, 2007). The effects of livestock grazing on an individual species are also variable and complex. Nests of bird species, including lesser prairie-chicken, can be damaged or destroyed by intensive grazing of domestic livestock as well as native ungulates (Pitman and others, 2005). However, grazing can produce the shorter grassland vegetation structure preferred by long-billed curlews (*Numenius americanus*) for nesting. Grazing lands are often positively associated with bird presence, particularly if intensity of grazing pressure is varied to increase landscape heterogeneity (Derner and others, 2009).

Energy development (including oil, gas, and wind) is highly detrimental to some species and brings with it associated infrastructure, including well pads, wind turbines, roads, pipelines, and transmission lines. The infrastructure used to support industrial activities across the prairie landscape has a range of effects from the redistribution of water resources (for drilling, hydraulic fracturing, and extraction of coalbed methane) to the introduction of exotic species. These activities result in a combination of habitat loss (direct surface use), habitat change (often considered degradation), and landscape-level habitat fragmentation. These activities are often juxtaposed with other land uses and natural patterns, resulting in cumulative effects on habitat condition and distribution.

The hydrology of major streams in the region has been altered as the result of impoundments, withdrawals (municipal and irrigation), groundwater depletion, and a multitude of upland land-use changes. As a result, timing and flow, water temperature, dissolved nutrients, and sediment levels have been altered (National Park Service, 2008). Water quality in the region has been affected by the use of herbicides, urban pollutants, and fecal contamination (National Park Service, 2008). Groundwater depletion is a regional major concern, centered on the Ogallala aquifer. The Ogallala aquifer underlies parts of all five states in the SGP REA project area and extends north into Nebraska and South Dakota (Nativ and Smith, 1987). The aquifer declined by an average of 3 m during a 40-year period in the middle of the 20th century (Dugan and others, 1994).

Prior to Anglo-European settlement and the introduction of livestock grazing, fences, and roads, fire was a major ecological process influencing the structure, composition, and productivity of shortgrass and mixed-grass prairie ecosystems on the Great Plains (Brockway and others, 2002). Since that time, however, the frequency and extent of fire has dramatically declined as a result of organized fire suppression efforts and heavy grazing by domestic livestock, which reduced the available levels of fine fuels. The altered fire regime in these ecosystems is thought to be responsible for ecologically adverse shifts in the composition, structure, and diversity of these grasslands. Over the last 125 years there has been a rise in ruderal species and invasion by less fire-tolerant species (Brockway and others, 2002). Historically, fire maintained grassland ecosystems by restricting the extent of woody plant species such as juniper (*Juniperus* spp.), mesquite (*Prosopis* spp.), eastern redcedar (*Juniperus virginiana*), and creosote bush (*Larrea tridentata*) (Engle and others, 1996; Weaver and others, 1996). However, woodlands have expanded at the expense of grasslands due to altered fire regimes, intensive livestock grazing, climate change, intentional planting, and removal of native herbivores (Weltzin and others, 1997).

Insects and disease include both native and introduced organisms. Recently introduced diseases, such as West Nile virus (Flavivirus Japanese encephalitis antigenic complex), can be especially devastating to species that lack any natural immunity. Chronic wasting disease is a significant concern in mule deer (*Odocoileus hemionus*) populations and has been reported in Colorado and western Kansas. The disease white-nose syndrome is associated with a fungal species that affects hibernating bats east of the SGP; however, white-nose syndrome is a concern for this region, as it could have a significant effect on bat populations. Invasive species also are a concern, including tamarisk (*Tamarix* spp.) in riparian areas to cheatgrass (*Bromus tectorum*) in shortgrass prairie. Zebra mussels (*Dreissena polymorpha*) have spread throughout lakes and reservoirs over the last 25 years, including the SGP region.

Climate change has the potential to change the landscape in fundamental ways, with potential consequences for natural communities and exacerbating many other CAs. Climate change predictions suggest higher temperatures in both summer and winter in the SGP. The signal for changes in precipitation is less clear, but decreases in rainfall are likely in more southern areas in the coming decades. Because the water cycle is strongly influenced by temperature, projected evaporation could increase, and soil moisture and relative humidity could decline (Kirtman and others, 2013).

Organizational Diagram of CEs and CAs

We developed an organizational diagram of CEs and CAs of the SGP REA. The framework highlights the primary CAs, ecological systems, and CEs that will be evaluated as a part of the SGP REA (fig. 6). The climate and physiography of the ecoregion limit where species and communities occur on the landscape, and influence the dynamics and spatial distribution of communities. Both natural and anthropogenic CAs alter the dynamics and spatial distribution of communities across the ecoregion. Feedback and interactions (such as competition, predation, flows of energy, and species movements) occur within and among terrestrial and aquatic systems and between CEs and CAs (Miller, 2005).

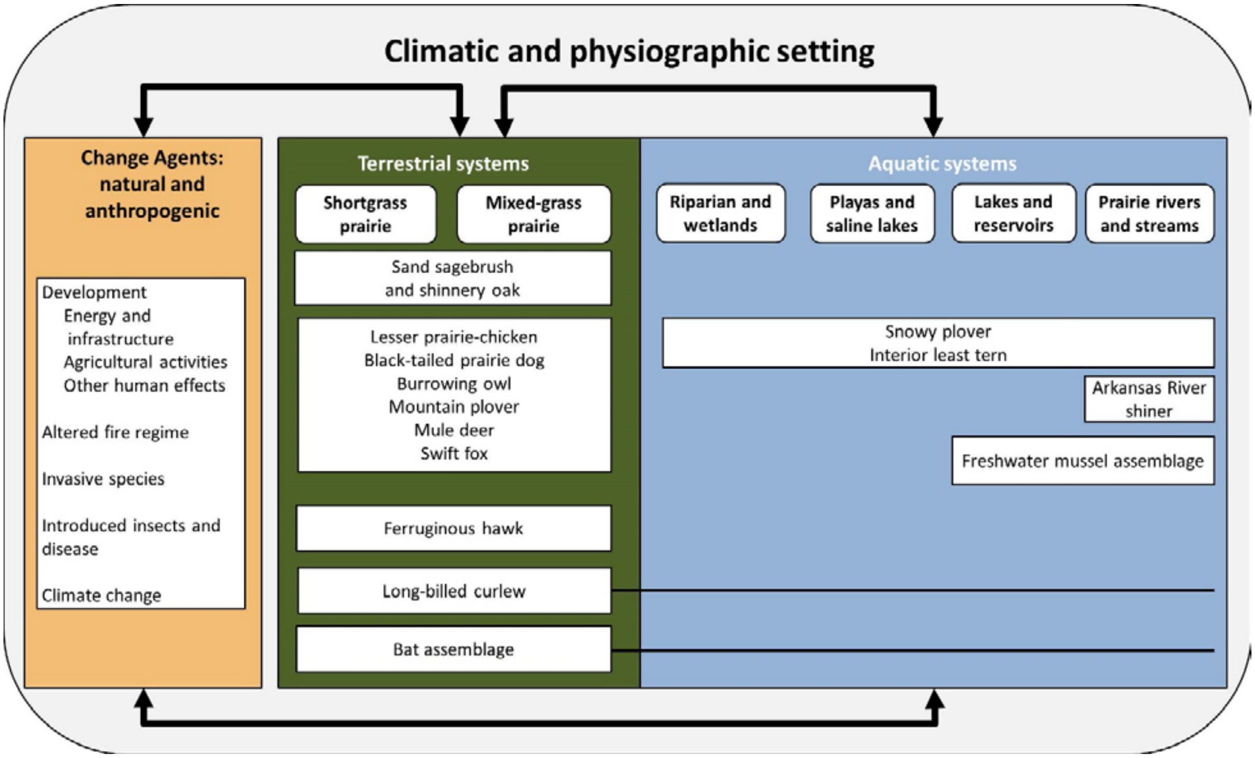

Figure 6. Organizational diagram representing primary components of the Southern Great Plains Rapid Ecoregional Assessment (indicated by colored boxes). Terrestrial (green box) and aquatic (blue box) ecological community Conservation Elements (CE) are shown in the top row of white boxes, and species and species assemblage CEs are shown in boxes below the primary community (or communities) in which they are found. For wildlife species that are strongly tied to terrestrial systems but also use adjacent aquatic habitats, horizontal lines are placed under the aquatic systems also used by that species. The arrows represent the direction of influence and feedback among the ecosystem components. Change Agents (light orange box) are shown in the box on the left.

Management Questions Overview

Description and Criteria

Management Questions (MQs), developed by the AMT and Technical Team for the ecoregion, identify the information needed for addressing land-management responsibilities, including land-use planning, developing best-management practices, authorizing uses, and establishing priorities for conservation and restoration. The MQs help to focus the REA process and ensure that the most relevant datasets are compiled, analyzed, and summarized. The MQs may pertain to CEs and CAs.

Identification and Review of Candidate Management Questions

The initial list of MQs was developed by compiling issues and questions suggested by the AMT and Technical Team during the initial workshop and subsequent webinars. To ensure that we adequately captured issues about each CE, we solicited Technical Team members with a management question matrix. Issues were not always posed as a question, although MQs refer to any management question or management issue. All responses were compiled and organized thematically, grouped by CE and CA. We delineated general "what" or "where" questions associated with each CE that were not affiliated with a specific CA. We also characterized three categories of MQs not affiliated with specific CEs: general key ecological attribute questions, general CA questions, and integrated management questions. We preliminarily characterized each MQ into one of four status categories, as follows.

- To be addressed in Phase I CE Packet.
- To be addressed in Phase II (depending on data availability).
- Unknown at this time whether MQ can be addressed or partially addressed in Phase II.
- Out of scope/cannot be addressed.

This MQ categorization effort provides sideboards on the direction of management issues as the REA progresses. Please note the status of each issue may change in Phase II of the assessment. The AMT members identified priority management issues listed in the matrix during a webinar in May 2014. This information is included in appendix B.

Next Steps for Management Questions

Additional evaluation of the MQs will be conducted in Phase II to finalize the questions or types of questions that will be addressed. The Phase II work plan will provide recommendations on how the MQs should be addressed. The AMT and Technical Team will have the opportunity to contribute to the process to address MQ recommendations. One option that other REAs have used is to develop a suite of core MQs that will be applied to each CA and CE. This is an efficient method to address numerous questions in an objective manner and allows the focus to shift to more complex integrated MQs. Discussion is needed to finalize and prioritize MQs in the next phase.

Overview of Assessment Framework

We will outline the assessment process in the Phase II work plan. We will begin compiling available datasets for the project area and assess the quality of the data and metadata. We will utilize and expand on the geospatial library that has been created by the BLM. We anticipate refining MQs based on input from the AMT and Technical Team. We will identify a suite of core MQs that will be applied to each CE. Example core MQs and the proposed map or metric to address each are summarized below.

- What and where are the key ecological attributes?
  —Individual maps will identify the amount and distribution of key ecological attributes for each CE.
- What and where are the CAs?
  —Individual maps will identify an index for each CA.
- Where do the CAs overlap with the key ecological attributes?
  —The overlap of key ecological attributes with the CA index will be calculated.
- How do the CAs affect the key ecological attributes?
  —The amount and distribution of key ecological attributes impacted by the CA index will be calculated.

Integrated MQs, which will be differentiated from core MQs, will be addressed by combining the metrics developed for core MQs. An example integrated MQ might be, “Where are priority areas for conservation, restoration, or development?” The question can be addressed by combining maps of the key ecological attributes for each CE with an integrated threat index to identify key ecological attributes at risk for each CE. Integrated MQs will be formulated during the refinement process of Phase II.

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Section II. Conservation Elements

Chapter 4. Mixed-Grass Prairie

By Daniel J. Manier, T. Luke George, and Cynthia P. Melcher

Key Ecological Attributes

Distribution and Ecology

The mixed-grass prairie ecoregion occurs in the central third of the Great Plains, where it forms a transition zone of mid-height grasslands between tall-grass prairie to the east, shortgrass prairie to the west (about 300 kilometers east of the Rocky Mountains), juniper-oak savanna to the south, and aspen parklands to the north (Bragg and Steuter, 1996). Some authorities further divide this region into the Sandhills prairie and the southern and northern mixed-grass prairies (Bragg and Steuter, 1996). In the SGP ecoregion, the southern mixed-grass prairie (SMGP) generally includes the grasslands of south-central Nebraska, the central third of Kansas and Oklahoma, small parts of the eastern Texas panhandle, and the western half of north-central Texas (fig. 7).[1] The zones of transition between the shortgrass and the tallgrass shift westward and eastward as the region's highly variable climate cycles in and out of deep and (or) prolonged drought (Knight, 1994; Weaver and others, 1996).

Mean annual precipitation across the entire mixed-grass prairie increases from 40 to 80 cm along a west-to-east gradient; to a lesser extent, there is also a north-to-south gradient along which precipitation increases (Weaver and others, 1996). More than 60 percent of the precipitation typically falls as rain during the growing season, although drought during the growing season is common (Bragg and Steuter, 1996). Average annual temperatures in the SMGP range from approximately 10 ºC in the north to 15 ºC in the south (Weaver and others, 1996). These climatic gradients are primary drivers of variation in primary productivity across the SMGP, where estimated annual standing crop ranges from 180−450 kilograms per hectare (kg/ha) in the south to 200−420 kg/ha in the north (Bragg and Steuter, 1996).

The SMGP is characterized by two major plant associations: bluestem-grama (*Andropogon gerardii, Schizachyrium scoparium, Bouteloua curtipendula,* and *B. gracilis*) throughout most of the region, and mesquite-buffalograss (*Prosopis* spp.–*Bouteloua dactyloides* [formerly *Buchloe dactyloides*]), primarily in the Rolling Plains; the overall plant community composition, however, varies widely from region to region within the SMGP (Bragg and Steuter, 1996). Depending on location and conditions, dominant short to mid-height grasses typically include buffalograss (*Bouteloua dactyloides* and blue grama (*B. gracilis*); common mid-height grasses include sideoats grama (*B. curtipendula*) and little bluestem (*Schizachyrium scoparium*); and tall bluestem, are usually dominant under relatively mesic conditions, whereas shortgrasses and drought-tolerant mid-height grasses, such as buffalograss, blue grama, hairy grama (*B. hirsuta*), and sideoats grama, are usually dominant in relatively dry locations (Bragg and Steuter,

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[1] Throughout this report, we limit most of our discussion to the SMGP; however, where information specific to the SMGP was not available, we refer to the mixed-grass prairie.

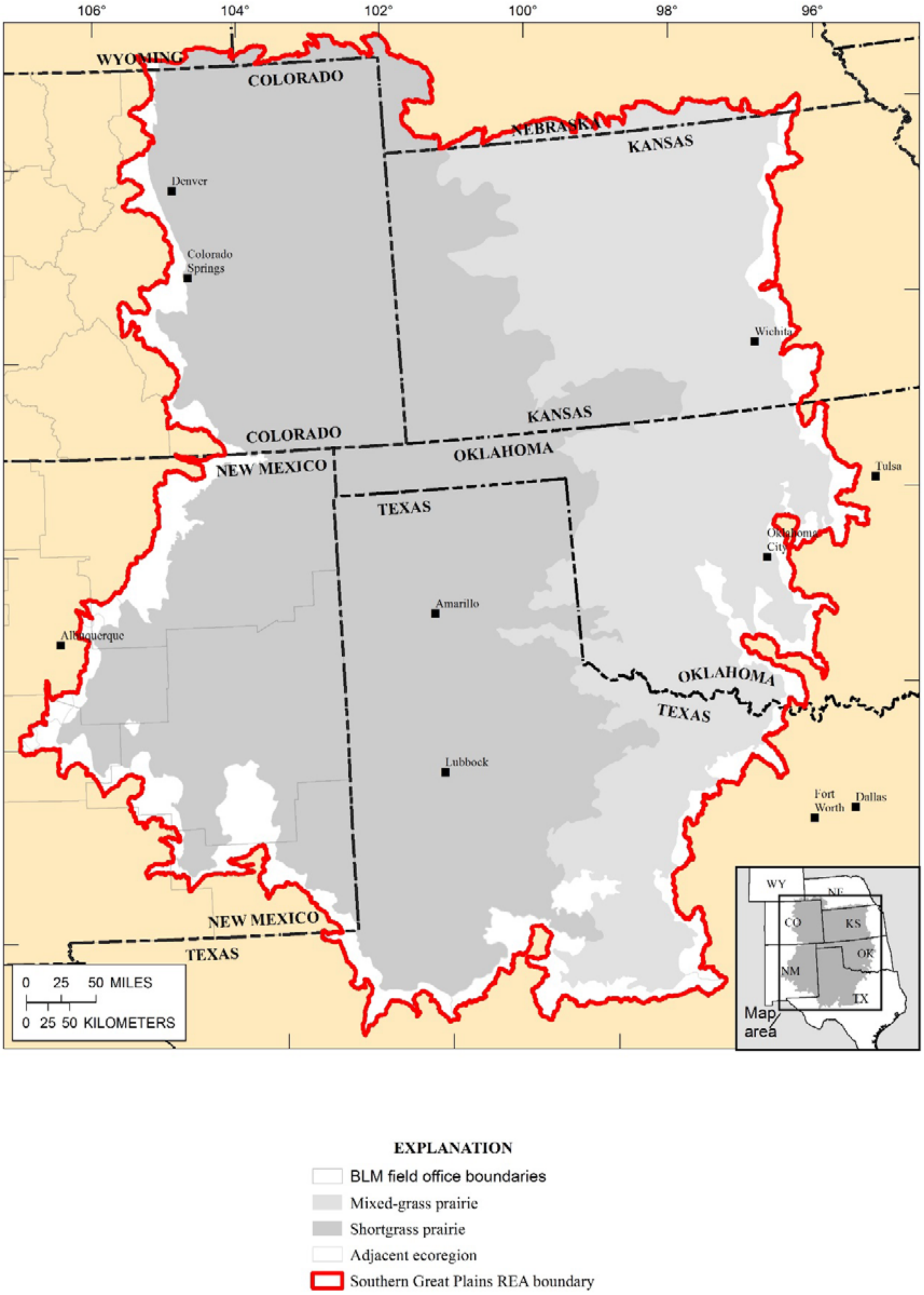

Figure 7. Extent of mixed-grass prairie based on Omernik's (1987) Level III Ecoregion classification (U.S. Environmental Protection Agency, 2013). (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (BLM, Bureau of Land Management; REA, Rapid Ecoregional Assessment)

1996), especially where soils have a significant clay component. Grasses contribute the largest portion of the primary productivity in the mixed-grass prairie, but forbs may comprise as much as 25 percent of the standing crop and contribute significantly more to the overall plant diversity (Bragg and Steuter, 1996). Forb species of the mixed-grass prairie commonly include blanketflowers (*Gaillardia* spp.), evening primroses (*Oenothera* spp.), dozedaisies (*Aphanostephus* spp.), asters (*Symphyotrichum* spp.), penstemons (*Penstemon* spp.), dotted blazing star (*Liatris punctata*), and annual sunflower (*Helianthus annuus*) (Bragg and Steuter, 1996; Natural Resources Conservation Service, 2014).

Although most of the SMGP is dominated by grasses and forbs, shrubs and trees can be dominant where conditions are suitable. Common tree species of the SMGP include redberry juniper (*Juniperus pinchotii*), and common shrubs include honey mesquite (*Prosopis glandulosa*), a predominant feature of the plant community on the Rolling Plains of Texas (Bragg and Steuter, 1996), shinnery oak, and sand sagebrush. Although it remains uncertain as to whether shinnery oak and sand sagebrush are more prevalent in the SMGP or the shortgrass prairie (Phase 2 of this assessment will entail mapping their distributions), both species are important components of the SMGP system, as they provide crucial habitat to some of the region's wildlife species of greatest concern. Shinnery oak is found on sandy soils from western Oklahoma and northern Texas south to central Texas and southern New Mexico. Important codominant species of shinnery oak communities include dropseeds (*Sporobolus* spp.), little bluestem, sand bluestem (*Andropogon hallii*), giant sandreed (*Calamovilfa gigantea*), and Havard's panicgrass (*Panicum havardii*). Sand sagebrush occurs from eastern Wyoming and western Nebraska, south through Texas and parts of northern Mexico. Like shinnery oak, sand sagebrush is strongly associated with deep, sandy soils and often co-occurs with shinnery oak. The understory of sand sagebrush communities is generally dominated by shortgrasses, such as blue grama and buffalograss; taller grasses, including little bluestem, needle-and-thread (*Hesperostipa comata*), sand dropseed (for example, *S. cryptandrus*), prairie sandreed (*C. longifolia*), and sand bluestem may co-dominate where conditions are sufficiently mesic. These shrub communities are important because the plant and animal communities associated with them differ from those that associate with the surrounding grasslands.

Landscape Structure and Dynamics

The mixed-grass prairie landscape is characterized by rolling plains interspersed with broad riparian valleys and scattered buttes, escarpments, and other topographic features; elevations gradually decline from 1,130 m in the west to 400 m in the east (Bragg and Steuter, 1996). Region-wide climate patterns drive the large-scale patterns in vegetation types and community structures. The transition from shortgrass to mixed-grass prairie occurs where the rain-shadow effect of the Rocky Mountains gradually yields to midcontinental climate patterns, including frontal systems and summer convective storms that deliver more annual moisture than what is delivered farther west. Overall, this moisture gradient promotes shorter vegetation structure in western portions of the mixed-grass prairie, whereas the extra moisture that falls farther to the east promotes high productivity of taller grasses. In addition to precipitation, however, the mixed-grass system is heavily influenced by topographic features, soil textures, temperature gradients, disturbances (fire, herbivory, and drought), and land-use history, the various combinations and interactions of which result in a mosaic of plant community composition across the landscape (for example, Martinson and others, 2011; Winter and others, 2011; Myster, 2012). As such, taller grasses can grow well within the rain shadow (Archer, 1984), and shorter grasses can dominate central portions of the mixed-grass prairie (Smith, 1940). Overall, the patchy distribution of heterogeneous habitat structures (that is, short, mid-height, mixed, and tall) is an important characteristic of the Great Plains. This heterogeneity drives much of the Great Plains biodiversity, and it is crucial for meeting the varied habitat requirements of many native prairie species. Indeed, a key habitat attribute for some of the regionally important wildlife species, including lesser prairie-chicken and swift fox, is the juxtaposition of different vegetative structures and community compositions created by local topographic features and disturbances (Archer, 1984; Winter and others, 2011).

Soil texture in the SMGP can range from clayey to loamy to sandy, which strongly influences vegetation type, largely due to the significant effect of soil texture on water infiltration and retention. Moisture easily and rapidly infiltrates sandy soils, whereas clay soils tend to resist moisture infiltration; loams fall between these extremes.[2] For example, the relatively drought-tolerant shortgrasses tend to dominate clayey soils, whereas shinnery oak and sand sagebrush require deep, sandy soils (Peterson and Boyd, 1998; U.S. Forest Service, 2014). Topographic features further influence vegetation structure by varying aspect, elevation, parent soil types and soil depth, and water accumulation. In turn, these influences can create myriad microhabitats where temperatures, moisture levels, substrates, and insolation can vary widely from those of surrounding habitats. Topographic features, such as breaks and riparian systems, also can serve as natural fire breaks that permit woody species, such as juniper and mesquite, to become locally dominant (Bragg and Steuter, 1996). Other factors that often promote woody species dominance include landscape features that discourage or exclude the presence of herbivores; greater, more reliable moisture levels such as those found in and near drainage systems; and soils that allow deeper infiltration and retention of moisture, such as sandy loams.

Because fire scar data are scant in landscapes dominated by herbaceous vegetation, there is little information about historical fire regimes in the mixed-grass prairie. It is believed, however, that fire suppressed woody species, removed fine fuels, and recycled nutrients. Historically, humans frequently used fire to manage habitat and drive game in the mixed-grass prairie system (Bragg and Steuter, 1996). Indeed, the mosaic of vegetation patterns and the ecological processes across the mixed-grass landscape are likely the result of interactions between local factors ("site potential"), herbivory, and fire (Limb and others, 2011; Winter and others, 2011). Although shinnery oak and sand sagebrush represent an important part of the structure and diversity in the mixed-grass prairie mosaic, there has been a perception that shinnery oak has undergone recent expansions at the expense of quality grassland habitats; in turn, this prompted several decades of management activities intended to remove and restrict the growth of shinnery oak. Although fire (prescribed and wild) can have strong, positive effects on the productivity of grasses and shinnery oak, treatment with chemical herbicides (such as Tebuthiuron) will kill the oak. Moreover, research has shown that recent expansion of shinnery oak is unlikely to have occurred because typically the root structures are extremely old (hundreds to thousands of years) and successful reproduction from seed is rare to nonexistent. Therefore, although shinnery oak responds well to disturbances that leave roots (mostly) intact, it does not expand readily to recolonize previous or new habitats. Sand sagebrush represents important habitats for several endemic and regionally important species, including the lesser prairie-chickens and the massasauga.

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[2] For a more detailed discussion of soil type and its influence on vegetation, see the "Landscape Structure and Dynamics" section in Chapter 5, "Shortgrass Prairie."

Whereas climate and soils have long-term (slow change) determinant effects on grassland systems, disturbances such as fire, herbivory (vertebrates and invertebrates), and domestic grazing have important short-term effects (year-to-year differences), which can become long-term effects (see discussion of shinnery oak below). Restoration of historical disturbance processes, including fire, grazing or both, in prairie management has been adopted by many land managers across the SGP region based on the understanding that these processes benefit species composition and diversity, primary productivity, habitat structure, and control of exotic species (Brudvig and others, 2007). Because most fuels on the Great Plains are herbaceous (standing crop and litter), interactions between site-based factors (site potential), herbivory, fire, and even drought are instrumental in maintaining the mosaic that composes the mixed-grass landscape (Limb and others, 2011; Winter and others, 2011). Modern understanding of these relations provides better understanding of the historic effects of fire-suppression actions and moderate to heavy grazing intensities on prairie fire regimes; namely, past management has resulted in a process-altering reduction in the frequency of fire in most mixed-grass prairie ecosystems (Brudvig and others, 2007; Winter and others, 2011).

In the short term, all disturbance types can reduce the standing crop, although in the longer term, the litter reduction and accelerated nutrient cycling brought about by disturbances may stimulate growth and increase the vegetation structure (Bragg and Steuter, 1996). Herbivory and fire alone may produce one set of effects, whereas herbivory and fire together may produce another set of effects. Prior to European settlement, the major herbivores that had a significant influence on the vegetative communities of the mixed-grass prairie included both mammalian and invertebrate species, especially grasshoppers. American bison (*Bison bison*), black-tailed prairie dogs (*Cynomys ludovicianus*), elk (*Cervus elaphus*), mule and white-tailed deer (*Odocoileus hemionus* and *O. virginianus*), and pronghorn (*Antilocapra americana*) all ranged across the mixed-grass prairie; indeed, the mixed-grass prairie was central to the bison's historical range (Bragg and Steuter, 1996). The varying degrees of grazing intensity and frequency imposed by these large, grazing mammals influenced the vegetation community structure, composition, and habitat conditions, in turn affecting other species that preferentially used or avoided heavily, moderately, or lightly grazed areas. Although bison were extirpated from the SMGP, they have been reintroduced into protected areas (for example, the Wichita Mountains and Maxwell National Wildlife Refuges). Mule deer are relatively rare in the SMGP, as the region is largely beyond the species' distribution. White-tailed deer, however, remain common in the SMGP, and although they are found primarily in woodland habitats, they may be found in mixed-grass habitats adjacent to woodlands and riparian areas; their habitat use and diet may vary based on forage availability. Elk were largely eliminated from the plains in the 1800s, and although populations have been reestablished in some areas (for example, the Wichita Mountains National Wildlife Refuge), only a few isolated populations persist in Oklahoma and Kansas. Pronghorn are more common farther west in the shortgrass prairie, but they can occur in mixed-grass habitats dominated by shorter statured species; isolated populations persist in eastern Kansas and Texas. Black-tailed prairie dogs (*Cynomys ludovicianus*) (Chapter 20) are often a controversial co-occupant of the mixed-grass prairie because of their perceived effect on forage availability. In addition to potentially important roles in soil-surface conditions and nutrient cycling, prairie dog colonies represent important habitats (especially their burrows, which provide nesting and denning sites) for several native species of conservation concern. Moreover, the burrowing activities of prairie dogs and other fossorial animals also play potentially important roles in soil surface conditions and nutrient cycling.

Associated Species of Management Concern

Mixed-grass prairie habitats of the SGP support or historically supported a number of breeding, wintering, and year-round species also evaluated for the SGP Rapid Ecoregional Assessment. They include the burrowing owl (*Athene cunicularia*) (Chapter 12), ferruginous hawk (*Buteo regalis*) (Chapter 13), lesser prairie-chicken (*Tympanuchus pallidicinctus*) (Chapter 15), black-tailed prairie dog (Chapter 20), mule deer (Chapter 21), and swift fox (*Vulpes velox*) (Chapter 22). The prairie-chicken, prairie dog, and fox are year-round residents of the SMGP. The burrowing owl occurs in breeding season (and to some extent year-round in extreme southern portions of the SMGP), the ferruginous hawk is primarily a winter resident, and the mule deer's range overlaps the SMGP only in western Kansas. Additional species considered priorities by Great Plains Landscape Conservation Cooperative and which occur or historically occurred in the SMGP include the American bison and (likely) American burying beetle (*Nicrophorus americanus*) (year-round residents); upland sandpiper (*Bartramia longicauda*), grasshopper sparrow (*Ammodramus savannarum*), Cassin's sparrow (*Peucaea cassinii*), and lark bunting (*Calamospiza melanocorys*) (breeding and migration; Cassin's sparrow and lark bunting also may winter in extreme southern portions of the SMGP); and Harris's sparrow (*Zonotrichia querula*) (winter resident). Overall, grassland birds are considered the fastest declining suite of birds in North America. Additionally, several species of migratory waterfowl nest and forage in the mixed-grass prairie uplands that surround the region's wetlands (playas in particular). Although black-tailed prairie dogs serve as prey for some species, their burrows also provide crucial nesting habitat for burrowing owls and denning sites and (or) escape cover for black-footed ferrets (*Mustela nigripes*) and swift foxes. The shinnery oak and sand sagebrush communities provide habitat for several endemic and regionally important wildlife species, including lesser prairie-chicken, mule deer, dunes sagebrush lizard (*Sceloporus arenicolus*), and massasauga (*Sistrurus catenatus*). It is important to note that the lizard is not found in human-altered landscapes.

Change Agents

Development

The structure and function of mixed-grass prairie, as well as the occurrence of keystone species, have been significantly altered by anthropogenic CAs. The CAs having the greatest influence include agricultural cultivation of large areas and the associated redistribution of surface water and reduced groundwater levels; altered fire regimes and the associated increased woodland area; development of extensive transportation corridors and the associated inroads for invasions and expansions of exotic species; energy-extraction infrastructure and urbanization; and effects of land-management practices (grazing in particular) within discrete management units (Bragg and Steuter, 1996). These activities and land uses are resulting in cumulative effects on habitat condition and distribution.

Energy and Infrastructure

The SMGP landscape and its biotic communities are being affected by a variety of energy-development activities, including oil and gas development and wind farming. Energy-extraction infrastructure, such as gas and oil well pads and wind turbines, along with access roads and distribution systems, such as pipelines and powerlines, can eliminate habitats and degrade and fragment habitats at a landscape scale. Although there has been little research designed to evaluate specific effects of energy development on prairie biomes, it is clear from work in other systems (such as sagebrush steppe) that corridors and patchworks of surface disturbance created during energy development result in direct habitat loss and fragmentation; they also lead to indirect losses, such as the inroads for invasions of nonnative species created by surface disturbance and transportation corridors (Bergquist and others, 2007; Finn and Knick, 2011; Knick and others, 2011).

Activities associated with energy development, such as drilling and hydraulic fracturing, also can affect the distribution and levels of surface water and groundwater. For example, an average of 4.8 million gallons of water are typically used to fracture natural gas shales in every horizontal well drilled (Freyman, 2014), including the Woodford and Barnett shales being developed in Oklahoma and Texas, respectively (U.S. Energy Information Administration, 2011). In these regions, however, the water stress is already considered medium to extremely high, and in some areas of the SMGP hydraulic fracturing is contributing significantly to further depletion of the groundwater resources (Freyman, 2014). Also of potential concern where fossil fuels are being developed is the possibility of spills and other accidents, as well as intentional dumping of contaminated fluids, that result in contamination of land and water resources. Fluids associated with energy development and its waste products may contain high levels of salinity, radioactivity, hydrocarbons, and (or) other contaminants that can alter vegetation, soils, and important processes, such as nutrient cycling (Irwin and others, 1996; Kharaka and Otton, 2003; Fisher and Sublette, 2005; U.S. Environmental Protection Agency, 2012).

Many large wind farms have already been developed throughout the SMGP, with many more being proposed. Indeed, wind energy development has become the source of significant concern in terms of its potential effects on habitats and wildlife of the Great Plains. As a result, the U.S. Fish and Wildlife Service initiated work on a Great Plains Wind Energy Environmental Impact Statement and Habitat Conservation Plan (U.S. Fish and Wildlife Service, 2011) that encompasses almost the entire mixed-grass prairie region, as well as parts of the shortgrass and tallgrass prairie regions. Potential effects include habitat loss, degradation, and fragmentation. As with development of fossil fuels, the footprint of wind energy development also provides inroads for introductions or expansions of exotic species.

Agricultural Activities

Based on a comparison of estimated historical and current (mid-1990s) distributions of native mixed-grass prairie vegetation, it was estimated that only 8 percent (bluestem-grama) to 60 percent (wheatgrass-needlegrass and mesquite-buffalograss) of the native mixed-grass prairie communities remained, largely the result of agricultural conversion (Bragg and Steuter, 1996). Although some wildlife species, such as geese and mule deer, will forage in croplands, the natural habitats preferred by most native grassland wildlife are removed or greatly reduced when native prairie is converted to agriculture. From a landscape perspective, however, there may be a gradient of habitat suitability across different geographies, land cover types, and land uses; thus, even if agricultural lands do not provide all the resources normally provided by natural habitats, they can be used by native species for some activities. For example, most native species can safely move through agricultural landscapes but would have difficulty moving through urban or suburban areas where exposure to predators, vehicles, and human activities presents greater risks. For some species, the size and composition of agricultural fields may determine the level of risk in traversing such a landscape, which, in turn, has direct implications for habitat connectivity and animal dispersals and migration movements. Mismanagement of croplands and rangelands also may alter essential soil and hydrological resources, as observed during dramatic erosion events that occurred in the early 1900s (Kothmann, 1996).

Great Plains ecosystems evolved under the influences of grazing by large ungulates, birds, and invertebrates, especially grasshoppers; indeed, it has been shown that large herbivores have been present in the Great Plains region for at least two million years (Collins and Barber, 1985; Kothmann, 1996). Precise effects of the widespread switch from wild-ranging herds and individuals of ungulates to the closely managed livestock grazing systems of today cannot be quantified accurately (Lauenroth and others, 1993). Despite the lack of quantitative information, however, published studies and common practices confirm that, despite clear differences and potential limitations, the vegetation and soils of the Great Plains are generally robust under regimes of moderate livestock grazing; indeed, exclusion of grazing may be considered a disturbance to ecosystem processes (Kothmann, 1996). On the other hand, high stocking rates and early (in the growing season) and continuous or long-duration grazing can result in the elimination of highly palatable plant species and the increase of woody, invasive plant species or other undesirable vegetation (Smith, 1940). Grazing also can determine the relative composition of dominant grasses and the productivity of mixed-grass ecosystems (Collins and Barber, 1985). Light to moderate grazing may increase or decrease productivity and diversity, whereas heavy grazing generally leads to decreased diversity (Collins and Barber, 1985). Although stocking rates are recognized as having significant effects on the composition and structure of grasses, research also indicates that climate and local soil conditions have strong, primary influences on the plant communities, and the responses of plants to grazing are fundamentally influenced by site-level conditions and annual precipitation (Gillen and Sims, 2006; Winter and others, 2011).

Altered Fire Regimes

Today’s fire-suppression practices likely have reduced fire frequency, and contemporary grazing management practices likely have reduced fuel loads where stocking rates have been high (Brudvig and others, 2007; Winter and others, 2011). Combined, these factors likely have altered mixed-grass prairie function and processes, and restoring fire as a disturbance agent is expected to help maintain native vegetation communities, support nutrient cycling, and generally improve habitat conditions for both wildlife and domestic livestock (Vermeire and others, 2005; Limb and others, 2011).

Observed changes in the structure of shinnery oak communities exemplify how fire and herbivory may interact on the prairie landscape, the role that fire places in determining community structure, and the responses of native grasses and shrubs to fire. The apparent expansion of shinnery oak across its range in the SGP ecoregion led to widespread efforts to control it with herbicides, thus reducing the cover of this regionally important species (Peterson and Boyd, 1998; Boyd and Bidwell, 2002). Recent studies, however, suggest that the apparent changes in shinnery oak communities were more likely due to fire suppression and the effects of livestock grazing on the structure of tall grasses (Peterson and Boyd, 1998). Based on indirect evidence, historical fire-return intervals in stands of shinnery oak were estimated at 5–10 years (Boyd and Bidwell, 2002), and restoration of fire in shinnery oak communities (especially frequent fires during the season of plant dormancy) can have immediate beneficial effects on the overall percent cover, height, and composition of both woody and herbaceous components of the shinnery oak community (Boyd and Bidwell, 2002). Similarly, burning can stimulate a rapid, beneficial response of vegetation in sand sagebrush associations, which in turn have positive effects on habitat heterogeneity and the distributions of animals (Winter and others, 2011).

Invasive Species

There are relatively widely distributed exotic and native invasive plant species in the SMGP, but not all of them attain dominance or alter ecosystem structure and function (Weaver and others, 1996). Plant species of concern or potential concern in more northern areas include annuals and perennials, such as field bromes (*Bromus arvensis*) and smooth bromes (*B. inermis*), crested wheatgrass (*Agropyron cristatum*), nonnative bluestems (*Bothriochloa* spp.), and forbs such as sweet clover (*Melilotus* spp.), knapweed (*Centaurea* spp.), kochia (*Bassia* spp.), and alyssum (*Alyssum* spp.) (Weaver and others, 1996). Cheatgrass (*B. tectorum*), which has invaded and is altering much of the sagebrush steppe, is well-distributed throughout the SMGP, especially in Kansas and eastern Colorado (Natural Resources Conservation Service, 2014), but its potential for altering the community structure and processes of mixed-grass prairie systems is not yet understood. It could have greater effects in communities dominated by fire-intolerant shrubs, such as shinnery oak, than in grassland-forb systems. Although introduced species like Kentucky bluegrass (*Poa pratensis*), smooth brome, and weeping lovegrass (*Eragrostis curvula*) persist in the southern plains, native invaders, such as honey mesquite (*Prosopsis glandulosa*), creosote bush (*Larrea tridentata*), and juniper species (*Juniperus* spp.), present more serious management challenges because of their persistence and modification of soil-vegetation relations (Weaver and others, 1996). Where fire and mowing are used to reduce these woody invaders (Risser, 1996), the native grass community can be expected to respond robustly (Collins and Barber, 1985; Brudvig and others, 2007), indicating that even though invasive species can be a management challenge on prairie landscapes, there are options for control, especially where desirable native species occur.

In some regions of the SGP, there is concern about range expansions of feral and wild exotic swine (including feral pigs and wild boars introduced from Europe), as their activities, including wallowing and rooting, can alter habitats significantly. In 1982, the range of these animals included only extreme southern portions of the SMGP in Texas; by 2010, however, they had expanded throughout the SMGP and even into states bordering Canada (Animal and Plant and Health Inspection Service, 2013). It is unclear whether these expansions represent scattered individuals or established populations, but their populations are expected to continue to expand. The extent to which feral and wild swine could have ecosystem-altering effects on the SMGP also remains unclear, although these animals tend to prefer bottomlands of drainage systems and along rivers (Animal and Plant and Health Inspection Service, 2013; Iowa State University Forestry Extension, 2012).

Insects and Disease

Although grasshoppers and other insect herbivores are often considered pests of the prairie, these species also represent an important food source for birds (Knopf, 1996) and reptiles. Grasshoppers are common and diverse across prairie grasslands (Branson and Sword, 2008), with herbivory patterns differing among species (grasses versus forbs); like ungulates, grasshoppers also can recognize and respond to differences in forage quantity and quality (Joern and others, 2012). Great Plains grasshopper populations respond to several primary drivers, including climate cycles, ungulate grazing, and recent fire history (especially years since fire).

Grasshopper numbers are generally greater in wetter years, presumably because more precipitation promotes greater productivity of plants (grasshopper forage), and grasshopper maturation tends to be faster during warmer springs, thus increasing the potential for greater grasshopper abundance later in the growing season. Strong connections between population cycles and environmental drivers, however, remain elusive (Jonas and Joern, 2007).

Climate Change

Current climate change models are projecting a range of potential shifts across the SGP, including increasing temperatures and more intense rainfall events despite a decrease in average amounts of total annual precipitation (Karl and others, 2009). Although topography, soils, disturbance history, and land-use activities have important determinant effects on community composition, structure, and productivity of the mixed-grass prairie, gradients in temperature (Martinson and others, 2011) and precipitation patterns (Myster, 2012) are the primary drivers of species' distributions and their relative dominances. Furthermore, despite the fact that drought and summer heat have long been a part of the SMGP climate patterns, primary production and water-use efficiencies of mixed-grass prairie species vary widely (Vermeire and others, 2005). Therefore, changes in overall moisture availability, seasonality, and rates of evapotranspiration driven by increasing temperatures and altered patterns in precipitation may affect the competitive relationships between plant species, which in turn could alter their relative abundances. For example, although blue grama and buffalograss are important components of mixed-grass prairie communities, they are not dominant across the SMGP; however, changes in the timing and amount of precipitation, accompanied by sustained or increasing temperatures, may promote increased abundance or more widespread dominance of these highly drought-tolerant species at the expense of the taller, less drought-tolerant grasses.

Also important to this discussion is the effect of climate and CO_2 levels on photosynthesis, primary productivity, and plant community structure and composition. The abundance of species that use the C_4 photosynthetic pathway (that is, "warm-season species"), including blue grama and buffalograss, is positively correlated with July temperatures, and water-use efficiency in C_4 species can be up to two times greater than that of C_3 (or "cool-season") species (monocots in particular) (Ehleringer and others, 1997), such as western wheatgrass and prairie Junegrass (*Koeleria macrantha*). Therefore, it is possible that C_3 species will become less productive as drought and temperatures increase (Vermeire and others, 2005), particularly if winter and spring moisture needed for the early-season growth of C_3 species becomes more limiting. On the other hand, C_3 species are generally favored under higher levels of CO_2 (Dukes, 2000), whereas C_4 species are generally favored under low concentrations of CO_2 (with a threshold of less than 500 parts per million [ppm] above which C_4 species lose their advantage) (Ehrlinger and others, 1997). Since 1958, the atmospheric concentration of CO_2 has risen from less than 300 to 400 ppm, with levels projected to go well beyond 500 ppm by the end of the 21st century (Dukes, 2000); this rise has the potential to somewhat offset the competitive advantage of C_4 species under increasingly warm conditions. Clearly, changes in seasonal patterns of temperature and moisture, coupled with extreme events and altered storm tracks (Heisler-White and others, 2009), have the potential to drive changes in the distributions and dominance of individual species and the overall composition and structure of mixed-grass prairie communities, but individual species respond differently and sometimes unpredictably (especially C_4 species) to altered environments (Dukes, 2000). Also of importance are the potential plant-animal interactions that could result from any shifts in the proportions of C_4 to C_3 species.

Overall, the nutritional quality of C_3 species is greater (more protein content and better digestibility) than that of C_4 species, and although mammalian and invertebrate herbivores differ with regard to their preferences for C_3 or C_4 species, increasing levels of CO_2 are likely to affect plant-animal interactions and overall distributions of C_4 and C_3 species, grasses especially, across the globe (Ehleringer and others, 2002).

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting mixed-grass prairie is illustrated in figure 8. Ecological attributes and CAs identified in the Rapid Ecoregional Assessment are enumerated in tables 7 and 8.

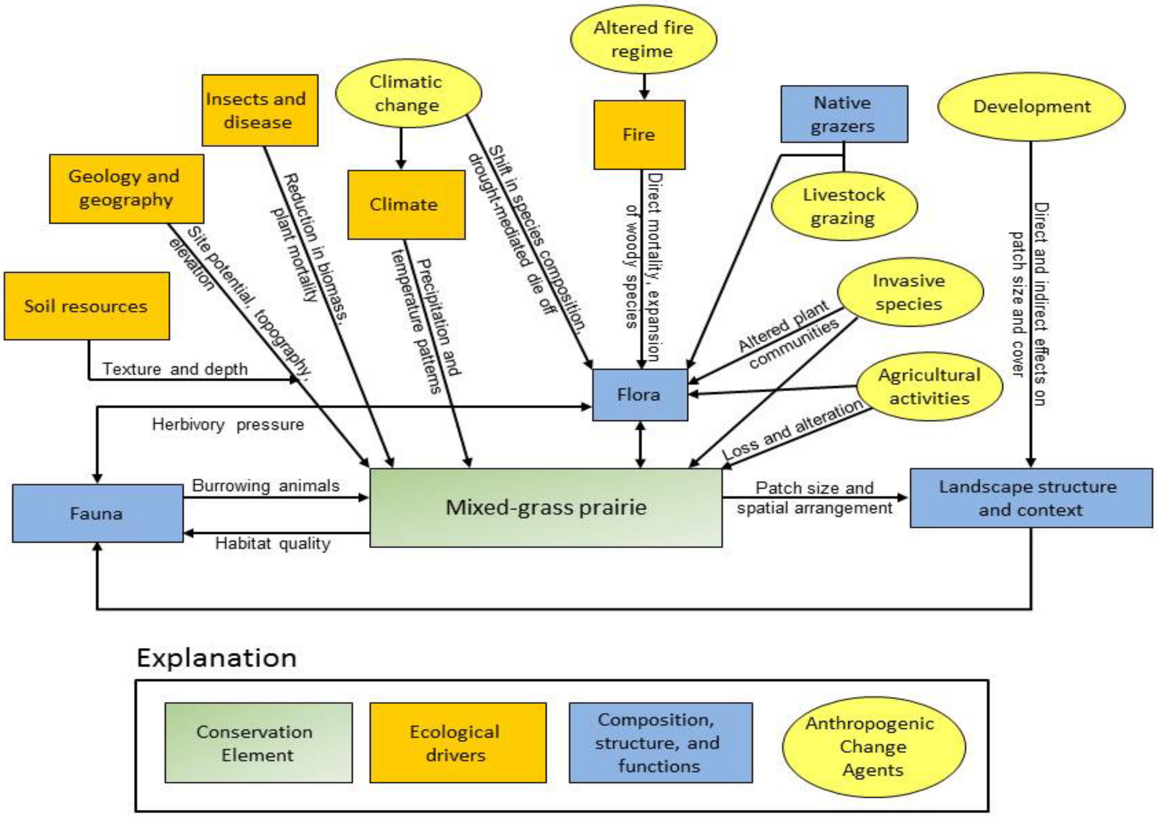

Figure 8. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for mixed-grass prairie in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of mixed-grass prairie are shown in orange rectangles (see also table 7); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 8).

Table 7. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for mixed-grass prairie.

Attribute	Variables
Amount and distribution	Distribution, vegetation condition of mixed-grass prairie (including sand sagebrush and shinnery oak).
Landscape structure	Patch size and spatial distribution (area, connectivity).
Landscape dynamics	Fire regime (frequency and severity), grazing intensity and frequency, climate dynamics.
Associated species management of concern	Abbreviated list with associated conservation elements: mountain plover, burrowing owl, black-tailed prairie dog, swift fox, and lesser prairie-chicken.

Table 8. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for mixed-grass prairie.

Attribute	Variables
Development (energy and infrastructure)	Loss and alteration (direct loss of mixed-grass community, decreased productivity, altered vertical structure), fragmentation (reduction in connectivity), discharge of produced waters.
Development (agricultural activities)	Loss and alteration (direct loss of shortgrass community, changes to vegetation communities, changes to soil structure and hydrologic processes [erosion]), fragmentation (reduction in connectivity), herbicide and pesticide application, livestock grazing (reduction of palatable species, changes in productivity and diversity [±]).
Altered fire regime	Alteration (changes to vegetation communities), dynamics (±) in shrubland mosaic (shinnery oak and sand sagebrush), suppression (reduction in and frequency = expansion of woody vegetation [mesquite and creosote]).
Invasive species	Alteration (changes to vegetation communities).
Insects and disease	Presence and effect on population of animal species (for example, chronic wasting disease, sylvatic plague), reduction of biomass or mortality of plant species (for example, elevated populations of beetle grubs, short-horned grasshopper, nematodes).
Climate change	Alteration (changes to species composition), drought effects.

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Chapter 5. Shortgrass Prairie

By Cynthia P. Melcher

Key Ecological Attributes

Distribution and Ecology

The North American prairie system stretches across the central North American landscape between the Rocky Mountains and the Mississippi River and from south-central Canada to northern Mexico. In the western 320 km of this range, the Rocky Mountains cast a significant rain shadow, and in this narrow strip of semiarid landscape lies the shortgrass steppe or shortgrass prairie (Weaver and others, 1996). In the SGP ecoregion, to which we limit most of this discussion, shortgrass prairie occurs across eastern Colorado and New Mexico, the westernmost counties of Kansas, and the panhandles and rolling plains of Oklahoma and Texas (fig. 9). East of this region, the shortgrass prairie transitions into mixed-grass prairie. As indicated in Chapter 4 for the mixed-grass prairie, the zone of transition from shortgrass to mixed-grass shifts westward and eastward as the region's highly variable climate cycles in and out of deep and (or) prolonged drought (Knight, 1994; Weaver and others, 1996).

Mean annual precipitation in the shortgrass prairie is 340 millimeters (mm), about 70 percent of which falls between April and September (Shortgrass Steppe Long Term Ecological Research, 2007). Early spring storms usually bring prolonged, soil-penetrating rains, whereas summer rains are generally intermittent, highly localized thunderstorms. Although drought conditions tend to increase as summer progresses, rainfall may be heavy where these thunderstorms do occur, and interannual variation in precipitation is significant, with annual rains often failing to occur (Weaver and others, 1996; Shortgrass Steppe Long Term Ecological Research, 2007).

From north to south latitudes, the average low−high annual temperatures in the shortgrass prairie increase from approximately 3−18 to 5−25 ºC, respectively (Weaver and others, 1996). Precipitation also increases along the same latitudinal gradient, with annual precipitation averaging 38 cm in the northwest and 61 cm in the southeast portions of the shortgrass prairie. The gradients in temperature and precipitation tend to counteract one another, such that precipitation is approximately 20 percent of potential evapotranspiration throughout the shortgrass prairie (Weaver and others, 1996). The shortgrass prairie is also characterized by persistent winds, which are generally greatest in spring and contribute significantly to the semiarid conditions of this region (Coupland, 1958).

The shortgrass prairie vegetation community is characterized by two codominant grasses: buffalograss (*Bouteloua dactyloides*) and blue grama (*B. gracilis*) (Weaver and others, 1996). These species are well adapted to grazing and can withstand moderate to heavy grazing pressure (Weaver and others, 1996; Hart, 2001). Both grazing pressure and water stress have been selective forces in shaping the relatively short stature of these and other shortgrass prairie plant species (Weaver and others, 1996). Buffalograss grows to heights of 10.15–20.3 cm, and blue grama typically grows 15.25–30.5 cm high. Buffalograss is a sod-forming grass, and blue grama is a bunchgrass that can form open mats of sod. The spaces between bunchgrasses permit other grass or forb species to grow among them, which promotes a diverse mixture of species. Both

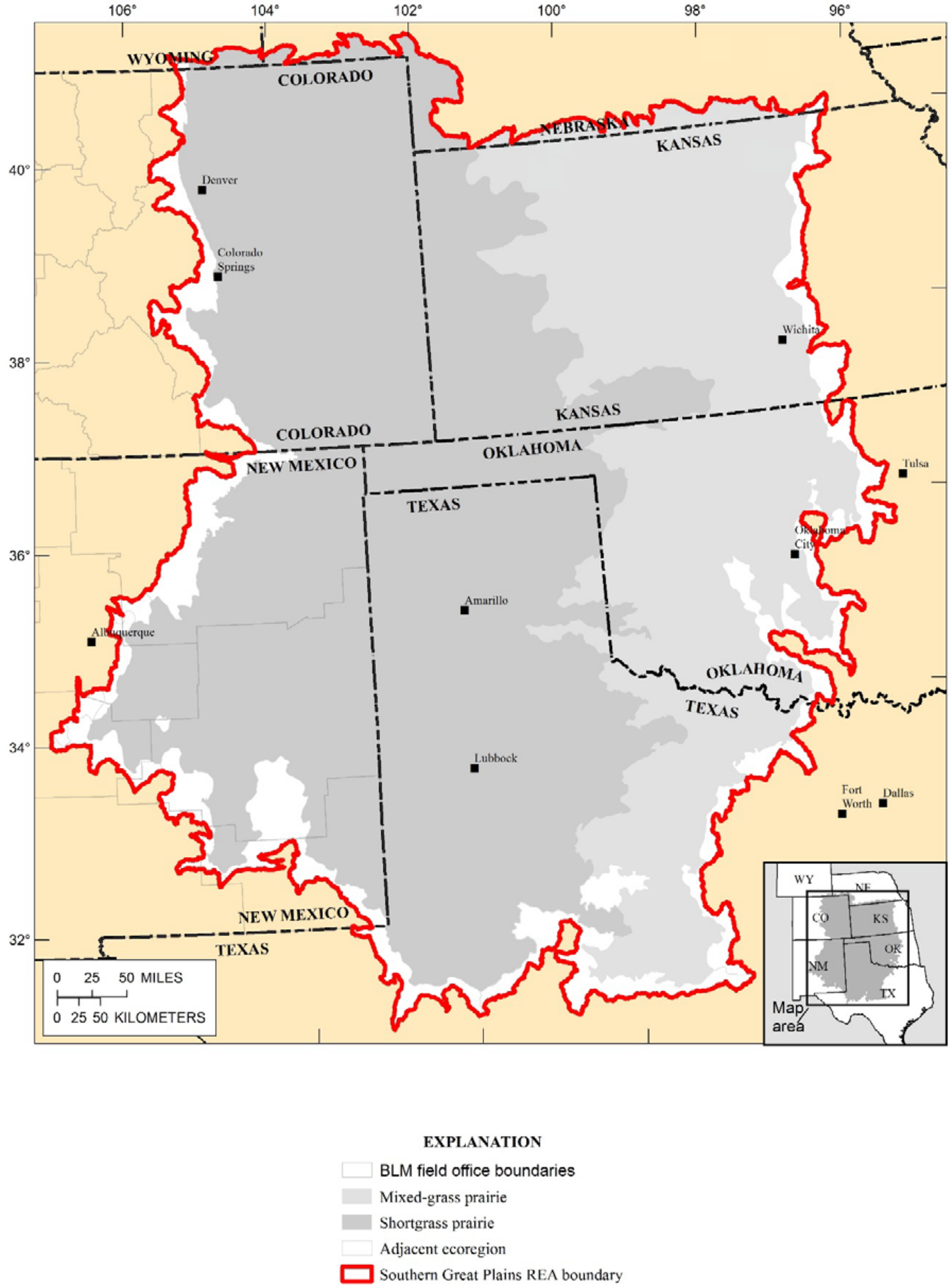

Figure 9. Extent of shortgrass prairie based on Omernik's (1987) Level III Ecoregion classification (U.S. Environmental Protection Agency, 2013). (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (BLM, Bureau of Land Management; REA, Rapid Ecoregional Assessment)

buffalograss and blue grama are warm-season (C_4) species (Weaver and others, 1996), the photosynthetic pathway of which requires less water and nitrogen than that of C_3 species (cool-season grasses) to maintain a given rate of photosynthesis; therefore, C_4 plants can grow under conditions of high daytime temperatures, drought, and (or) limiting soil nitrogen, all of which characterize the shortgrass prairie. As a result, C_4 species comprise more than 80 percent of the shortgrass community at lower latitudes (30−42° N.), whereas north of 42° N. the percentage of C_3 species increases dramatically (Pieper, 2005). Some herbaceous species commonly associated with the codominant grasses include threadleaf sedge (*Carex filifolia*), prairie Junegrass (*Koeleria macrantha*), hairy grama (*B. hirsuta*), sand dropseed (*Sporobolus cryptandrus*), squirreltail (*Elymus elymoides*), curlycup gumweed (*Grindelia squarrosa*), and scarlet globemallow (*Sphaeralcea coccinea*) (Weaver and others, 1996). In some areas, woody species, sand sagebrush (*Artemisia filifolia*) and shinnery oak (*Quercus havardii*) in particular, form distinct communities that differ significantly from the surrounding grasslands.[3]

Landscape Structure and Dynamics

The shortgrass prairie is a patchwork of community types shaped primarily by soil type and climate variability, including microclimatic conditions associated with topographic heterogeneity (Knight, 1994; Weaver and others, 1996; Limb and others, 2009). Shortgrass prairie soils are generally perpetually dry because the scant precipitation usually infiltrates only a few centimeters of the upper soil horizon and the moisture is quickly lost to evapotranspiration (Sala and others, 2013). Infiltration of up to 135 cm can occur, but it is usually attributable to infrequent but major storm events (Sala and others, 2013). The rate and depth of infiltration, however, are strongly influenced by soil type. The “tighter” clayey loams that characterize much of the shortgrass prairie are relatively resistant to infiltration, whereas the “looser” sandy loams allow faster, deeper infiltration. In turn, these soil-moisture dynamics influence the distribution of vegetation types in the shortgrass prairie. For example, woody species such as sand sagebrush and shinnery oak[3] require more water than the shortgrasses, thus they occur on relatively sandy soils, whereas the shortgrasses have life-history characteristics, such as limited aboveground biomass, that allow them to tolerate water stress fairly well and become dominant on tight soils (Weaver and others, 1996). Rates of decomposition, both above and below ground, are also slow in the semiarid shortgrass prairie climate; thus, soils lack the high humus content and nutrient availability—nitrogen in particular—found in more mesic prairie soils (Lauenroth and others, 1978; Pieper, 2005).

The patchy structure is significantly enhanced by “pulses” of plant growth and seral setbacks caused by disturbances, including drought, herbivory, other animal activities, and fire. The pulses are generally responses to significant rainfall, sudden releases of nutrients through fire and animal droppings, or enhanced nutrient cycling through the activities of fossorial mammals and harvester ants. The setbacks are generally caused by deep and (or) prolonged drought, grazing and trampling by large herbivores, outbreaks of grasshoppers or other herbivorous invertebrates, burrowing by fossorial animals, and fire. The spatiotemporal scales of these dynamics vary widely, which also contributes to the patchy structure. For example, short-term drought typically happens on an annual basis and often has relatively short-term and sometimes localized effects. Longer term droughts, driven by shifts in sea-surface temperatures

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[3] See Chapter 4, “Mixed-Grass Prairie,” for a more in-depth discussion of these shrub communities.

and random fluctuations, generally occur at decadal time scales and typically occur over entire regions (Hoerling and others, 2014). Overall, the high levels of spatial variability promote species diversity and overall ecosystem resilience in the shortgrass prairie. For example, growth pulses of relatively succulent and nutritious forage are sought by wandering ungulates (Fuhlendorf and others, 2001).

Major herbivores believed to have had significant influence on the structure and dynamics of the shortgrass prairie are the American bison (*Bison bison*), black-tailed prairie dogs (*Cynomys ludovicianus*), short-horned grasshoppers (Caelifera), and belowground herbivores such as beetle grubs and nematodes (Rottman and Capinera, 1983; Martinsen and others, 1990; Milchunas and others, 1998; Minnick and others, 1999). Herbivory helps to recyle nutrients in a system where decomposition is otherwise relatively slow, thus contributing to pulses of vegetative growth and the development of patches in various seral stages. The juxtaposition of disturbed patches created by large mammal grazing and trampling with relatively undisturbed patches also may support grasshopper populations, which require bare soil in which to lay their eggs and nearby vegetated patches where grasshopper nymphs can feed (Knight, 1994). Drought also favors grasshoppers, possibly because drought suppresses some of the fungal and bacterial infections that can affect grasshoppers, and population outbreaks are known to coincide with drought (Weaver and others, 1996).

Reconstructing natural fire regimes of the shortgrass prairie has been difficult because burned shortgrass vegetation provides little fire-scar data; however, fire is also believed to have been an important process in shaping the structure and dynamics of the shortgrass prairie (Ford and McPherson, 1997; Brockway and others, 2002). Fire provided a mechanism for cycling nutrients through the system and kept woody vegetation from expanding and becoming dominant. Historically, the dry climate and slow rates of decomposition led to buildups of fine fuels that could sustain large prairie fires until cold or wet weather, or a natural break such as a large river, extinguished or stopped the fire (Knight, 1994). Mean fire-return intervals in the North American prairie system is estimated to have been 2–30 years, with longer intervals in more broken terrain, where bunchgrasses were dominant (the more continuous cover of sod-forming grasses promotes fire more readily than the discontinuous cover of bunchgrasses), and in the more arid regions where fuels built up more slowly. With lightning being the primary source of ignition, most fires occurred during July and August, when conditions and fuels were dry and thunderstorms were frequent (Knight, 1994). Native Americans also frequently used fire to drive game and alter vegetation. Probably due to evolving with frequent fire, warm-season grasses are generally more fire tolerant than cool-season grasses, which in part explains their dominance in in the shortgrass prairie (Knight, 1994).

Whether or not these and other disturbances lead to changes in community composition is a function of complex interactions between climate variability and disturbance regime. For example, significant rainfall in late summer or fall will have different effects on community composition than significant rainfall in spring and early summer. The same principle applies to the seasonality, intensity, and between-disturbance intervals of herbivory (Fuhlendorf and others, 2001). Pyric-herbivory dynamics (fire followed by herbivory) may be especially important in promoting a shortgrass mosaic that supports wildlife diversity (Fuhlendorf and others, 2010).

Associated Species of Management Concern

The shortgrass prairie supports nearly every terrestrial species of wildlife treated as a CE in this REA, including ferruginous hawk (*Buteo regalis*) (Chapter 13), mountain plover

(*Charadrius montanus*) (Chapter 17), long-billed curlew (*Numenius americanus*) (Chapter 16), burrowing owl (*Athene cunicularia*) (Chapter 12), black-tailed prairie dog (Chapter 20), and swift fox (*Vulpes velox*) (Chapter 22). The lesser prairie-chicken (*Tympanuchus pallidicinctus*) (Chapter 15) also uses shortgrass prairie habitats, particularly the transition zones where there is a juxtaposition of shortgrass types that provide brood-rearing habitat and mixed-grass shrub-steppe (shinnery oak, sand sagebrush, and mixed grasses) types that provide the taller cover needed for nesting and escape. Lesser prairie-chickens also establish their leks (male display grounds where mating takes place) in open, often disturbed sites in shortgrass types (Hagan and Giesen, 2005). In addition to the species listed above, the Great Plains Landscape Conservation Cooperative lists several priority grassland species whose ranges overlap the shortgrass prairie: prairie falcon (*Falco mexicanus*), Cassin's sparrow (*Peucaea cassinii*), lark bunting (*Calamospiza melanocorys*), grasshopper sparrow (*Ammodramus savannarum*), Harris' sparrow (*Zonotrichia querula*), black-footed ferret (*Mustela nigripes*), and American bison. All of these species are shortgrass obligates and (or) use grass-shrub-steppe types.

Change Agents

Anthropogenic activities since European settlement have contributed to significant changes in the shortgrass prairie (Hart, 2008). Among the threats considered most serious are habitat loss and fragmentation resulting from agricultural cultivation, fencing, and development, as well as changes in community composition arising from fire suppression, invasive species, and climate change (Weaver and others, 1996; Hart, 2001; Brockway and others, 2002; Pieper, 2005; Heisler-White and others, 2008).

Development

Energy and Infrastructure

Most of the shortgrass prairie is ranked as having fair to superb wind energy potential; accordingly, development of wind turbine farms has been accelerating across much of the region. The greatest wind farm density may be found in the northern Texas panhandle and western Oklahoma (Open Energy Information, 2013). The recent advent of directional drilling coupled with hydraulic fracturing also has led to booms in oil and gas development in some areas of the shortgrass prairie, particularly in northeastern Colorado and central Oklahoma (U.S. Energy Information Administration, 2011). Energy-development infrastructure, including well pads, wind turbines, roads, pipelines, and transmission lines, not only results in direct loss of shortgrass prairie—it also causes significant ecosystem fragmentation. Most research, however, has focused on the effects of energy development on wildlife. Little research has been devoted to the effects of energy development on vegetation communities in the shortgrass prairie. It is clear from work conducted in other regions, however, that corridors and patchworks of surface disturbance often create conditions conducive to agents of change in vegetation communities, such as range expansions of nonnative and (or) invasive plant and animal species (Bergquist and others, 2007). This is particularly true of ruderals, which are native or nonnative species that quickly colonize recently disturbed sites and remain dominant for several years; where disturbance is severe and (or) chronic, ruderals may become permanently established. Accidental or intentional discharges of fluids associated with oil and gas production, such as the "produced waters" that arise from drilled formations, drilling "muds," or hydraulic fracturing fluids, are also potential sources of disturbance. These fluids often contain high levels of salinity, radioactive components, hydrocarbons, and other chemicals capable of creating perturbations to vegetation, soil communities, and nutrient cycling processes (Irwin and others, 1996; Kharaka and Otton, 2003; Fisher and Sublette, 2005; U.S. Environmental Protection Agency, 2012).

Agricultural Activities

Agriculture has resulted in significant habitat loss and degradation in the shortgrass prairie, with 50–75 percent of the shortgrass prairie having been cultivated at some point. The majority of studies evaluating effects of cultivation have involved birds, and it is well established that cultivation is the single greatest cause of decline among grassland bird species. Cultivation not only alters the vegetative community structure and composition, it also entails frequent applications of pesticides, herbicides, and mechanical disturbances (such as disking) to remove weeds, all of which can result in diminished nesting success. Furthermore, cultivation has altered the soil structure and soil communities to the extent that return to its original condition in the near future is unlikely. Winter wheat is a common crop in the shortgrass ecoregion because it does not require irrigation, and with water becoming increasingly expensive to pump as the main aquifer under the SGP (Ogallala) is mined, the emphasis on dryland farming will increase as more drought- and herbicide-tolerant varieties of corn, soybeans, and other crops are developed.

Prairie soils were lost at a rapid rate once cultivation exposed them to significant erosion from water and wind, furthered by slow recovery in these droughty lands. During the 20th century, the Conservation Reserve Program (CRP) was introduced in the United States as part of the Farm Bill to help curtail the loss of highly erodible soils. Farmers enrolled in the CRP were paid to rest their lands and plant them with perennial grasses, initially exotics, but more recently native grass-forb mixes have been encouraged. Although the CRP has greatly diminished the rates of decline among some grassland birds (Wiens and McIntyre, 2008), appropriations for the CRP have declined, and economics often drive farmers out of the program. For example, the emphasis on manufacturing biofuels, as well as the development of more drought-resistant crop varieties, has reduced the incentive to reenroll in the CRP and created an incentive to expand sod busting in native prairie in more arid regions.

Most authorities believe that because shortgrass systems evolved with heavy grazing, plants of the shortgrass prairie are well adapted to grazing effects, and that livestock grazing serves as a surrogate for the intensive grazing of native large mammals, such as bison, that once dominated the plains (Knight, 1994; Weaver and others, 1996). Moderate to heavy grazing is probably not unlike that with which the shortgrass system evolved (Hart, 2001). Indeed, plant diversity in exclosures was lower than it was in lightly to moderately grazed plots, likely because grazing removed competitive species that readily became dominant. The heavily grazed plots, on the other hand, were dominated by the two main shortgrass species: blue grama and buffalograss (Hart, 2001). Despite considerable research on the topic, there is still debate as to whether livestock grazing is beneficial or detrimental to plants, likely due to a suite of factors considered (for example, species- to community-level responses, patch size, aboveground/ belowground biomass). It is clear that plant responses to grazing are complex interplays of grazing intensity, seasonality, and duration. Under natural conditions of perturbation, patchworks of forage types led large mammalian herbivores to wander widely in search of preferred forage types (Knight, 1994); however, fencing, season and duration of grazing, and number of animal units grazing a given unit of area can have widely different effects on plant communities. Clearly, too many animals foraging in a small area, particularly during severe drought, will lead to reduced plant vigor and increased mortality, and it also can lead to invasions of ruderals (Knight, 1994). Grazing also may interact with fire suppression to alter fire regimes of the shortgrass prairie by reducing the rates of fuel buildups (Knight, 1994; Weaver and others, 1996; Brockway and others, 2002).

Fencing is another, albeit indirect, effect of grazing. Although fencing influences grazing patterns, it also fragments the shortgrass prairie ecosystem. Furthermore, fencing provides a perch for predators, which may alter predation patterns in systems where vertical structures are rare, and many bird species collide with fences, including lesser prairie-chickens (Wolfe and others, 2007).

Altered Fire Regime

The spatiotemporal properties of a fire regime are complex and governed by drought, fuel biomass and condition, and topographic factors (McKenzie and others, 2011), which make it difficult to generalize about altered fire regimes over large areas. Furthermore, the effects of fire in the shortgrass steppe have received little attention compared to more productive grassland ecosystems (Stapp and others, 2008). Overall, the passive change in fire regime that resulted when Native Americans were displaced by European settlers and the active change that resulted from modern fire suppression have resulted in greatly diminished fire frequency and size across the Great Plains (Knight, 1994), but this effect is believed to be greater in mixed-grass and tallgrass systems than in shortgrass systems. The consequences of fire suppression are complex and not fully understood, although shifts in community composition are a known consequence (Brockway and others, 2002). The expansion of woody vegetation, junipers (*Juniperus* spp.) and mesquite (*Prosopis* spp.) in particular, is of concern where fire suppression has taken place in shortgrass systems. Fire suppression also can result in greater buildups of fuels, which in turn can promote hotter fires and greater plant mortality (Knight, 1994), although moderate to heavy grazing in shortgrass systems may counteract any effect of fire suppression.

Invasive Species

In addition to woody plant expansion resulting from fire suppression (Weaver and others, 1996), several exotic grass and forb species are invading (or may invade) the shortgrass prairie where surface disturbance associated with development has occurred and along travel corridors. These species are altering or have the potential to alter shortgrass prairie communities (Weaver and others, 1996; Hart, 2001). Leafy spurge (*Euphorbia esula*) is an aggressive, nonnative species now distributed throughout western reaches of the shortgrass prairie ecoregion, including much of eastern Colorado, a few counties in Kansas, and parts of northeastern New Mexico (Natural Resources Conservation Service, 2014). It is unpalatable to most North American herbivores and livestock, thus it can grow and reproduce relatively unchecked, and where moisture is not limiting, it can form monocultures. Exotic bromes (*Bromus arvensis*, *B. japonicas*), which have invaded much of the shortgrass prairie, often compete with native grasses. Cheatgrass (*Bromus tectorum*), which has invaded and is altering much of the sagebrush steppe, is well-distributed throughout the SGP, but its potential for altering shortgrass prairie systems is not yet clear. Also capable of altering shortgrass communities are invasions of crested wheatgrass (*Agropyron cristatum*), knapweeds (*Centaurea* spp.), kochia (*Kochia scoparia*), and yellow sweetclover (*Melilotus indicus*) (Weaver and others, 1996). Of increasing concern are the Old World bluestems (*Bothriochloa* spp.), which are relatively tall, sod-forming, very drought-and grazing-tolerant, warm-season perennial grasses frequently planted as forage or in restoration/CRP sites; although they have escaped cultivated sites and are invasive in local areas (Ruffner, 2012), their potential for altering the shortgrass prairie or how they may respond in a warming climate is not clear.

Insects and Disease

No exotic insects or diseases are known to have significant effects on the shortgrass prairie. There are native species of invertebrates, however, that can have significant effects on plant mortality. For example, white grubs (*Phyllophaga fimbripes*) consume belowground parts of buffalograss, and large population increases of this species, possibly driven by certain environmental factors, can cause widespread mortality of buffalograss (Rottman and Capinera, 1983). Short-horned grasshoppers and nematodes also can attain high population numbers under various conditions, which can lead to episodic changes in overall aboveground and (or) belowground plant biomass (Knight, 1994; Weaver and others, 1996; Blossey and Hunt-Joshi, 2003).

Climate Change

Although average annual precipitation over most of the SGP increased from 1958 to 2008 (Karl and others, 2009), projections for the remainder of the 21st century indicate that annual average precipitation may decrease in the SGP, especially in the southwestern portion of the region. The greatest decreases are projected to occur in spring and summer (Karl and others, 2009); thus, the historical timing of annual rains may shift, potentially decoupling rainfall-plant phenlogy cycles. Very heavy rainfall events are also projected to increase, which could help to offset overall annual decreases in precipitation and increased rates of evapotranspiration resulting from increasing average temperatures. Even if heavy rainfall events offset these other effects, overall patterns in soil moisture could be altered, which in turn could alter species distributions.

A modeling effort designed to better understand the effects of temperature and soil water on germination and establishment patterns was conducted for two shortgrass species: (1) blue grama, which is distributed throughout much of the SGP, and (2) black grama (*B. eriopoda*), which occurs south and west of the SGP. The model indicated that the number of years in which temperature and soil-water conditions were suitable for germination and establishment decreases along a north-to-south gradient for blue grama and increases for black grama, which matched observed dominance patterns (Minnick and others, 1999). In the ecotone where the two species are codominant, the number of years in which the conditions were favorable for germination and establishment were roughly equal for the two species. Further modeling efforts (using the climate data scaled for use in three general circulation models) indicated that climate change may result in a northward shift of the ecotone and a northward expansion of black grama (Minnick and others, 1999). The extent to which this phenomenon may hold true for other important SGP species is not well understood, but similar scenarios for other species are likely. Increasing levels of atmospheric CO_2 also have been shown to enhance seedling recruitment of at least one low-digestibility grass—needle-and-thread (*Hesperostipa comata*)—and there is evidence that increasing CO_2 levels may drive a shift towards C_3 species dominance in the shortgrass prairie (Morgan and others, 2004).

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting shortgrass prairie is illustrated in figure 10. Ecological attributes and CAs identified in the Rapid Ecoregional Assessment are enumerated in tables 9 and 10.

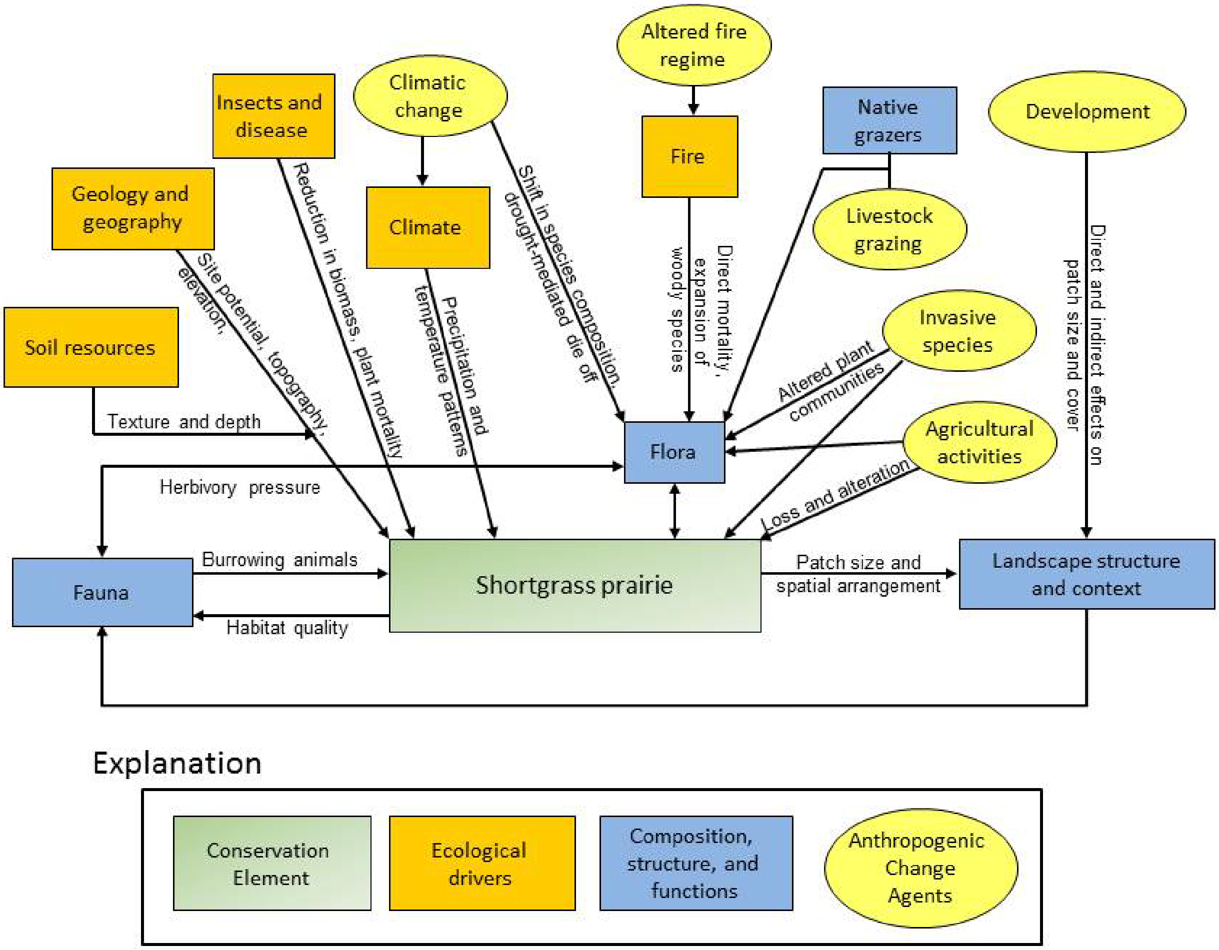

Figure 10. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for shortgrass prairie in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of shortgrass prairie are shown in orange rectangles (see also table 9); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 10).

Table 9. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for shortgrass prairie.

Attribute	Variables
Amount and distribution	Distribution, vegetation condition of shortgrass prairie.
Landscape structure	Patch size and spatial distribution (area, connectivity).
Landscape dynamics	Fire regime (frequency and severity), grazing.
Associated species management of concern	Ferruginous hawk, mountain plover, long-billed curlew, burrowing owl, black-tailed prairie dog, swift fox, and lesser prairie-chicken.

Table 10. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for shortgrass prairie.

Attribute	Variables
Development (energy and infrastructure)	Alteration (direct loss of shortgrass community), fragmentation (reduction in connectivity), discharge of produced waters.
Development (agricultural activities)	Alteration (direct loss of shortgrass community, changes to vegetation communities, changes to soil structure and communities, reduction in seed production, removal of surface fuel), fragmentation (reduction in connectivity), herbicide and pesticide application.
Climate change	Alteration (changes to vegetation communities, latitudinal shifts in plant populations), drought effects.
Invasive species	Alteration (changes to vegetation communities).
Altered fire regime	Alteration (changes to vegetation communities, decrease in shrubland, increase in exotic species, such as cheatgrass), suppression (reduction in fire size and frequency).
Insects and disease	Presence and effects on population of animal species (for example, chronic wasting disease and sylvatic plague), reduction of biomass or mortality of plant species (for example, pathogens, fungi, beetle grubs, short-horned grasshopper, and nematodes).

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Chapter 6. Riparian Areas and Wetlands

By Lucy E. Burris

Key Ecological Attributes

Distribution and Ecology

Riparian areas and wetlands are important transition zones between aquatic environments and upland, terrestrial ecosystems. Both riparian areas and wetlands are considered "emergent" systems in that they exhibit properties not found in the adjacent deepwater or dry terrestrial systems. Wetlands are permanently or intermittently flooded and occur where the water table is at or near the land surface or where land is covered by shallow water creating saturated or periodically anaerobic or hydric soils (Mitsch and Gosselink, 2000). Wetland habitats include freshwater marshes, swamps, bogs, seeps, wet meadows, and shallow ponds. Wetlands can occur naturally or they can be created by humans, such those as created by sustained irrigation, canals, or dredging. Wetland vegetation includes wetland obligates (or hydrophytes, those plants requiring saturated soils and able to tolerate anaerobic conditions) and wetland facultatives (those plants which can exist in but do not require saturated soils).

In the SGP ecoregion, most wetlands are playa wetlands—shallow, precipitation-filled, clay-lined, marsh-like ponds (Mitsch and Gosselink, 2000; Haukos and Smith, 2003). Playas are covered in detail in Chapter 7, "Playas and Saline Lakes," therefore only general information on wetlands is provided here. Riparian areas are heterogeneous bank zones along rivers and streams that are periodically inundated by flood events (Gregory and others, 1991; Naiman and Decamps, 1997). While wetland soils are saturated for long periods, riparian soils are saturated only during the flooding period and will dry after flooding subsides; the typical water table may be several meters deep in riparian zones. Depending on the duration of flooding, riparian vegetation can range from obligate upland plants (when flooding is extremely limited) to facultative wetland plants (when flooding is of long duration; Mitsch and Gosselink, 2000).

Many common riparian plants such as cottonwood (*Populus* spp.) and willow (*Salix* spp.) rely on flooding events for seed dispersal, streambank scouring to create exposed moist substrates, and water table maintenance (Scott and others, 1996; Amlin and Rood, 2002). In the SGP, riparian vegetation types can range from grasses near small and ephemeral streams, to shrubs as stream size and water availability increases, to patchy linear forests along major drainages like the South Platte, the Arkansas, the Canadian, and the Red Rivers (Dodds and others, 2004). Forests can be cottonwood and willow, as mentioned above, eastern redcedar (*Juniperus virginiana*), or a mixture of native and introduced species (Nagler and others, 2011; Wine and Zou, 2012). Vegetation is highly variable and controlled by physical factors such as stream gradient, sinuosity, channel width-to-depth ratios, topography, and soil type (Knight, 1994; Scott and others, 1996; Naiman and Decamps, 1997). Riparian plant communities reflect histories of both fluvial disturbance from floods and the nonfluvial disturbance regimes of adjacent upland areas, such as fire, wind, plant disease, insect outbreaks, and native and nonnative grazing (Gregory and others, 1991; Scott and others, 2003; Glenn and Nagler, 2005; Skagen and others, 2005). Both wetland and riparian habitats are characterized by high diversity, density, and productivity of both plant and animal species, particularly when compared to surrounding drier uplands (see for example Smith, 2003).

Both terrestrial uplands and water systems contribute to riparian and wetland function: uplands through organic matter inputs and water systems through disturbance (primarily flooding but also sediment deposition) and moist soil maintenance. Since riparian and wetland areas are transitional between water ways and drier uplands, demarcating riparian/wetland and not riparian/not wetland areas can be difficult. Where possible, the presence or absence of hydric or flood-adapted and flood-tolerant plant species, as well as topography (the presence of incised channels, flood plain morphology, or surface depressions) and (or) the presence of hydric soils, are used to delineate riparian and wetland areas.

Landscape Structure and Dynamics

The spatial distribution of riparian and wetland areas is influenced by precipitation, soils, topography, proximity to streams and rivers, and the dynamics of natural and anthropogenic disturbances (Gregory and others, 1991). Riparian hydrologic regimes are characteristically dynamic, and the amount, timing, and temporal variability of groundwater and surface water inputs affect riparian structure and function. Both seasonal and interannual variability in water flow affect the native plant and animal communities (Baron and others, 2002). Whereas cycles of inundation and drought create significant disturbance in wetland and riparian systems, these cycles are critical to ecosystem health (see for example Euliss and others, 2004).

A number of key drivers regulate the structure and function of riparian and wetland areas (Baron and others, 2002). In particular, flow regime defines the rates and pathways by which precipitation enters, circulates, and exits these systems. In areas like the SGP with warm-season grasses, intermittent streams dominate over perennial streams (Dodds and others, 2004). Flow in ephemeral and intermittent streams is driven by short-lived but intense convective storms (thunderstorms) and, in some locations, by rapid melting of heavy spring snows with resulting flooding and subsequent drying until the next rain event. Ephemeral and intermittent streams are both dry during some portion of the year, but intermittent streams have a longer wet period (generally the duration of the wet season). At the extreme, severe flooding can remove streamside vegetation and reposition the stream channel, opening up new areas for colonization.

During storm events, upland runoff can accumulate sediments, nutrients, and toxins from upland areas and deposit them in riparian and wetland areas (Skagen and others, 2008). While nutrients and sediments can be beneficial, in excess and in combination with toxins such as pesticides, fertilizers, and herbicides, they can be detrimental to riparian and wetland ecosystem health. In the absence of periodic floods or during drought conditions, water tables drop, riparian and wetland areas become drier, and dry fuels accumulate, resulting in increased fire frequency and intensity; thus, riparian corridors may provide pathways for fire to spread (Petit and Naiman, 2007). Many plants such as cottonwood and willow can resprout after fire, but susceptible species may be lost. Sedimentation affects physical structure, nutrient levels, and chemical characteristics, which in turn regulate pH, productivity, evapotranspiration, and water quality (Baron and others, 2002). Excess sedimentation, for example, can suppress the emergence of wetland vegetation and invertebrates (Jurik and others, 1994; Gleason and others, 2003).

Ecosystem process rates and community structure are governed by the biotic assemblage. In grassland settings for example, where intermittent streams dominate and riparian trees are absent, streams have lower leaf litter input and grazer invertebrates dominate (Dodds and others, 2004). Aboveground structures of riparian areas and wetlands include precipitation and sunlight intercepting surface area, surface roughness, water storage capacity, and litter fall. Belowground structures include roots providing soil stabilization, nutrient exchange, and filtration. Both above- and belowground structures are essential to the maintenance of stream health through velocity reduction of eroding overbank flows, nutrient exchange, and water storage (Tabacchi and others, 2000; Richardson and others, 2007). Since riparian areas have historically been low-stress, moderate disturbance environments, plants are adapted to variable water levels, high nutrients, and disturbance (Glenn and Nagler, 2005). Adaptations include pioneer-species traits such as rapid growth rates, colonization from roots and seeds, pulsed seed dispersal via wind and water, moisture-driven germination, and seed establishment on bare substrates (Glenn and Nagler, 2005; Richardson and others, 2007).

Associated Species of Management Concern

Snowy plovers (*Charadrius nivosus*) (Chapter 18) nest in riparian areas, although at much lower frequencies than at saline lakes, but their nest success is similar in riparian and saline lake habitats (Conway and others, 2005a). For nesting, snowy plovers select dry ground at riparian areas where dry mud, mud, and water occur rather than locations with only dry ground (Grover and Knopf, 1982; Conway and others, 2005b). Interior least tern (*Sterna antillarum athalassos*) (Chapter 14) nesting in nonforested riparian areas along the Platte, Arkansas, Canadian, and Red Rivers has been reported (Conway and others, 2003). Nest sites along the Red River in Texas are primarily on sand or gravel bars roughly 200 m from water, at least 10 m from vegetation, and within 15 cm of debris (driftwood, rocks) but do not differ in general characteristics from the surrounding landscape. Nest site fidelity is low, and nest initiation can be delayed due to annual variations in flooding and nesting season water levels. Long-billed curlews (*Numenius americanus*) (Chapter 16) breed primarily in open grasslands in the SGP, and their need for nearby water sources is poorly understood (Fellows and Jones, 2009). Nonbreeding migrants have been observed at wetland areas like Cheyenne Bottoms in Kansas and reservoirs and rivers in Colorado, and overwintering birds will use playas in Texas. Loss of grassland habitat has led to a 30 percent reduction in historical range, primarily from the eastern region. Pallid bats (*Antrozous pallidus*) (Chapter 19) in the Red Hills of Kansas (at the eastern edge of their range) roost within 50 m of ephemeral water, which may be due to insect activity and hydration maintenance (Miller and Jenson, 2013). In the panhandle of Oklahoma, red-tailed hawks (*Buteo jamaicensis*) primarily nest in riparian cottonwood trees within grassland settings (McConnell and others, 2008). In the Texas panhandle, female Rio Grande wild turkeys (*Melegris gallopavo intermedia*) prefer riparian habitat for nesting; both males and females prefer riparian habitats over urban or Conservation Reserve Program lands (Hall and others, 2007). Freshwater mussels (Chapter 10) generally require moving freshwater, so their occurrence in wetlands is limited (Angelo and others, 2009). An extensive study in Kansas found very limited occurrences of freshwater mussels along western river drainages.

In general, little land area in the SGP has been given protection status to maintain biodiversity or for multiple uses (Aycrigg and others, 2013). Across the United States, only about 5 percent of floodplain and riparian areas have been set aside for diversity management, less than the 17 percent suggested by the Aichi Biodiversity Target of the Convention on Biological Diversity. If this same fraction holds for riparian areas in the SGP, only limited areas will continue to be available for species of concern.

Change Agents

Wetlands across the United States have been heavily degraded in the past, particularly during the 1950s through the 1970s (Dahl, 2011). Recent legislation and Federal programs, however, have resulted in a reduction in wetland acreage loss and even some recovery, particularly in freshwater ponds. Wetlands continue to be vulnerable to development in the form of energy infrastructure, agricultural activities, altered fire regime, and climate change.

Development

Development has frequently included dams, ditches, groundwater pumping, and other anthropogenic alteration of hydrology affecting the flow regime, reducing water tables, increasing sedimentation, and altering riparian biota (Copeland and others, 2010). Of particular importance, flow regulation driven by development (whether for water control or irrigation) is a key driver in the presence of invasive riparian species and a decrease in species richness (Uowolo and others, 2005; Copeland and others, 2010). Generally, increases in flow alteration and decreases in water table are accompanied by a decrease in cottonwood dominance (Merritt and Poff, 2010). Changes in timing of peak flows can be detrimental to native species and facilitate the dominance of introduced species such as tamarisk (*Tamarix* spp.). A secondary effect of development is the redistribution of water on the landscape: creating or expanding wetlands where none existed previously (Crifasi, 2005). For example, in eastern Colorado, moist areas along canals and return flows from irrigation have created extensive wetland habitats which are now at risk for dewatering as water rights are sold and water is redirected to Front Range municipalities (Wiener and others, 2008).

Energy and Infrastructure

Energy development (wind generation, gas and oil drilling, and biofuels agriculture) does not pose particular unique risks for riparian areas and wetlands. Key areas of concern are alteration of surface flow timing, volume and loss of periodic flooding through impoundment or channelization, groundwater depletion through pumping, and contamination from upland disturbance via runoff and sediment inputs (Smith and others, 2008; Brinson and Eckles, 2011).

Agricultural Activities

Loss of playas due to sediment infilling from surrounding agriculture is an area of major concern (Brinson and Eckles, 2011). Although riparian and wetland areas are less vulnerable to the conversion to cropping because of their generally wetter conditions, when near agriculture they are also at risk from sedimentation and contamination. Another area of agricultural threat is livestock grazing. While interest in the effects of general grazing as a research topic peaked in the 1980s, it is still one of the most important areas of concern in the SGP (Poff and others, 2011, 2012). Heavy grazing, either by duration or intensity, and poorly timed grazing by introduced and native herbivores can reduce or eliminate riparian vegetation, allow bank downcutting, increase sediment runoff, increase nutrient load, and lower water tables (Chaney and others, 1990). Riparian vegetation is likely to recover when grazing pressure is reduced as long as flow modifications have been minimal (Chaney and others, 1990; Skagen and others, 2005). In

Oklahoma, for example, degraded grasslands are now experiencing expansion of eastern redcedar and riparian forests (Wine and Zou, 2012).

Altered Fire Regimes

The effects of fire on streams and their associated riparian and wetland areas are poorly understood (Bisson and others, 2003; Dwire and Kauffman, 2003). Although fires are not uncommon in the tallgrass prairie, on Konza Prairie (near Manhattan, Kans.) riparian fires are infrequent and of low intensity (Briggs and others, 2002). Grazing reduces fire severity and allows an increase of woody plants into grasslands. As mentioned above, flooding is an important riparian disturbance; its effects can be compounded when occurring with fire. When fire precedes flooding, erosion may be severe; when fire follows flooding, regenerating vegetation may be destroyed (Petit and Naiman, 2007). Water control and fire suppression since European settlement may be increasing riparian fire potential through buildup of fuels while increased human use of riparian areas is increasing ignition sources (Dwire and Kauffman, 2003; Wine and Zou, 2012).

Invasive Species

Much has been published about Russian olive (*Elaeagnus angustifolia*) and tamarisk (saltcedar) invasions of western riparian areas. This material is only briefly summarized here; refer to the cited references for additional information. Introduced species, particularly Russian olive and tamarisk, have altered riparian ecosystems and the species that rely on them. Across the western United States (including the SGP), tamarisk and Russian olive are the third and fourth most frequently found tree species, and tamarisk provides the second densest canopy cover after cottonwood (Friedman and others, 2005). Both tamarisk and Russian olive are found throughout the SGP, although Russian olive is not found south of the Texas panhandle, likely because of temperature constraints.

Russian olive is a naturalized shrub or small tree from Europe planted in the western United States since about 1900 for windbreaks, particularly along riparian areas (Lesica and Miles, 2001a; Katz and Shafroth, 2003). Although it is a pioneer species like cottonwood and willow, Russian olive is able to establish in the shade beneath established cottonwoods, which are shade intolerant and establish on unvegetated, moist substrates (Katz and Shafroth, 2003). After fire, Russian olive can resprout from root crowns.

Also a pioneer species, tamarisk is a drought- and salt-tolerant European shrub/tree originally imported for horticultural use (Lehnhoff and others, 2011). The extensive irrigation ditch systems of the southern Great Plains may be promoting the spread of tamarisk (Stohlgren and others, 1998; Hart, 2002). During the late 1800s, the plant became naturalized and spread to most of the watercourses in the southwestern United States by the early 1900s (Katz and Shafroth, 2003; Merritt and Poff, 2010). Originally, spread was thought to be limited to the southwest because of cold intolerance, but since the 1950s it has been found in the northern Great Plains where plants are successful though less productive (Lesica and Miles, 2001b).

Russian olive and tamarisk produce easily dispersed seeds with longer viability than native cottonwood or willow (Jarnevich and Reynolds, 2011; Lehnhoff and others, 2011). Both Russian olive and tamarisk are able to access groundwater at depths of 3 m, deeper than cottonwood or willow (Katz and Shafroth, 2003; Nagler and others, 2011). Like Russian olive, tamarisk will resprout from root crowns after fire, although less vigorously than cottonwood or willow (Stromberg and Rychener, 2010). Although tamarisk provides a good fuel source, increased fire frequency in riparian zones may be due to the absence of scouring floods and resulting fuel accumulation on regulated rivers (Stromberg and Chew, 2002).

Research conclusions are mixed about habitat interchangeability of willow and cottonwood stands and (or) Russian olive and tamarisk stands. It is not uncommon to find cottonwood stands with a Russian olive or tamarisk understory or streambanks with cottonwood trees younger than co-occurring Russian olive or tamarisk (Lesica and Miles, 2001 a, b). Beaver have been found to browse more on native cottonwood than on Russian olive, resulting in more suppression of cottonwood close to streams than of Russian olive. Migrating birds that use riparian corridors prefer cottonwood and willow patches with structural diversity (Pocewicz and others, 2013); Russian olive and tamarisk within cottonwood stands may provide an important midcanopy habitat for birds although they offer limited habitat for cavity nesters (Katz and Shafroth, 2003; van Riper and others, 2008; Fischer and others, 2012). In the Southwest, soil salinity in tamarisk stands has been found to be higher than tolerable for cottonwood and willow; however, increased soil salinity in tamarisk stands may be due to the lack of soil-cleansing floods rather than tamarisk presence (Bagstad and others, 2006; Ladenburger and others, 2006).

Climate Change

Riparian areas provide an extensive array of ecosystem services ranging from regulating functions, such as disturbance prevention, to habitat functions like corridors, to production functions like raw materials (Capon and others, 2013). Climate change may affect any of these, through, for example, altered vegetation communities. Across the west, changes in streamflow, temperature, and snow pack since 1950 can be attributed to climate change (Barnett and others, 2008). While direct effects of climate change (warming temperatures and reduced precipitation) on southwestern United States grassland systems are becoming better understood, the direct effect to riparian areas is a current knowledge gap (Polley and others, 2013). Grazers may concentrate in riparian areas, increasing stress on vegetation and increasing streamside erosion. Drier conditions will reduce moisture (precipitation and groundwater) available to riparian systems, so drought-tolerant plant species such as tamarisk and eastern redcedar and fire frequency may increase (Bisson and others, 2003; Polley and others, 2013). Generally, smaller and less permanent wetland and riparian habitats will be most at risk, particularly from increases in evapotranspiration or decreases in precipitation (Matthews, 2008). Riparian habitats (including ravine areas) will be at increased risk for severe flooding and erosion because of increased storm intensity. Although total annual precipitation is anticipated to decrease slightly in the future, a recent examination of rainfall in the south Central Plains showed an increase in rainfall from 1920 to 2000 (Garbrecht and others, 2004; Burris and Skagen, 2013). Importantly, most of the increase occurred during spring, contributing to a short-term streamflow increase rather than increasing moisture during the drier summer and fall months. This seasonality shift in precipitation, coupled with increased summer temperatures, suggests that wetland and riparian areas may become increasingly drought stressed (Stocker and others, 2013). As previously mentioned, reduced flows and droughts contribute to increased fire frequency and increased dominance of introduced species as well as reduction in ecosystem services such as water storage, nutrient cycling, and habitat maintenance.

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting wetlands and riparian areas is illustrated in figure 11. Ecological attributes and CAs identified in the Rapid Ecoregional Assessment are enumerated in tables 11 and 12.

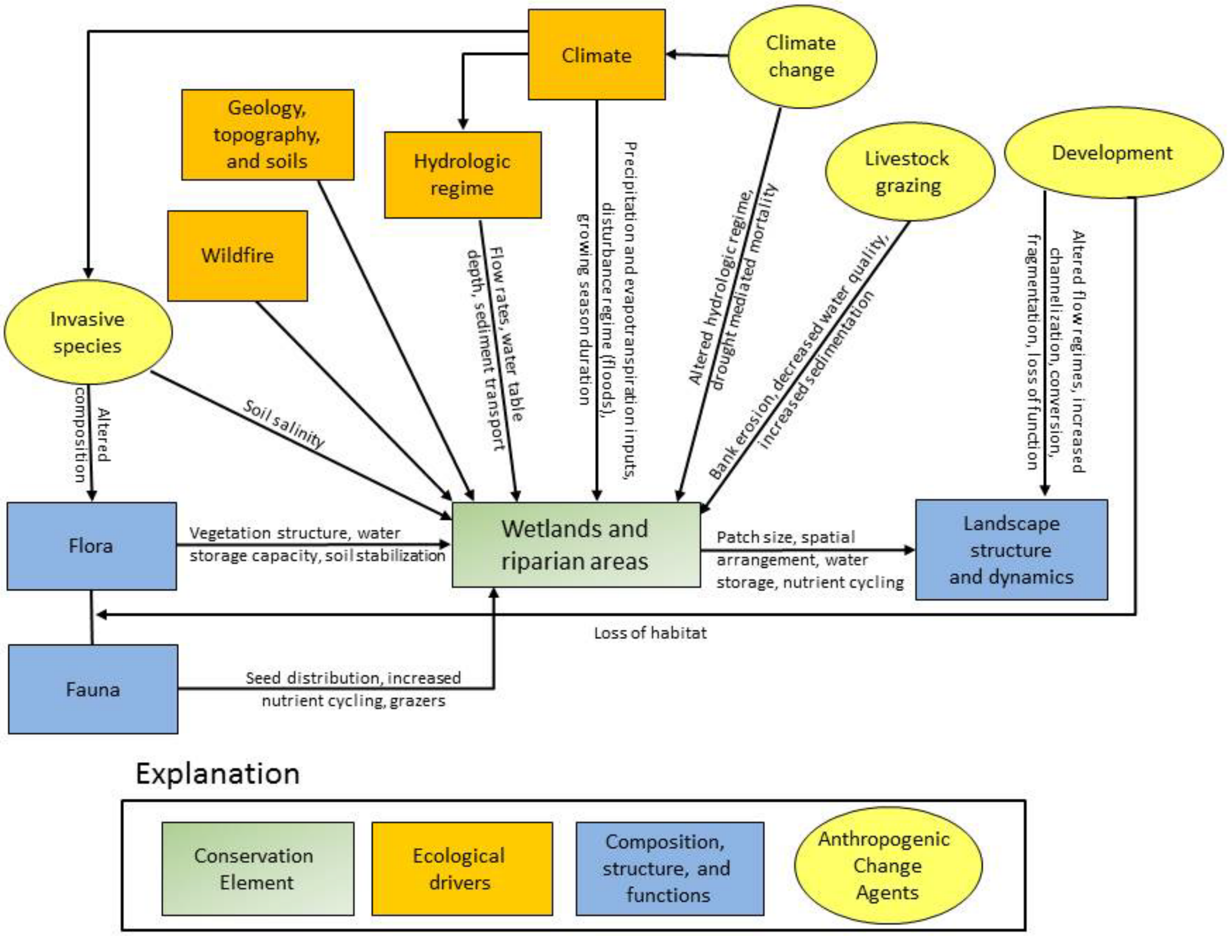

Figure 11. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for wetlands and riparian areas in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of wetlands and riparian areas are shown in orange rectangles (see also table 11); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 12).

Table 11. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for wetlands and riparian areas.

Attribute	Variables
Amount and distribution	Mapped distribution of wetland and riparian areas.
Landscape structure	Size and spatial distribution (area, density, connectivity), biotic assemblage.
Landscape dynamics	Hydrologic regime (amount, timing, temporal variability of groundwater and surface water inputs), cycles of inundation and drought, fire regime (frequency and severity).
Associated species management of concern	Snowy plover, long-billed curlew, interior least tern, pallid bat, red-tailed hawk, wild turkey, and freshwater mussel species.

Table 12. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for wetlands and riparian areas.

Attribute	Variables
Development (energy and infrastructure)	Alteration (changes to surface flow, loss of periodic flooding due to impoundment or channelization, groundwater depletion, contamination such as sediments and toxins from upland disturbance), fragmentation (reduction in connectivity), discharge of produced waters.
Development (agricultural activities)	Loss (infill/conversion into agricultural areas), alteration (sediment inputs from cropland, herbicide application), fragmentation (reduction in connectivity), livestock grazing (loss of riparian vegetation, bank erosion, lowering of water table).
Altered fire regime	Alteration (changes to vegetation communities), suppression (expansion of woody vegetation, buildup of fuels).
Invasive species	Alteration (changes to vegetation communities, such as effects of tamarisk and Russian olive on establishment of native species; soil salinity).
Climate change	Alteration (changes to species composition), drought effects (increase in evapotranspiration), seasonality shift in precipitation (effects on hydrologic regime).

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Chapter 7. Playas and Saline Lakes

By Cynthia P. Melcher

Key Ecological Attributes

Distribution and Ecology

In the SGP ecoregion, playas and saline lakes range from temporary to semipermanent wetlands of arid and semiarid climates. Based on Tiner (2003), both playas and saline lakes are "geographically isolated wetlands" (hereafter, isolated wetlands); that is, they are depressional wetlands completely surrounded by uplands and generally lack direct connection to other surface waters except when flooding may cause spilling over between isolated wetlands. In the SGP, playas are more specifically defined as freshwater wetlands that occur at the lowest points of small watersheds in relatively level terrain (Smith, 2003). They are perched above the water table, thus they function as recharge wetlands to the underlying aquifer (Gurdak and Roe, 2009). In contrast, saline lakes function as discharge wetlands (Hall, 2001; Allen, 2005); that is, their distribution is limited to areas where groundwater makes contact (or where it made contact, historically) with the surface, which occurs in only a few places of the SGP. They are found in association with terrain incised by erosion or at the low points of pluvial paleolakes, thus the local topography generally has some relief. Playas and saline lakes, however, differ in terms of their ecohydrologies and associated biota.

Playa distribution in the SGP corresponds closely with the Ogallala aquifer, although the greatest density is found on Llano Estacado, or the Staked Plains, of the western Texas panhandle and eastern New Mexico (Smith, 2003). Estimates of playa abundance range from 25,000–37,000 (Smith, 2003) to 80,000 playas (Playa Lakes Joint Venture, 2014), although estimates for the SGP proper are not available in the literature. Playa size varies from less than 1 to more than 400 ha, although 86 percent are less than 12 ha (Smith, 2003). Most playas are circular, flat-bottomed, and shallow (less than 2 m deep) with little gradient in depth. They undergo annual to multiyear cycles of inundation, primarily during heavy thunderstorms in late spring and summer, followed by periods of drying out from late autumn through early spring. Because thunderstorms are highly localized and surface runoff is the only natural source of inflow to playas, these wet-dry cycles are erratic. As such, a given playa may be inundated repeatedly in a year or it may remain dry for more than a decade. There are many playas, however, that also receive irrigation tailwater, which alters their ecohydrology and biota.

In contrast to playas, saline lakes are rare in the SGP. There are 40–50 on the Llano Escatado between Lubbock, Texas, and Portales, New Mexico (Hall, 2001; Andrei and others, 2008), about 70 in the Estancia Basin of east-central New Mexico (Allen, 2005), and a small number scattered elsewhere in the SGP. Compared to playas, saline lakes are much older and larger (up to 112,500 ha), more irregular in shape, and are usually deeper with a greater depth gradient; they are typically associated with draws, escarpments, or mountains, and many have well-developed leeward dunes (Reeves and Temple, 1986; Hall, 2001; Allen, 2005). As such, the watershed areas of these lakes can be quite large (up to 12,600 square kilometers [km^2]), and lake size and hydroperiod may vary significantly with short- and long-term changes in climate. The Lubbock-Portales lakes are fed by freshwater springs from the Ogallala aquifer; thus, many were or still are semipermanent bodies of water. The Estancia lakes, however, are remnants of a much larger, pluvial paleolake that once filled the basin. Currently, most of the lakebeds remain dry throughout the year except when there is significant rainfall. Rainfall seasonality in both the Llano Estacado and Estancia Basin is similar to that already described for playas, although the average amount of rainfall varies more locally.

Although playas represent only 2 percent of the southern High Plains landscape area (Haukos and Smith, 1994), they contribute disproportionately to SGP biological diversity. More than 350 vascular species have been found growing within the various physiognomic zones of playa basins (Smith, 2003). To migrating birds, the seeds and tubers of playa basin plants provide excellent nutritional value when the birds' energy demands are high. Plant communities of playas vary widely by region, individual wetland, season, and hydroperiod, but some of the more commonly found vascular plant species include pondweed (*Potamogeton* spp.), smartweed (*Polygonum* spp.), cattail (*Typha* spp.), bulrush (*Schoenoplectus* spp.), devilweed (*Chloracantha spinosa*), ragweed (*Ambrosia* spp.), barnyard grass (*Echinochloa crus-galli*), sprangletop (*Leptochloa* spp.), Johnsongrass (*Sorghum halepense*; nonnative and listed in some SGP states as a noxious weed), buffalograss (*Bouteloua dactyloides*), and western wheatgrass (*Pascopyrum smithii*) (Smith, 2003).

By contrast, floral diversity in saline lakes is limited because of higher levels of total dissolved solids (more than 200 grams per liter [g/L]). The water of some freshwater springs carry relatively high levels of dissolved sodium, chloride, and other major ions, and because the rate of evaporation generally exceeds the rate of inflow, these ions can accumulate and lead to high salinity levels in closed basins. Salinity also can vary seasonally with variation in discharge rates and surface inflows (Stout, 2003), which strongly affects the biota of saline lakes. A floral survey of seven saline lakes during spring and fall sampling periods in northwestern Texas and northeastern New Mexico yielded only 49 vascular plant species (primarily Asteraceae, Amaranthaceae, Cyperaceae, and Poaceae), 6 of which were nonnatives (Rosen and others, 2013). Most of the 49 species were found in just one large lake, with significantly fewer species found in the other lakes. Nonetheless, these wetlands and those in the Estancia Valley do provide crucial wildlife habitat, particularly for waterbirds (Mitchusson, 2003; Andrei and others, 2009).

Landscape Structure and Dynamics

The SGP playas are relatively young compared to the saline lakes, and new ones still form today through both natural and anthropogenic processes; their origin and formation, however, are much debated. Primary hypotheses include geohydrological processes of dissolution and subsidence within the underlying caprock (caliche), eluviation and wind deflation, and possibly other phenomena (Smith, 2003). Most likely it was some combination of these processes, which vary in their degree of influence across the SGP region. The origin and formation of saline lakes in the SGP are notably different from those of playas. Hypotheses include dissolution and collapse of underlying limestone or other deposits; incising from Pleistocene-Holocene surface flows; wave action, deflation and aeolian deposition; fractures and faulting; and (or) a combination of these and other factors (Reeves and Temple, 1986; Hall, 2001; Allen, 2005).

In the SGP, playa density and size tend to be greater where average rainfall amounts are greater, which increases along a southwest-to-northeast gradient (Smith, 2003). Soil texture is believed to influence playa density as well, although the exact nature of this relationship is unclear. Soil texture and wind also influence playa size in that basins with finer textured soils are more readily deflated by wind, which can lead to playa enlargement (Smith, 2003). The size of saline lakes, however, is largely a function of the terrain and watershed area. Wind also creates lunettes on the leeward edges of saline lakes and playas that generally or frequently have dry, unvegetated floors from which wind can remove loose substrates and carry them downwind.

Most playa floors are lined with hydric vertisol clay soils, which form cracks when the soils dry out. Within the first few days of playa inundation, water is able to drain through the cracks and into pathways that eventually enter the underlying aquifer. Once the clay has swelled, however, the basin floor becomes relatively impermeable, and the remaining water either seeps through the playa annulus (edge) and (or) is lost primarily through evapotranspiration, thus providing a prolonged period of inundation (Smith, 2003; Gurdak and Roe, 2009). As a result of these dynamics, the rates of recharge from playas to the underlying aquifer are 1–2 orders of magnitude greater than they are from interplaya areas (Gurdak and Roe, 2009).

The spatial distribution of isolated wetlands has important ecological consequences. In concert with the highly localized nature of rainfall that inundates them, their relative isolation promotes differences in community composition across the region and even among neighboring wetlands. On the other hand, their connectivity across broad landscapes is important to many wildlife species that use them for foraging, nesting, and (or) roosting (Smith, 2003; Andrei and others, 2009). Highly mobile wildlife species, such as birds, large mammals, and bats, are able to track the spatial and temporal dynamics in composition, cover, and water across the SGP landscape. Where playa density is low, movements of smaller, less mobile aquatic wildlife are likely more restricted than they are where playa density is greater.

For playas and perhaps saline lakes of the Llano Escatado, fire, herbivory, and (or) activities of large mammals undoubtedly have played some role in their formation, persistence, and (or) ecology, but few studies have been conducted to understand these processes and their effects (Smith, 2003). Natural fire regimes in playas and saline lakes are virtually unstudied, but historically they probably mirrored the frequent fires of the surrounding prairie uplands. Historically, prairie fuels likely supported primarily low-intensity burns, although, after particularly productive growing seasons, the standing crop in playas may have been sufficient to promote greater fire intensity. Fire in playas precludes woody vegetation, recycles nutrients and increases the nutritional value of new vegetative growth, and promotes species diversity (Samson and Knopf, 1996; Smith, 2003). The effects of fire exclusion in playas is not well studied, but cattails in some playas can become dominant to the exclusion of other species, so burning and grazing, especially in cattail-dominated playas, has been a common practice for reducing fuel loads and managing vegetation. The effects of disturbances, natural or anthropogenic, to the surrounding uplands of playas and lakes can have significant effects on the wetlands as well. Disturbances that remove a significant amount of the vegetative cover in a given watershed render the wetland more susceptible to sedimentation (Burris and Skagen, 2013). By the same token, buildups of mulch and rank growths of tall, dense vegetation are capable of greatly diminishing the flow of water into the wetland (Melcher and Skagen, 2005). Therefore, although disturbances in the surrounding watershed are necessary for maintaining vegetative diversity and vigor, both chronic disturbance and exclusion of disturbance can have profound implications for the wetland's biotic community structure and ecology.

Associated Species of Management Concern

Playas support a tremendous diversity of vascular plants and algae. The vegetation composition differs not only across physiognomic regions of the SGP, but it also differs among two to three distinct vegetative zones within individual playas (Smith, 2003). The vegetation community also may turn over completely as the water recedes after inundation, with aquatic plants being replaced by grasses and other terrestrial species (Haukos and Smith, 1997; Tsai and others, 2012; Rosen and others, 2013). Compared to the surrounding uplands, these wetlands also support a disproportionately large diversity of native wildlife, including more than 115 bird species, 10 mammal species, 13 amphibian species, and many invertebrate species (Smith, 2003). Wetland birds in particular rely on playa habitats for winter cover or roosts, migration stopovers, and nesting habitat. The bare ground that may follow receding waterlines also provides important foraging habitat for many nesting or migrating shorebird species, and when dry, the residual standing crop on a playa floor may serve as nesting habitat or winter cover for grassland birds.

Fauna assessed in this REA that use isolated wetlands in the SGP for migration stopovers, nesting, foraging, and (or) roosting include the snowy plover (*Charadrius nivosus*) (Chapter 18), long-billed curlew (*Numenius americanus*) (Chapter 16), interior least tern (*Stena antillarum athalassos*) (Chapter 14), and pallid bat (*Antrozous pallidus*) (Chapter 19). Bird species listed as priorities for the Great Plains Landscape Conservation Cooperative that use playas and (or) saline lakes include northern pintail (*Anas acuta*), sandhill crane (*Grus canadensis*), least sandpiper (*Calidris minutilla*), western sandpiper (*Calidris mauri*), long-billed dowitcher (*Limnodromus scolopaceus*), and Wilson's phalarope (*Phalaropus tricolor*). Although amphibians contribute less to the overall species diversity in playas than birds, in biomass they far surpass other vertebrates and contribute heavily to the overall prey base exploited by all major taxonomic groups (Smith, 2003). Comprehensive listings of flora and fauna known to associate with playas and saline lakes are provided in Haukos and Smith (1997), Smith (2003), and Andrei and others (2008, 2009).

Change Agents

Although some States and local jurisdictions provide some protection for isolated wetlands from unregulated filling, dumping, and dredging (for example, see Larimer County, 2014), a 2001 ruling by the Supreme Court now excludes most isolated wetlands from protection by the Federal Clean Water Act (Downing and others, 2003; Leibowitz, 2003). Although a proposal to overturn this ruling is in the works, it remains unclear whether protection for isolated wetlands will be restored. An estimated 70 percent of isolated wetlands in the United States have already been altered by various anthropogenic activities, including agricultural practices; contamination from energy development, pesticides, fertilizers, and feedlot runoff (Irwin and others, 1996); groundwater withdrawals; and invasive species and diseases. Furthermore, these wetlands are poorly understood because most of them occur on private land, and although playas have been studied significantly more than saline lakes, more research is needed for both types. Overall, in-depth knowledge of the ecology and function of playas and saline lakes represents a data gap for this ecoregion.

Development

Energy and Infrastructure

Energy development and its associated infrastructure have been affecting playas in the SGP for many years. Energy development companies have dumped produced waters (brines) and other contaminants associated with oil and gas production into playas (Irwin and others, 1996), and oil and gas wells, access roads, and pipelines have been built in and around playas. Several shale-gas and oil formations that underlie the SGP are undergoing intensive energy development via hydraulic fracturing, including the Niobrara play in eastern Colorado, the Barnett play in north-central Texas, and the Woodford play in central Oklahoma (U.S. Energy Information Administration, 2011). Because groundwater withdrawals for irrigation and municipal use are already having notable effects on the discharge rates of springs that discharge into saline lakes of the SGP (Hall, 2001; Andrei and others, 2008; Rosen and others, 2013), additional withdrawals for hydraulic fracturing could further affect the hydroperiod and ecological function of these wetlands.

The SGP encompasses some of the best areas for wind energy potential in the United States. There are a couple of ways in which wind energy could affect the biotic communities of playas and saline lakes. First, the vertical structure and movement of wind turbines, as well as the associated powerlines, can represent significant disturbances to wildlife of grasslands, open shrublands, and wetlands, as wildlife typical of such habitats are generally not accustomed to having significant overhead structures in their habitats. Studies have shown that bird abundance can decrease in the vicinity of wind farms, and collision fatalities can be significant (Stewart and others, 2005). The declines in abundance can occur as the result of avoidance, which effectively eliminates the usefulness of those habitats. Birds also frequently collide with powerlines, especially when they are sited near wetlands where waterfowl and shorebirds congregate. Perhaps even more than birds, bats are frequently killed at wind turbines (Horn and others, 2008). Developing and maintaining the infrastructure for all forms of energy development may further affect isolated lakes by increasing their sediment loads due to significant surface disturbances within their watersheds.

Agricultural Activities

Crop production, irrigation, grazing, and filling have been the primary agricultural drivers of change to playas in the SGP. Tilling and disking or filling playa floors are common practices to increase crop production area, while pitting and ditching of playas are commonly practiced to enhance their water-holding volume for irrigation and livestock (Smith, 2003). These practices alter a playa’s structure, hydrology, and function, ultimately altering or eliminating its communities. The accumulation of sediments is also a major concern. Sediments are mobilized by precipitation and irrigation runoff from surrounding croplands or overgrazed rangelands and carried to playa floors, thereby decreasing playa water volumes and altering their hydrological function (Luo and others, 1997). Recent modeling efforts predict a total loss of playas to sedimentation by the early 2100s if practices are not changed (Burris and Skagen, 2013).

Groundwater withdrawal for irrigating crops and providing livestock watering ponds is an important driver of change for saline lakes. The discharge rates in 75 percent of the saline lakes on the Llano Estacado are greatly diminished or the springs have dried up altogether, which is having notable effects on the biota of these systems (Smith and Anderson, 1982; Stout, 2003; Andrei and others, 2009; Rosen and others, 2013). Another negative outcome of overdrafting the groundwater is saltwater intrusion into underlying aquifers in the region (Haukos and Smith, 1997; Gurdak and Roe, 2009). Playas that receive irrigation tailwater from croplands may remain wetter longer than they did historically (Haukos and Smith, 1997), which affects their vegetation communities and the wildlife that use them (Tsai and others, 2012). Tailwater also carries pesticides, heavy metals, sediments, and excessive nutrients into the wetlands, which can further alter the biotic communities.

Historically, grazing by large mammals was undoubtedly an important force in shaping the North American prairies and associated wetland habitats, but little is known about the effects of livestock grazing in playas or the differences between the effects of native grazers and livestock grazing. It has been suggested that forage-species selection and grazing pressure by livestock are probably different from those of native grazers (Smith, 2003). At least into the mid-1980s, larger playas were more likely to be grazed than smaller playas, and some of these grazed playas were found to have altered vegetative communities (Guthery and Stormer, 1984). If livestock are allowed to graze in playas during the growing season, seed production may be considerably diminished, and buffalograss and cocklebur (*Xanthium* spp.) have been observed to replace western wheatgrass and vine mesquite (*Hopia obtusa*) on heavily grazed playas. Fairly intense grazing, however, may generate or maintain habitat for birds that use relatively unvegetated wetlands, such as shorebirds and some waterbirds. Intensive grazing in the uplands around playas also has resulted in shifts from grassland to hummocky mesquite dunes in some areas of the SGP, which in turn has altered runoff characteristics and sedimentation rates from surrounding watersheds.

Altered Fire Regime

Fire has been an important force in shaping the North American grasslands; however, historically the SGP ecoregion was relatively treeless, thus fire scars useful for reconstructing natural fire regimes are lacking. Studies of lakebed charcoal deposits in the northern Great Plains indicate that fire frequency decreased after European settlement, and although it may be surmised that the same trend occurred in the SGP, such studies there are lacking (Ford and McPherson, 1996). Compared to current fire patterns, historical accounts of fire in desert grasslands indicate a reduction in fire size and frequency, most likely due to fire suppression and livestock grazing practices. Fire likely helped to reduce fine fuels, eliminate invasive woody vegetation, and stimulate plant growth both in and around isolated wetlands. Today, prescribed fire is used to accomplish these processes, but the spatial scale at which they are conducted is likely very different than that of historical wildfires.

Invasive Species

Exotic and invasive species are a tremendous threat to playas, particularly in playas in which cultivation has taken place, sediment loads are high, and (or) the hydroperiods have been altered (Smith, 2003). A survey of plant species in saline lakes revealed six nonnative plant species: rescuegrass (*Bromus catharticus*), Canada bluegrass (*Poa compressa*), annual rabbitsfoot grass (*Polypogon monspeliensis*), common sowthistle (*Sonchus oleraceus*), burningbush (*Kochia scoparia*), and tamarisk (*Tamarix ramosissima*) (Rosen and others, 2013). Species of special concern for playas include tamarisk, reed canary grass (*Phalaris arundinacea*), and purple loosestrife (*Lythrum salicaria*) (Smith, 2003; Lavergne and Molofsky, 2004). Fire suppression also may have contributed to the expansion of both native and nonnative woody plants in and immediately around isolated lakes of the SGP.

Insects and Disease

Although disease does not affect isolated wetlands directly, diseases can emerge in isolated wetlands when their properties are altered by other CAs described herein. For example, outbreaks of botulism and avian cholera can cause significant die-offs of waterfowl, shorebirds, and other waterbirds (Friend, 1982; Nelson and others, 1983; Haukos and Smith, 1992; Irwin and others, 1996). Although not completely understood, factors that can lead to these outbreaks include high pH and warm water temperatures, chemical composition, and anaerobic conditions. For example, excessive runoff of nutrients from fertilizers and manure can flow into isolated wetlands in association with sediments, often creating the conditions necessary to promote anaerobic conditions that can lead to outbreaks of botulism. Avian cholera in the SGP playas is more problematic among wintering waterfowl, whereas botulism tends to occur in late summer and fall.

Climate Change

Although average annual precipitation over most of the SGP increased from 1958 to 2008 (Karl and others, 2009), some projections for 2009–2099 indicate that annual average precipitation may decrease in the SGP, especially in the southwestern portion of the region. The greatest differences are projected to occur in spring and summer (Karl and others, 2009), which is when playa inundation typically occurs. On the other hand, amounts of very heavy rain also are projected to increase, which could help to offset potential increases in evapotranspiration due to warmer temperatures. In either case, resulting hydroperiods could differ from historical hydroperiods. Collectively, these changes have the potential to alter playa hydrology and formation, recharge rates from playas to the Ogallala aquifer, discharge rates to saline lakes, and the rates of evapotranspiration. Overall, the hydroperiods may become shorter and (or) the wetlands may remain dry for longer periods. Further declines in groundwater recharge driven by these and other changes could eliminate the hydroperiods of the few saline lakes that still have functioning springs (Gurdak and Roe, 2009).

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting playas and saline lakes is illustrated in figure 12. Ecological attributes and CAs identified in the Rapid Ecoregional Assessment are enumerated in tables 13 and 14.

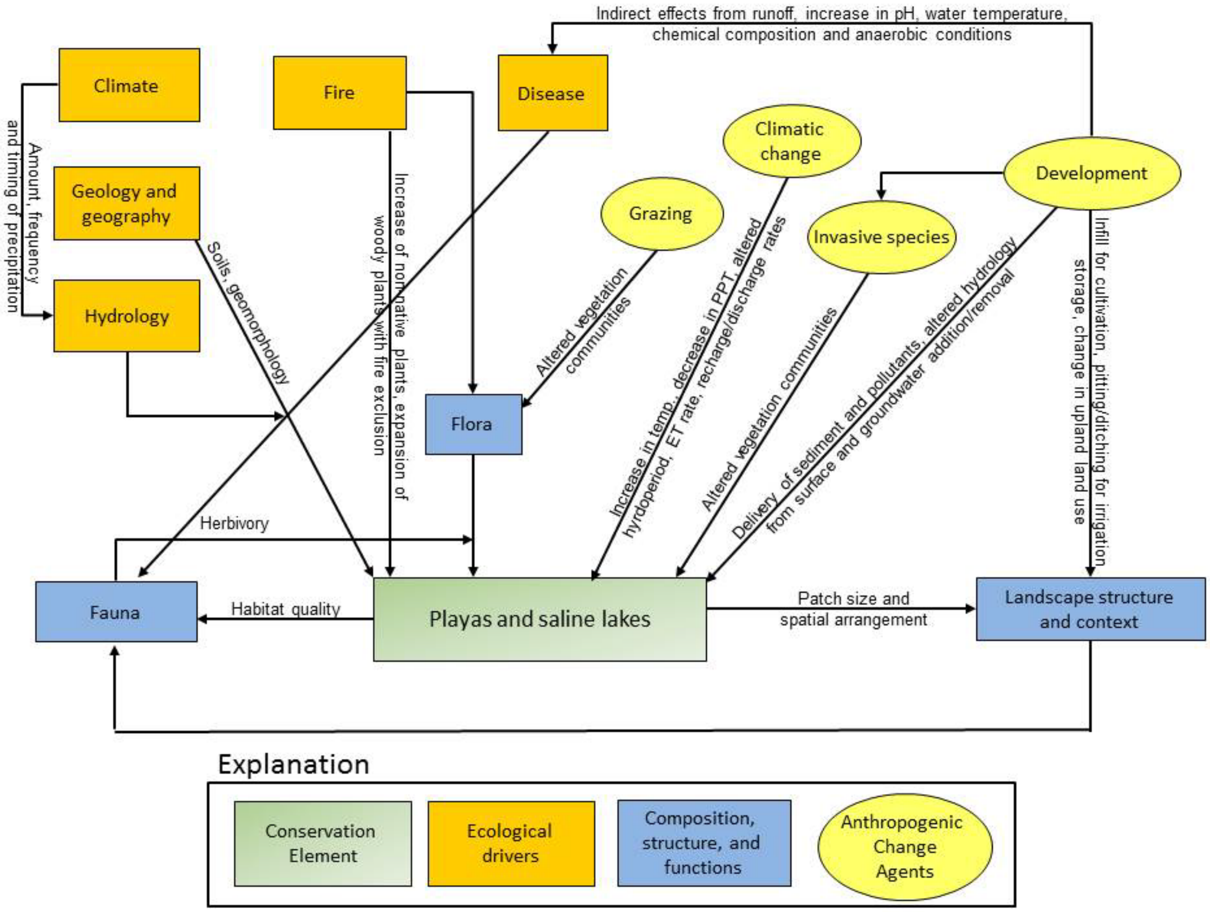

Figure 12. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for playas and saline lakes in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of playas and saline lakes are shown in orange rectangles (see also table 13); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 14).

Table 13. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for playas and saline lakes.

Attribute	Variables
Amount and distribution	Mapped distribution of all playas and saline lakes, geology and geography (geomorphology, soils).
Landscape structure	Size and spatial distribution (area, density, connectivity).
Landscape dynamics	Hydrology (hydroperiod, recharge or discharge type), fire regime (frequency and severity), climate variability (drought).
Associated species of management concern	Snowy plover, long-billed curlew, interior least tern, and pallid bat.

Table 14. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for playas and saline lakes.

Attribute	Variables
Development (energy and infrastructure)	Hydrological regime: altered hydrology (surface and groundwater addition/removal). Sedimentation regime: upland surface disturbance (delivery of sediments and pollutants). Connectivity: isolation (upland surface disturbance). Water quality: nonpoint source runoff (delivery of sediment and pollutants).
Development (agricultural activities)	Alteration (changes to vegetation communities, reduction in seed production). Hydrological regime: altered hydrology (surface and groundwater addition/removal). Sedimentation regime: alteration (pitting/ditching for irrigation storage). Connectivity: alteration (infill for cultivation). Water quality: nonpoint source runoff (delivery of sediment and pollutants).
Fire	Expansion of woody plants (via fire exclusion).
Invasive species	Altered vegetation communities.
Insects and disease	Increase in disease from indirect effects of development, runoff, invasive species, and climate change (increases in pH and water temperatures, altered chemical composition, anaerobic conditions).
Climate change	Habitat alteration (plant community changes), hydrologic regime (altered hydroperiod, evapotranspiration rates, recharge/discharge rates).

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Chapter 8. Lakes and Reservoirs

By Lucy E. Burris

Key Ecological Attributes

Distribution and Ecology

Lakes are water impoundments created by natural processes along stream courses and at depressional locations on landscapes. Generally, lakes include one to several inflows, either surface flows (streams) or groundwater (springs), and a single unrestricted outflow, allowing free movement of aquatic species between upstream and downstream reaches. Lakes lacking an outflow are generally saline and are discussed in Chapter 7, "Playas and Saline Lakes."

In contrast to lakes, reservoirs are manmade impoundments developed to control flooding, enhance water storage for use offstream (for example, irrigation), and to create hydraulic head for electric power generation. Many natural lakes, such as Lake Meredith in Texas, were converted to reservoirs by dam construction. In addition to Lake Meredith, other large reservoirs created by damming rivers or converting large lakes to reservoirs in the SGP are John Martin reservoir in Colorado, Cedar Bluff and Cheney reservoirs in Kansas, Tom Steed reservoir and Great Salt Plains Lake in Oklahoma, and Lake Kemp in Texas. Although reservoir expansion during the 1970s significantly increased the amount of open water in the SGP compared to earlier periods, lakes and reservoirs are relatively sparse and represent important sources of open water for birds, fish, and human activities (Sprague and others, 2002; Havel and others, 2005). In northeastern Colorado for example, reservoirs provide important stopover habitat during fall shorebird migration (Andres, 2007).

Reservoirs tend to be shallower and have more irregular shorelines than lakes, and typically they are much younger (often less than 50 years old); have higher levels of suspended solids and greater rates of sedimentation and therefore are more turbid and have shorter life spans; have greater nutrient loading; and are more saline (Havel and others, 2005; Johnson and others, 2008). Fish and zooplankton biomass is similar in lakes and reservoirs, but benthic invertebrate biomass is greater in lakes. Shoreline vegetation along reservoirs associated with hydropower dams is often poorly developed because of significant fluctuations in water levels. Nationally, siltation, which includes suspended solids, turbidity, sediments, and shoreline erosion, has been identified as one of the biggest effects on fish habitat in reservoirs (Miranda and others, 2010). Other concerns about habitat impairment include the lack of aquatic macrophytes, absence of woody debris, and poorly timed changes in water levels.

Although dams restrict the upstream and downstream movements of fish, and comparatively uncontrolled streams are rare in the SGP, reported losses of native fish species after impoundment development have been limited. Increases in reservoir-dwelling and introduced species, however, have been reported (Gido and others, 2002; Falke and Gido, 2006). In particular, reservoirs allow the expansion of species that cannot tolerate strong, unidirectional flows, and they reduce habitat required by shallow-water species (Havel and others, 2005); however, it may not be the presence of reservoirs, per se, that drives increases in the number of introduced species (Gido and others, 2004). Rather, the young ages of reservoirs, their high connectivity, and their high frequency of disturbance may enhance opportunities for invasive species, particularly pioneer species (Johnson and others, 2008). Also, traits of fish species found in reservoirs tend to mirror those found in large streams, such as species that consume plankton and invertebrates, distribute their eggs by broadcasting them, and prefer gravel and sand substrates (Gido and others, 2009). Indeed, streams in watersheds with the highest human population densities have been found to have the highest number of introduced species, possibly because of the easy introduction of bait-bucket and sport fish. Future research is needed to better understand the interactions between native and introduced species, including changes in presence and abundance after reservoir construction.

Reservoirs also affect sediment and nutrient flows to downstream reaches. Upstream inflows and surface runoff, particularly in agricultural areas, can be enriched with nitrogen from inorganic fertilizers and animal wastes, can be contaminated with chemicals, and can contain significant amounts of fine-grained sediments (Sprague and others, 2002; Baron and others, 2013). Residence time (the period of time spent in the reservoir before passing downstream), water depth, and maximum reservoir length all contribute to the degree of deposition of these components (Sprague and others, 2002; Cunha and others, 2014). Reservoirs may entrap up to 88 and 86 percent of nitrogenous and phosphorus input, respectively (Sprague and others, 2002). As reservoirs age, however, nutrient loading in reservoir sediments appears to stabilize rather than continue to increase, which suggests the possibility that there is a maximum capacity for nitrogen processing (Baron and others, 2013). Microbial denitrification is enhanced by longer residence times and warmer temperatures (Sprague and others, 2002; Baron and others, 2013), and phosphorus inputs also can enhance nitrogen removal. Indeed, lakes with low phosphorus levels can have much higher levels of nitrogen, thus increasing the risk of eutrophication (excessive algal growth that leads to low oxygen levels). In the SGP, however, this risk may be comparatively low compared to that of well-watered regions like the southeastern United States where intensive agriculture is more prevalent (Miranda and others, 2010; Finlay and others, 2013). Erosion control and sewage treatment have largely removed phosphorus inputs to lakes and reservoirs, but nitrogen has been less controlled because of the more dispersed sources, including atmospheric input, soil leaching, and surface runoff (Finlay and others, 2013).

Landscape Structure and Dynamics

Lake water levels fluctuate seasonally and annually in response to variation in stream inflow and rates of evaporation. In the Great Plains, sources of streamflow transition along a north-to-south gradient from primarily groundwater (snowmelt) to primarily rainfall (Kustu and others, 2010). Therefore, changes in winter snow accumulation, warm season precipitation, and groundwater depth (which can be lowered substantially by tapping and pumping from aquifers) have different effects on streams and, consequently, lakes within their drainages. Since the 1940s and the onset of intensive irrigation, flow in most streams of the western Great Plains have been diminished. In contrast to lakes, water levels in reservoirs frequently are controlled by water-use agreements, power-generation requirements, and flood-control decisions. Because of flow restrictions, reservoirs entrap more sediments than lakes, and they even out streamflow by reducing overall annual discharges, increasing minimum discharges, and reducing maximum discharges (Costigan and Daniels, 2012). Moreover, the timing of annual maximum discharge from an impoundment is typically a month later than it was prior to impoundment, potentially disrupting reproductive cycles of fish and pulse-synchronized seed dispersers, such as cottonwood (*Populus* spp.).

Lake and reservoir management is complicated by various ownerships and management goals at all levels. For example, water level at Lake Meredith in Texas is controlled by the U.S.

Bureau of Reclamation Sanford Dam under the Canadian River Compact, an agreement between New Mexico, Texas, and Oklahoma and operated by the Canadian River Water Authority. The surrounding area is a national recreation area managed by the National Park Service, and fishing and hunting are regulated by the Texas Parks and Wildlife Department. The primary objectives of this impoundment were to supply water to 11 cities in the area, control flooding, create fish and wildlife habitat, and provide recreation opportunities. These objectives can be conflicting, however. For example, during summer, increases in both municipal and agricultural water use for irrigation result in drawdowns that can reduce boating and fishing access and recreational value. Even small impoundments can have conflicting demands. For example, Lake Kemp in Texas (6,300 ha) is a public water body surrounded by private land with controlled access requiring a daily use permit. Fishing is regulated by the Texas Parks and Wildlife Department, and water levels are controlled by the City of Wichita Falls, Texas. In the past, water levels have dropped enough to preclude the use of most private and public boat ramps (Chizinski and others, 2005).

Many small flood-control reservoirs built since the 1940s were designed with a capacity for 50 years of sedimentation; impoundments since the 1960s have been designed with a 100-year life span (Hanson and others, 2007). For many of these impoundments, no initial survey was conducted, so their current sediment levels are unknown (Dunbar and others, 2013). As an impoundment fills, pools become shallower and water temperatures increase, and eventually the reservoir becomes a “reservoir wetland” with no storage capacity (Powers and others, 2013). Sedimentation processes, however, are highly variable—depending on surrounding land use, topography, and precipitation patterns—and the relationships influencing sedimentation can be nonlinear. For example, sediment yields are highly influenced by variations in annual precipitation: a 33 percent variation in annual rainfall can result in a 100 percent variation in sediment yield (Garbrecht, 2008).

Associated Species of Management Concern

In general, little land area in the SGP has protection status for maintaining biodiversity (Aycrigg and others, 2013). To the extent that this holds for aquatic systems as well, secure environments for species of concern associated with lakes and reservoirs may be limited. Some of the freshwater mussel species that occur in the SGP ecoregion, such as the pondhorn (*Uniomerus tetralasmus*), inhabit muddy-bottomed lakes (see Chapter 10). In western Kansas, an extensive survey of streams revealed very limited occurrences of freshwater mussels due to the semiarid conditions and limited occurrences of free-flowing freshwater (Angelo and others, 2009). Historically, however, mussel distributions may have been more extensive in the west, as indicated by the discoveries of weathered shells and specimens in museum collections.

Although playa wetlands and saline lakes (Chapter 7) provide suitable nesting habitat for shorebirds, including the snowy plover (*Charadrius nivosus*) (Chapter 18), steep-banked freshwater lakes and reservoirs are unlikely to provide the shallow water and mudflats required for these species (Conway and others, 2005b). If wide, sparsely vegetated mudflats and shallow water are available along lake or reservoir shorelines; however, these habitats can provide suitable nest sites with extended nesting periods (much like saline lakes) compared to those of playa wetlands with shorter hydroperiods (Conway and others, 2005a). To the extent that reservoir habitat mirrors saline lake habitat in vegetation structure, freshwater availability, and invertebrate food resources, migrating snowy plovers and long-billed curlews (*Numenius americanus*) (Chapter 16) will use reservoirs as stopovers during spring and fall migration

(Andrei and others, 2006, 2008; Andres, 2007). Although evidence is limited, the persistence of reservoirs during dry years may provide the only suitable habitat for these species during very dry years when playa habitats are dry and have significant vegetation cover (S. Skagen, wildlife biologist, U.S. Geological Survey, oral commun., March 6, 2014).

Change Agents

Development

Increasing human densities will lead to increased use of lakes and reservoirs, for both recreation and municipal water needs (including thermoelectric cooling) (Roy and others, 2012).

Energy and Infrastructure

Effects of energy development can include potential changes to groundwater flows and contamination from hydraulic fracturing (making spring-fed lakes particularly at risk), road and pipeline construction disturbance, and contamination from surface spills. Shale-oil fields underlie most of the SGP, so development risks exist across the region. Energy development may also contribute to land cover change resulting in increased erosion.

Agricultural Activities

Land-use change such as the conversion of grassland to cropland in the SGP during the 1970s and 1980s or from increased livestock grazing without appropriate buffer strategies, conservation tillage practices, and irrigation management can lead to increased sedimentation and nutrient inputs in affected watersheds (Chaney and others, 1990; Skagen and others, 2008; Drummond and others, 2012). Grassland conversion is anticipated to continue through the year 2100, primarily because of cultivation and to a lesser extent urban and other development (Sohl and others, 2012). Conversion from irrigated to dryland farming will reduce water lost to evaporation and leakage from storage lakes and reservoirs. On the other hand, both direct and indirect inputs of irrigation tailwater could diminish as more farmland is transitioned to dryland farming. Another significant factor that could affect water levels in managed reservoirs is the transfer of water and the sale of water rights (particularly to municipalities), as water may be diverted to remote areas and water storage may be discontinued or significantly decreased (Crifasi, 2005; Wiener and others, 2008).

Altered Fire Regimes

Increased fire intensity or frequency has the potential to remove or suppress vegetation surrounding lakes and reservoirs. Vegetation loss results in increased runoff, more highly contaminated inflows, increased inputs of sediments and nutrients, and increased bank erosion due to greater runoff velocities.

Invasive Species

As of 2009, the zebra mussel (*Dreissena polymorpha*) was not present in any western areas of Kansas (Angelo and others, 2009). Since then, zebra mussels have been recorded in Texoma (in 2009), Ray Roberts (in 2012), and Lewisville (in 2013) Lakes in central

Texas/Oklahoma (Churchill, 2013). The shallow water and elevated water temperatures common in SGP aquatic systems, however, are not conducive to zebra mussels, thus their populations cycle as conditions change. There is some indication that zebra mussels are early invaders that can be outcompeted by quagga mussels (*Dreissena bugensis*) (Mills and others, 1999). Because they can attain extremely high population numbers, both species can alter the benthic environment substantially and displace native bivalves (Strayer, 2009) (see Chapter 10). Their high filtering rates remove phytoplankton and zooplankton, resulting in fewer pelagic fish, lower dissolved oxygen levels, increased water clarity, more aquatic plants, and larger littoral fish populations (Strayer, 2009; Higgins and Vander Zanden, 2010). Because both invasives are small, they are easily spread by human activities (for example, in boat ballast or attached to boat hulls), and they are difficult to control once established. They attach to all types of below-water surfaces, and as their numbers grow, they can clog intake pipes, damage hydroelectric systems, and create drag on boat hulls and buoys.

Asian clams (*Corbicula fluminea*) have been found in western Kansas in the Cimarron, Upper Arkansas, Smoky Hill/Saline, and Solomon River drainages (Angelo and others, 2009). Lakes and reservoirs in these drainages are also likely at risk for invasion because clams are easily transported and transferred among basins by human activities (Sousa and others, 2008). Asian clams reproduce rapidly, occupy habitat used by other species, and deplete benthic food resources. Like other mollusks, attachment to structures such as water intake pipes can create significant infrastructure maintenance problems.

Introduced Insects and Disease

The Great Plains has experienced a high incidence of West Nile virus, which is spread by *Culex* spp. mosquitos (likely *C. tarsalis*), particularly in irrigated, undeveloped, and grassland areas (DeGroote and Sugumaran, 2012). Margin areas of lakes and reservoirs provide important breeding and larval habitats for these mosquitos (Barker and others, 2009). Several sister species (*C. pipiens*, *C. quinquefasciatus*, *C. restuans*) have been shown to emerge more rapidly at temperatures above 24 °C, but they also have lower survivorship to adulthood (below 50 percent) and adult longevity (about 0 percent) at 32 °C (Ciota and others, 2014).

Climate Change

The water levels of lakes and reservoirs are driven by both small- and large-scale weather patterns that affect precipitation and temperature. During a period of peak precipitation in the 1980s, most of the moisture increase occurred during fall, winter, and spring, rather than summer months when anthropogenic water demands are greatest (Garbrecht and others, 2004). An annual 12 percent increase in precipitation led to a 64 percent increase in streamflow but only a 5 percent increase in evapotranspiration. Overall, streamflows in the western Great Plains generally have been decreasing since the 1950s even as precipitation has been increasing (including the wet 1980s) (Garbrecht and others, 2004; Brikowski, 2008; Polley and others, 2013). This suggests that water removed for irrigation and municipal water use, for example, is not being returned. This effect, coupled with predictions of future temperature increases and consequent increases in evaporative losses with little change in precipitation, suggests that water input to reservoirs and lakes will diminish. With anticipated human population growth, water shortages in the western Great Plains could become extreme by the middle of the century (Roy and others, 2012). Under climate projection simulations, water levels in reservoirs are anticipated to decline steadily through the next century, resulting in significant reservoir inefficiency (evaporating more than half the inputs from streamflow), eventually becoming inactive (that is, reservoir water levels drop below intake structures and the water becomes stagnant) (Brikowski, 2008).

Although reservoirs are at risk, their role as open water on the landscape may become more ecologically important than it is today. Water-management trends, such as water rights transfers and reductions in irrigation, have the potential to deplete other wetland habitats currently maintained by inflowsarising from water transfers and irrigation run-off (Crifasi, 2005). Playas are at risk from both sedimentation and increased evaporation, which shortens their hydroperiods (Burris and Skagen, 2013). Comparatively, lakes and reservoirs are at lower risk for extensive habitat loss because their water volumes are often regulated for nonhabitat purposes. For example, particularly during dry years, reservoir habitats may provide the only available stopover locations for migrating shorebirds.

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting lakes and reservoirs is illustrated in figure 13. Ecological attributes and CAs identified in the REA are enumerated in tables 15 and 16.

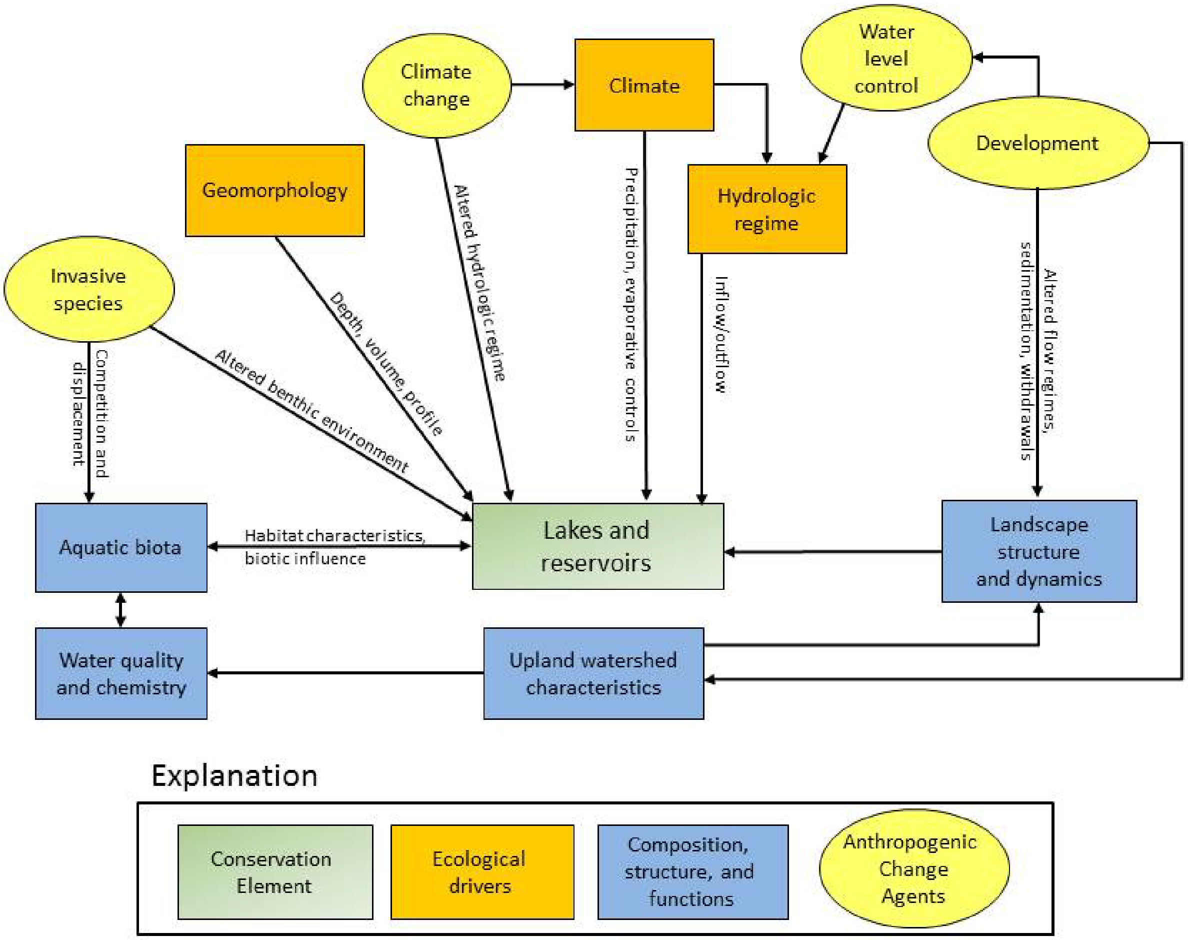

Figure 13. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for lakes and reservoirs in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of lakes and reservoirs are shown in orange rectangles (see also table 15); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 16).

Table 15. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for lakes and reservoirs.

Attribute	Variables
Amount and distribution	Mapped distribution of lakes and reservoirs.
Landscape structure	Size and spatial distribution (area, density).
Landscape dynamics	Hydrologic regime (amount, timing, temporal variability of groundwater and surface water inputs), cycles of wet/dry periods.
Associated species management of concern	Snowy plover, long-billed curlew, and freshwater mussel species.

Table 16. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for lakes and reservoirs.

Attribute	Variables
Development (energy and infrastructure)	Alteration (groundwater depletion, sedimentation and toxics contamination from upland disturbance), discharge of produced waters.
Development (agricultural activities)	Loss (infilling, conversion into agricultural areas), alteration (runoff containing sediments and pesticides from croplands, reduced groundwater input if irrigated land converted to dryland).
Development (human activities)	Increased pressure via recreation and municipal water needs.
Altered fire regime	Alteration from increased intensity and (or) frequency (increased sediment and nutrient inputs, increased runoff velocity, greater rate of bank erosion).
Invasive species	Alteration of benthic system (changes to species communities due to invasions of species such as Asian clam, zebra mussels, and quagga mussels).
Introduced insects and disease	Presence of West Nile virus.
Climate change	Drought effects (decrease in stream flow inputs, increase in evaporative losses).

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Chapter 9. Prairie Streams and Rivers

By James J. Roberts

Key Ecological Attributes

Distribution and Ecology

Streams and rivers of the SGP ecoregion are highly variable in size, type, and flow patterns (Dodds and others, 2004). Their hydrologic and geomorphic characteristics, however, change progressively along a predictable and consistent gradient from their headwaters to the main-stem rivers (fig. 14). Small headwater streams of the SGP undergo intermittent channel drying, especially in areas that lack the groundwater springs that typically promote more consistent flows. The drainage area of medium-sized reaches in the middle sections of a watershed can be engulfed by storm events, thus they are more prone to large flooding events than the upper reaches. In contrast, most of the large, main-stem rivers in the SGP are buffered against all but the largest flood events by virtue of their flood plains, where water can dissipate laterally when large flow events do occur. These larger river sections also are less likely to undergo periods of channel drying than upstream sections.

The channel geomorphology of streams and rivers in the SGP can be coarsely described as meandering and braided, with channel-bed substrates of mostly sand (Costigan, 2013). In the SGP, however, there is a progressive change in channel-bed substrate and structure from the headwaters to the large rivers. Substrate material of headwaters is composed of roughly equal proportions of sand and gravel, but 50–200 km downstream the substrate is primarily sand. Channel-bed slope (elevation drop per unit length of channel) also decreases progressively along the same gradient, and the channel width and width-to-depth ratio increase (Costigan, 2013). These progressions are difficult to see on landscapes today, as myriad stressors, including land-use changes, have significantly altered the geomorphology and flow regimes of SGP streams and rivers.

Prairie streams and rivers support a diversity of primarily omnivorous fauna, including invertebrates such as predatory insects and crayfish and vertebrates such as fish. Macroinvertebrate assemblages in headwater streams of the SGP are dominated by grazers. Farther downstream, where leaf litter input is sufficient to support shredder insects that depend on leaf litter as a forage base, grazers and shredders occur in roughly equal proportions. Within large, main-stem rivers, where there is significant input of leaves from adjacent riparian plants and terrestrial vegetation transported from upstream, the macroinvertebrate assemblage is dominated by shredders. Although these general trends in macroinvertebrate composition and stream size hold, the position of a stream in the watershed has much less influence than the composition of particulate and coarse organic matter on functional feeding groups of the fauna (Dodds and others, 2004).

Landscape Structure and Dynamics

The flow patterns of streams and rivers in the SGP are driven by the highly variable precipitation patterns typical of prairie and plains regions (Poff and Ward, 1989). Snowmelt runoff is the dominant influence on streamflow in the western SGP, and rainfall runoff from

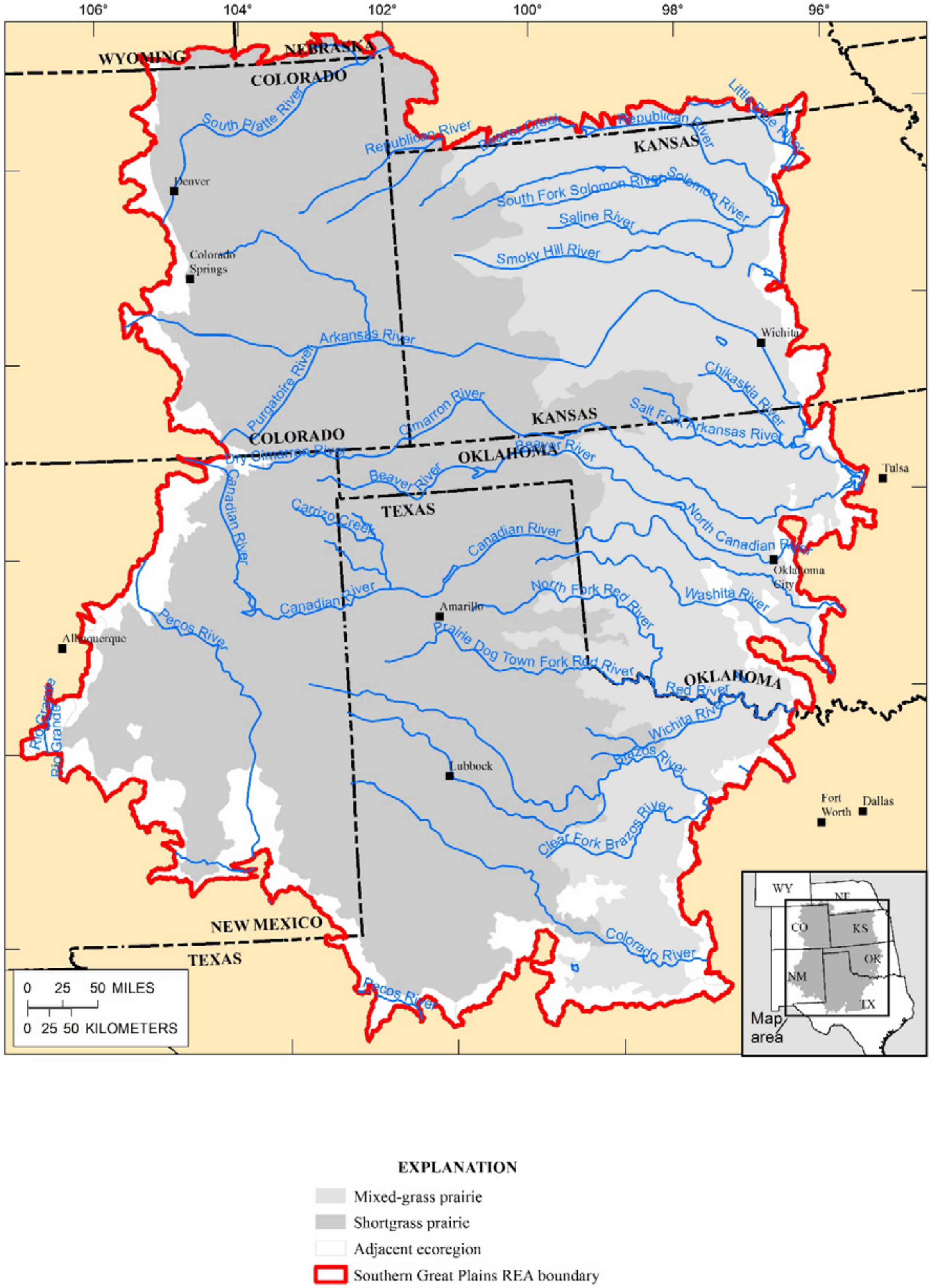

Figure 14. Major rivers of the Southern Great Plains ecoregion. (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (REA, Rapid Ecoregional Assessment)

convective thunderstorms is the dominant influence on streamflow in central and eastern portions of the region. The SGP encompasses three different ecoregions (the Central Great Plains, the High Plains, and the Southwestern Tablelands) that also impart unique characteristics to streams and rivers in those ecoregions. Overall, the vast majority of SGP streams and rivers are characterized by flow regimes of high variability punctuated by periods of large discharge (water volume per unit time) (Poff and Ward, 1989). As a result, their biotic communities (at every trophic level) are highly adapted to frequent disturbance (Lytle, 2002).

The highly variable precipitation patterns and resulting biotic communities in SGP streams and rivers are unique to west-central North America, but riparian plant characteristics are similar to those of prairie grassland systems throughout the world. For example, much of the vegetation, such as C_4 grasses, is dominated by plants adapted to hot, dry climates (Samson and Knopf, 1994). Intermittent streams are numerous, the proportion of intermittent to perennial streams is high, and the vegetation requires little moisture from precipitation, either directly or indirectly (Dodds and others, 2004). Most streams and rivers in the SGP lack a significant riparian canopy, which reduces the input of carbon and nutrients to headwater streams. Geologic characteristics, which affect the consistency of groundwater flowing from springs, can buffer stream habitats from the variable conditions that affect streamflow. The entire SGP sits atop the Ogallala aquifer (Dodds and others, 2004), which allows the persistence of some refugia (pool) habitats and isolated regions of perennial flow during times of extreme drought. This groundwater source, however, has been mined heavily for agricultural irrigation, and in some areas this has led to more incidences of channel drying (Falke and others, 2011, 2012). Groundwater is extracted for municipal use and energy development as well, although little is known about how those uses affect streamflow patterns in the SGP.

The surrounding riparian and upland vegetation also influences the flow regimes and associated ecosystem processes of flowing waters in the SGP. From headwaters to large, main-stem rivers, there is a gradient in riparian vegetation that structures the community and ecosystem dynamics of flowing waters in the SGP (Dodds and others, 2004). Headwater and midorder reaches are dominated by grasses and shrubs, respectively, and large, main-stem rivers are characterized by minimal tree canopy cover (Cross and Moss, 1987). This linear vegetation pattern results in distinct regions of trophic energy structure, specifically as it relates to primary production and community respiration. Upstream headwater reaches with limited canopy cover and clear water allow light to penetrate the water to the stream bottom, resulting in relatively equal ratios of primary production to respiration (Stagliano and Whiles, 2002). Farther downstream in midorder streams and rivers, the primary production-to-respiration ratio drops and continues to decline along the downstream gradient to the larger main-stem rivers. In turn, the riparian vegetation composition, structure, and function influence the structure of aquatic macroinvertebrate communities (Dodd and others, 2004).

Given the variable nature of streams and rivers in the SGP, the recolonization of disturbed habitats is important for maintaining proper functioning of these unique systems, particularly in the frequently flooded and dewatered headwater portions of SGP watersheds. There are various strategies used by aquatic fauna for recolonizing or repopulating a disturbed reach of stream (Dodds and others, 2004). Spatial strategies include aquatic, terrestrial, and aerial movements. Upstream migrations are the most typical aquatic pathway by which post-disturbance recolonization occurs. Headwater springs, however, often form pools (refugia during drying events) that can serve as population sources from which organisms can migrate (via downstream drift) to lower reaches after the disturbance, a recolonization strategy common to a diversity of macroinvertebrate taxa. Species that can move both upstream and downstream are generally highly mobile, such as fish or crayfish. Aerial dispersal of invertebrates is another common way in which recolonization takes place, and it can occur in either direction along a stream corridor and laterally to adjacent corridors. Some organisms use temporal pathways for repopulating a disturbed reach, such as entering diapause during periods of drought. This diverse array of dispersal and recolonization techniques allows for rapid reestablishment of ecosystem structure and function following disturbance events that are common in SGP streams and rivers.

Whereas mobile invertebrates have been shown to return within a week after intense flooding (Fritz, 1997), the overall post-disturbance composition of invertebrate communities varies with an index of stream harshness composed of flooding intensity, drying occurrence (frequency and duration), and distance from a colonization source. In general, abundance and richness of macroinvertebrates decreased with increasing indices of harshness (Dodds and others, 2004). This work suggests that preserving important refugia throughout SGP watersheds is important for conserving locally adapted macroinvertebrate communities of these systems.

Fishes can be highly mobile, with distances moved ranging from meters to kilometers within SGP streams and rivers (Dodds and others, 2004). Fish responses to disturbances in SGP streams and rivers depend on their abilities to reproduce rapidly (Fausch and Bestgen, 1997; Scheurer and others, 2003) and their proximity to refugia (Larimore and others, 1959). The effect of disturbances on fishes in the SGP is also influenced by time of year. Fish refugia can include both side channel habitats during flood events (Adams and others, 2000) and isolated spring-fed pools during drying events (Labbe and Fausch, 2000; Falke and others, 2011). Fishes of the SGP are adapted to survive and thrive under the harsh conditions typical of these systems, but they may be severely affected by a combination of normal disturbances and anthropogenic CAs, such as channel incision or groundwater pumping, that can eliminate important refuge habitat. The guild of pelagic-spawning, large-river minnow species, which is unique among the guilds of SGP fishes, is particularly sensitive to altered flows and habitat fragmentation (Perkin and Gido, 2011, 2012).

Recolonization events are not limited to fauna. Both microbial communities, which are crucial to nutrient cycling processes in these systems, and primary producers, such as algae, also reestablish quickly after disturbance events (Dodds and others, 1996). Previous experimental work indicated that algae affected by periods of drought can recover to pre-disturbance levels of biomass 2–3 weeks after normal streamflow resumes (Dodds and others, 1996), and in some instances post-disturbance primary production was greater than pre-disturbance levels. Similar recovery trends also have been observed among microbial communities (Fisher and others, 1982).

Associated Species of Management Concern

The large area encompassed by the SGP ecoregion, the pervasive threats to naturally flowing waters throughout this region, and the highly specialized nature of native freshwater biota in the SGP have resulted in many State and Federal listings of SGP species at various levels of imperilment. Of particular concern in the SGP is the Great Plains guild of pelagic-spawning fishes: Arkansas River shiner (*Notropis girardi*) (Chapter 11), plains minnow (*Hybognathus placitus*), peppered chub (*Macrhybopsis tetranema*), Rio Grande silvery minnow (*Hybognathus amarus*), and silver chub (*Macrhybopsis storeriana*). Most of these species have been extirpated from approximately 90 percent of their historical habitat and are found only in isolated and fragmented portions of their native drainages (Luttrell and others, 1999). Because the major threat to this group of fishes is fragmentation (Perkin and Gido, 2011), there is a need for restoration and conservation efforts that focus on creating or maintaining large, unimpeded reaches of streams and rivers.

Three other fish species found within the SGP are listed under the Federal Endangered Species Act: Topeka shiner (*Notropis topeka*), Pecos bluntnose shiner (*Notropis simus pecosensis*), and Pecos gambusia (*Gambusia nobilis*); the Arkansas darter (*Etheostoma cragini*) is a candidate for listing. There are many other inhabitants of SGP streams and rivers that are not federally protected but are listed as species of concern by State agencies, including fish and mussels. Of species that occur in the SGP, Texas lists 4 mussel species of concern, and Colorado, New Mexico, and Kansas list 11, 8, and 6 (respectively) fish species of conservation concern. Oklahoma, on the other hand, lists no aquatic species of conservation concern that occur within the SGP ecoregion.

Change Agents

Streams and rivers of the SGP have been severely affected by multiple stressors that have profoundly changed the ways in which these systems function. Dominant existing CAs include agricultural development and associated activities, although energy development and urbanization contribute to the degradation and fragmentation of streams and rivers. In particular, urban centers have led to the channelization of many SGP rivers and streams, which decreases habitat heterogeneity and severs important connections with floodplains. Projected average temperatures and precipitation patterns associated with climate change also represent significant potential agents of change to streams and rivers.

Development

Energy and Infrastructure

There are many energy resource extraction projects occurring within the SGP, particularly projects associated with several shale gas/oil formations. These formations include the Barnett play in north-central Texas, the Niobrara play in eastern Colorado, and the Woodford play in central Oklahoma (U.S. Energy Information Administration, 2011). Hydraulic fracturing, which can have potentially negative effects on streams and rivers, is the primary type of energy development in these areas. Primary threats to aquatic systems from hydraulic fracturing include groundwater withdrawals and discharges of water with high levels of total dissolved solids.

Agricultural Activities

Because the SGP is an arid landscape that nonetheless supports landscape-scale agricultural production, irrigation demand is high. To irrigate croplands, surface waters are diverted and groundwater is pumped to the surface, fundamentally altering the flow regimes of SGP streams and rivers. Diverting surface water can reduce streamflow within stream channels, diminish aquatic habitat volume, and sometimes completely dewater stream channels. Pumping groundwater can have a very large footprint on the landscape because water being extracted miles away from an active stream channel can reduce water levels in the aquifers from which these stream channels receive inflow. This is particularly troublesome for the SGP because the rate of recharge to the underlying aquifer is very slow (Falke and others, 2011). Pumping from this aquifer can influence streamflow in multiple ways, depending on its proximity to the stream channel. If a well is within the alluvial sediment, a cone of depression can be created, which can reverse the flow of groundwater away from stream channels. Pumping farther away from stream channels can decrease the overall elevations of regional aquifers and decrease the amount of groundwater input to streams and rivers. Reducing groundwater input to streams and rivers in an environment as arid and hydrologically variable as the SGP can drastically reduce perennial pool habitats that otherwise could serve as refugia during low-flow events. In addition, dams have been constructed to store water for both irrigation and municipal use and to provide recreational opportunities. The net result of damming streams and rivers in the SGP is a high level of discontinuity in the historically continuous river networks of this region (Costigan and Daniels, 2012).

Agricultural conversion and livestock grazing practices also could be drivers of change in SGP streams and rivers. The conversion of grassland to cropland can involve installing drainage tiles below the land surface and channelizing or straightening river channels (Wiley and others, 1990; Dodds and others, 2004). Drainage tiles can influence the flow regime of streams and rivers by preventing infiltration of precipitation into various soil layers. Channelization of streams and rivers can homogenize the instream habitat because it removes river meanders and their associated side and back channels. Straightening river channels can reduce structure that slows velocity, and without such breaks, biota can be displaced during high-flow events.

While most land in the SGP has been modified through agricultural conversion and urbanization, a large part of the remaining land is managed for cattle grazing. Cattle can trample streambanks, which in turn leads to increased erosion and sedimentation, thus significantly reduced habitat quality in the affected streams and rivers. In addition many cattle grazing operations use groundwater and surface water for creating farm ponds to provide water for cattle and mitigate flooding downstream of these ponds (Dodds and others, 2004). The dam structures associated with these small impoundments can alter flow regimes in small streams, and they can disrupt stream network connectivity by blocking movement pathways or causing stream channels to dry out (Fausch and Bestgen, 1997). Farm ponds in the SGP also serve as source habitats for nonnative, piscivorous fishes. For example, it is common practice to develop sport fisheries by stocking private farm ponds with species like largemouth bass (*Micropterus salmoides*) (Dodds and others, 2004). These nonnative predators can have deleterious effects on native fish populations, especially species like the Topeka shiner, which is already imperiled (Schrank and others, 2001).

Altered Fire Regime

While fire is recognized as an important structuring force in North American grasslands typical of the SGP, the paucity of trees precludes the use of fire scars for reconstructing natural fire regimes. Studies of lakebed charcoal deposits in the northern Great Plains indicate that fire frequency diminished after European settlement (Ford and McPherson, 1996). It is likely that direct fire-suppression activities and landscapes altered by livestock grazing practices have suppressed the natural fire regime. The influences that altered fire regimes may have on SGP streams and rivers are relatively unstudied, but changes in postfire hydrologic transport of sediment and nutrients could affect overall input of these components in SGP aquatic systems.

Invasive Species

Although invasive species are not currently threatening SGP prairie streams and rivers, per se, there are invasives threatening their biotic communities. The greatest threat comes from nonnative fishes escaping farm ponds and other impoundments and directly entering streams or waters adjacent to stream channels. Schrank and others (2001) found a positive relationship between the number of small impoundments within a watershed and local extirpation of the Topeka shiner, a federally endangered fish. The expansion of nonnative species, from impoundments of all sizes, homogenizes fish communities in stream and river habitats (Rahel, 2000; Falke and Gido, 2006b). Expansion of these nonnative fish also depends on the availability of lentic habitat types (for example, deep pools) within the adjacent streams, as these nonnative species require lentic habitat types to persist and displace native species (Falke and Gido, 2006a). Indeed, the structure of fish assemblages upstream of reservoirs is influenced by the geomorphology of the inlet streams, including stream size, elevational gradient, reservoir connectivity, and catchment area (Falk and Gido 2006a). Overall, it appears that fish assemblages homogenize in streams adjacent to reservoirs, regardless of proximity, although some native fish diversity can be maintained even when these nonnative species are present (Falke and Gido, 2006 a, b).

Climate Change

Changing climatic conditions are likely to influence the precipitation patterns and thermal regimes in the SGP, which could have effects on the functioning of prairie streams and rivers and the population dynamics of biotic communities. Projected changes threaten to intensify the already naturally harsh and variable hydroclimate of the SGP with more drought and greater thunderstorm intensity. Winter air temperatures for the entire Great Plains region have increased during the past 30 years (Karl and others, 2009). Spring precipitation also is projected to decrease, especially in the western SGP (Karl and others, 2009). In turn, these changes are likely to alter the hydrologic regimes of SGP streams and rivers and the recharge rates of important aquifers that influence the streams and rivers (Perkin and others, 2010). An overall effect of more extreme drought (Wuebbles and others, 2013) could be increasing intermittency in streamflow; in turn, more channel drying could result in loss of SGP stream network connectivity and a reduced likelihood of recolonization in certain watershed areas (Fausch and Bestgen, 1997; Falke and others, 2011). If storm events become more extreme, as projected (Wuebbles and others, 2013), precipitation amounts also may increase, and in urban areas with large amounts of impervious surface area, this could result in larger amounts of pollution runoff.

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting streams and rivers is illustrated in figure 15. Ecological attributes and CAs identified in the Rapid Ecoregional Assessment are enumerated in tables 17 and 18.

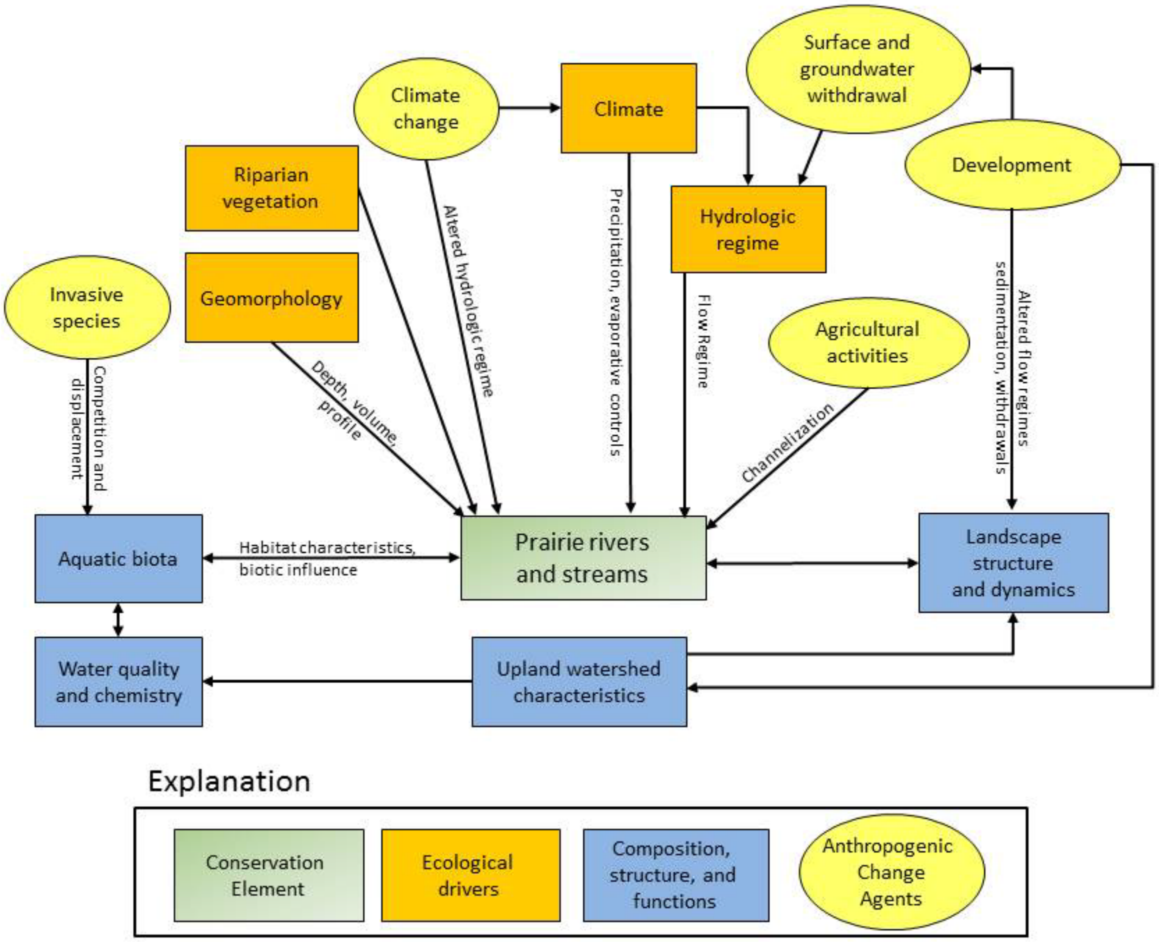

Figure 15. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for rivers and streams in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of rivers and streams are shown in orange rectangles (see also table 17); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 18).

Table 17. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for rivers and streams.

Attribute	Variables
Amount and distribution	Mapped distribution of rivers and streams.
Landscape structure	Size and spatial distribution (length and density).
Landscape dynamics	Hydrologic regime (amount, timing, temporal variability of groundwater and surface inputs), flow regime.
Associated species management of concern	Great Plains pelagic fishes spawning guild (Arkansas River shiner, plains minnow, peppered chub, Rio Grande silvery minnow, silver chub), Topeka shiner, Pecos bluntnose shiner, *Pecos gamusia*, and freshwater mussel species.

Table 18. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for rivers and streams.

Attribute	Variables
Development (energy and infrastructure)	Alteration (groundwater depletion, contamination from upland disturbance, including sediments and toxins), discharge of produced waters.
Development (agricultural activities)	Alteration (sediment inputs from cropland, herbicide runoff, reduction in groundwater input, diversion of surface water, channelization, alteration of flow regime, grazing effects).
Altered fire regime	Alteration from increased intensity and (or) frequency (increased sediment and nutrient inputs, increased runoff velocity, higher rate of bank erosion).
Invasive species	Introduced sport species (displacement of native species).
Climate change	Changes to temperature and precipitation could alter function.

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Chapter 10. Freshwater Mussels Species Assemblage

By Brent N. Tweedy

Key Ecological Attributes

Distribution and Ecology

Freshwater mussels of the family Unionidae (hereafter mussels), by far the most abundant and diverse group of native mussels in North America, are represented by more 300 native species. Much of this diversity is centered in the eastern United States, but species do occur in the western portion of the country. Species from both the Interior Basin and the Texas faunal groups occur in the SGP ecoregion. The Texas faunal species are limited to Texas, whereas most Interior Basin species occur across a much broader geographical range outside the SGP. Of the 30–40 species that occur in the SGP, four species have been selected to represent mussels in the SGP because they are the widest ranging species within the region and because they represent species that occur in a variety of habitats (from fast-flowing streams to ponds and lakes). Moreover, there is very little overlap between the SGP ecoregion and the ranges of most the other species. Because the species within the Texas and Interior Basin faunal groups occur in very different (geographically and ecologically) regions, the two groups may respond differently to a given event or policy. For example, if a severe hurricane were to decimate mussel habitats in eastern Texas, the overall effect on the Texas faunal group could be significant, whereas the effect on the Interior Basin faunal group likely would be negligible. On the other hand, Interior Basin mussel populations in Texas could be affected by events and policies occurring outside of Texas.

Texas Faunal Group

One species from the Texas faunal group, the Tampico pearlymussel (*Cyrtonaias tampicoensis*), was selected to represent mussels within the SGP. This species occurs throughout Texas, extending into the western portion of the state and the boundaries of the SGP. It is native to the Rio Grande, Nueces, Frio, San Antonio, Guadalupe, and Brazos River systems. It is likely that this species also has been introduced to the Trinity and Red Rivers. It can be found in both slow- and fast-flowing waters (Howells and others, 1996).

Interior Basin Faunal Group

The Interior Basin faunal group occurs throughout most of the Mississippi River watershed, including the Missouri, Arkansas, and Ohio River watersheds but excluding the Cumberlandian and Mobile Basins (in Tennessee, Mississippi, and to a lesser extent in nearby states), the Ozarkian Basin (in four corners area of Kansas, Missouri, Arkansas, and Oklahoma), and the Sabine Basin (in northeastern Texas and southwestern Louisiana). Three species from the Interior Basin faunal group were selected to represent mussels within the SGP: the giant floater (*Pyganodon grandis* [formerly *Anodonta grandis*]), the pondhorn (*Uniomerus tetralasmus*), and the yellow sandshell (*Lampsilis teres*). The giant floater occurs throughout Texas, Oklahoma, Kansas, and Colorado within the boundaries of the SGP (Howells and others, 1996; Angelo and others, 2009). It is found on a variety of substrates and can even live in areas of deep, soft mud where other species might sink and die (Coker and others, 1922). It is typically found in slow moving or standing waters (Murray and Leonard, 1962; Howells and others, 1996). It is one of three mussel species still known to occur in Colorado (Harrold and Guralnick, 2010).

The pondhorn occurs throughout Texas, Oklahoma, Kansas, and Colorado within the boundaries of the SGP (Howells and others, 1996; Angelo and others, 2009). It is often found in muddy-bottomed lakes, although it also occurs in some rivers and streams. It is very resistant to drought and is often found in semipermanent waterbodies (Murray and Leonard, 1962; Howells and others, 1996). Although it is still known to occur in Colorado, it has not been reported there since 2001 (Harrold and Guralnick, 2010).

The yellow sandshell occurs in Texas, Oklahoma, and Kansas within the boundaries of the SGP (Howells and others, 1996; Angelo and others, 2009). It is found in flowing waters on substrates ranging from rock to mud. It is not tolerant of drought or drying (Murray and Leonard, 1962; Howells and others, 1996).

Mussels in New Mexico

In New Mexico, there are no remaining mussel populations within the boundaries of the SGP, although the Texas hornshell (*Popenaias popeii*) was once native to the area (Carman, 2007). The yellow sandshell was once native to the Rio Grande, which extends into the SGP in New Mexico (Coker, 1921), and the historical native range of the pondhorn also included New Mexico (Cummings and Cordeiro, 2012).

Mussel Ecology

Historically, mussels were very abundant in North American streams and lakes, but overharvesting for the production of buttons from mussel shells led to dramatic declines in mussel populations (Strayer, 2008; Haag, 2012). Although harvesting pressures are no longer a substantial threat to most mussel populations, other anthropogenic factors have continued the declining trend (Haag, 2012). Today, freshwater mussels are the single most threatened faunal group in North America, with 72 percent of species being listed as threatened or endangered (Haag, 2012).

Many mussels are benthic-dwelling filter feeders and can become quite large (from a few to 20 cm) and are long lived (60–100 years) (Haag, 2012), although some species within the SGP tend to have shorter, faster life cycles. They occur in dense, species-rich aggregations called mussel beds, where they can dominate the benthic biomass (Spooner and Vaughn, 2006). These beds are usually several hundred meters in length and encompass 10–100 individuals per square meter (m^2) (Haag, 2012). Historically, these beds were much larger, as one harvest of a mussel bed in the Mississippi River yielded 100,000,000 mussels from an area of 730,000 m^2, a density of 136.9 individuals/m^2 (Strayer, 2008). Within these beds, mussels can have profound effects on ecosystems. Mussels are filter feeders that remove algae and other food items (bacteria, fungal spores, particulate matter) from the water. They excrete the nutrients from their food into surrounding sediments and water. Through their filter-feeding and excretion activities, they serve as important links between the pelagic (water column) and benthic (sediment, stream/lake bottom) portions of the habitat (fig. 16) (Vaughn and Hakenkamp, 2001). The nutrients they excrete can increase primary productivity (Vanni, 2002; Vaughn and others, 2007) and provide a significant portion of the nitrogen required by the system (Atkinson and others, 2013). These nutrients benefit not only aquatic ecosystems, but also they can fuel surrounding terrestrial systems (Allen and others, 2012).

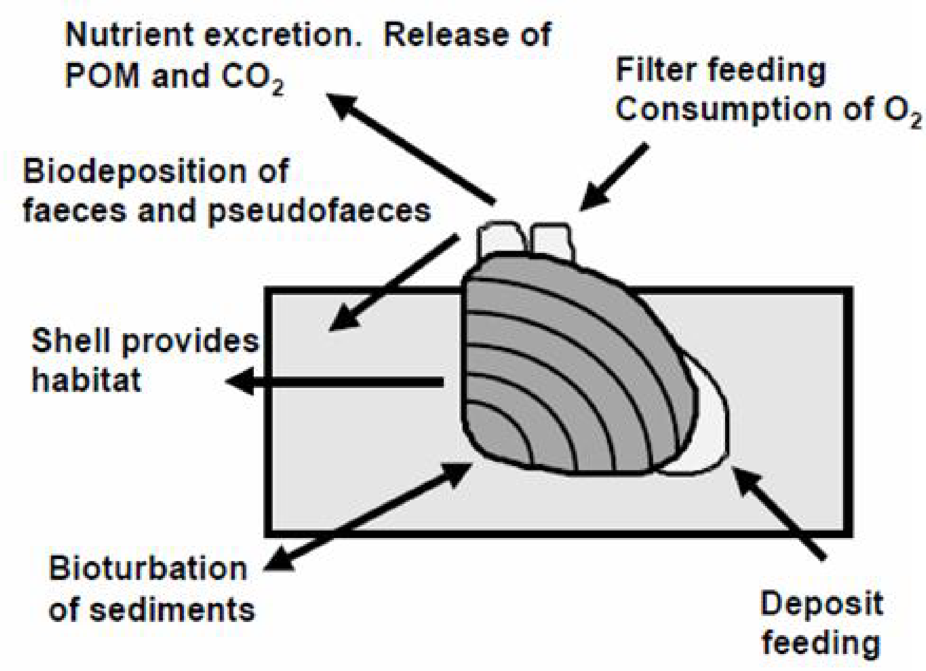

Figure 16. Potential ecosystem function performed by burrowing mussels in freshwater ecosystems (Vaughn and Hakenkamp, 2001). (Illustration courtesy of C.C. Vaughn) (POM, particulate organic matter)

Freshwater mussels use a unique reproductive strategy. Males broadcast sperm directly into the water, which the female mussel filters out of the water to fertilize her eggs. She then broods the larvae on her gills for up to several months until they can attach to the gills or fins of a host fish as an encysted ectoparasite. They remain on the host fish until they mature, at which point they release and drift down to the sediment. Some mussels infect a specific fish host, whereas others are not host-specific. After maturing on the fish, the glochidia release and drift down to the sediment. If they land on suitable sediment, they will burrow into it and then emerge some time later as adult mussels; this phase of a mussel's lifecycle is very poorly understood. During reproduction, there must be adequate water available for broadcasting sperm and for female mussels to have access to fish hosts.

Landscape Structure and Dynamics

There is currently no model that adequately predicts habitat conditions that are likely to support mussel beds. Several factors, however, have been suggested as likely playing a major role in determining where mussels occur (Strayer, 2008). Among the most important are consistent water and flow, sediment stability, and temperature. Flow refugia, such as downstream facets of cobble and boulders, or backwaters, side channels, and oxbows also may be important, especially in rivers with frequent disturbances (Strayer, 1999b).

Mussels require water to live, and because they are unable to move large distances under their own power, suitable habitat in both lentic and lotic systems must be consistently wet. Adult mussels are somewhat tolerant of changes in water-column depth, as long as such changes do not make other conditions such as flow, temperature, food availability, or oxygen unsuitable (Chen and others, 2001; Allen and Vaughn, 2010; Gough and others, 2012). In lotic systems, flow also becomes a factor. Flows that are too low can cause water to become too warm (25–35 °C), resulting in mussel mortality (Allen and others, 2013). Some mussel species are more tolerant of thermal stress than others, but there are no such data for the four species selected to represent mussels in the SGP (Spooner and Vaughn, 2008). Overall, however, flows that are too fast (50 cubic meters per second [m^3/s]) can scour mussel beds and cause sediment instability (Allen and Vaughn, 2010). Additionally, decreased summer temperatures due to hypolimnetic releases (30 versus 25 °C) can lead to increased hermaphroditism and parasitism rates and reduced body condition and population densities of mussels (Galbraith and Vaughn, 2011).

Mussels also require suitable sediments, and different species have different sediment needs. Some can thrive in silty bottoms, whereas others require coarser sediments to burrow into; still others can live on bare rock. No model of sediment properties has been able to predict where beds of a given species will occur (Strayer, 2008). Mussel beds are not uniformly distributed within in a stream; rather they have very patchy distributions, and it is not uncommon for mussel beds to be separated by several km of river (Downes and others, 1993). The role of substrate selection in habitat selection is an ongoing focus of mussel research (Strayer, 2008).

There are no published studies examining the effects of wildfire on freshwater mussels, although there is evidence that forest fires can alter water chemistry (for example, potassium, chloride, sulfate, and nitrate) (Carignan and others, 2000) and zooplankton (Patoine and others, 2000) in boreal lakes. Fires also can affect assemblages of benthic algae by damaging dry biofilms (Cowell and others, 2006) and delaying the recovery of insect communities in intermittent streams as a result of subsequent flooding (Vieira and others, 2004). Although mussels are unlikely to be located in intermittent streams because they dry out regularly, these changes may affect downstream reaches where mussels do occur, although this has yet to be studied. Given the numerous effects of land cover and land practices on aquatic systems at the watershed level, altered fire regimes undoubtedly have effects on mussel populations within the SGP, though more research is needed to understand the specifics of this connection.

Associated Species of Management Concern

Because mussels require fish to serve as hosts for their offspring, their welfare is tied directly to the welfare of host species they utilize. Not only are the fish necessary for the glochidia to mature into juveniles, but they also represent the only phase of a mussel's life cycle when it is truly mobile. Some mussels are generalists and use a wide variety of fish species as their hosts, whereas others are specialists, using only a few or a single species of fish. The known host species for the four species in this paper are listed in table 19. The Arkansas River shiner (*Notropis girardi*) (Chapter 11), however, is not currently known to serve as a host for any mussel species.

Mussels also are important food items for some species. Many mammals prey heavily on mussels, including raccoons, otters, and muskrats. Fish, birds, and turtles also consume mussels, but not much is known about their effects on mussel populations. Some crayfish also consume mussels, but again their effects on populations is unknown (Strayer, 2008).

Table 19. Fish species known to serve as hosts for mussel species in the Southern Great Plains ecoregion.

Species	Known host species	Sources
Cyrtonaias tampicoensis	*Lepisosteus osseus*	(Wootten, 1973; Wiles, 1975; Howells, 1997)
Lampsilis teres	*Lepisosteus osseus*, *Lepisosteus platyrhincus*, *Micropterus salmoides*, *Atractosteus spatula*, *Etheostoma jordani*, *Lepisosteus platostomus*, *Lepomis auritus*, *Lepomis cyanellus*, *Lepomis gulosus*, *Lepomis humilis*, *Lepomis macrochirus*, *Perca flavescens*, *Pomoxis annularis*, *Pomoxis nigromaculatus*, *Rutilus* spp., *Scaphirhynchus platorynchu*	(Lefevre and Curtis, 1912; Surber, 1914; Howard, 1914; Wilson, 1916; Coker and others, 1922; Prentice, 1994; Keller and Ruessler, 1997a)
Pyganodon grandis	*Anguilla* spp., *Apeltes quadracus*, *Gymnocephalus cernuus*, *Oncorhynchus mykiss*, *Perca fluviatilis*, *Pungitius* spp., *Salmo trutta*	(Lefevre and Curtis, 1912; Surber, 1914; Howard, 1914; Wilson, 1916; Coker and others, 1922; Wootten, 1973; Wiles, 1975; Prentice, 1994; Keller and Ruessler, 1997a)
Uniomerus tetralasmus	*Notemigonus crysoleucas*	(Stern and Felder, 1978)

Change Agents

Mussels have biological and behavioral characteristics that make them less able to respond quickly to CAs. They are relatively immobile and cannot easily move to new habitats if their current habitats become unsuitable. Because many species are long lived and have slow and complicated reproduction cycles, the effects of many CAs are often delayed and recovery can be very slow. Additionally, little is known about the juvenile stages of mussels; thus, it is difficult to accurately assess the full extent of many CAs on mussels, and there may be additional CAs of which we are not yet aware.

Development

Energy and Infrastructure

The development of natural gas has resulted in an increase in hydraulic fracturing in the SGP. Hydraulic fracturing can result in flowback (fluids that return to the surface after being pumped underground) of salts and other chemicals in very high concentrations, which can contaminate surface waters and groundwaters if not properly contained and treated (Gregory and others, 2011). Most research, however, has focused on how aquifers may be affected; effects on the health of surface ecosystems have not received much attention (Davis and Robinson, 2012), but we do know that the construction of roads, pipelines, and other surface disturbances place surface waters at risk by mobilizing sediments and chemical contaminants into nearby streams and rivers and altering streamflow (Entrekin and others, 2011). Additionally, aquatic ecosystems are susceptible to mercury contamination caused by emissions from coal-fired power plants (Selin, 2009). Mercury is a potent neurotoxin that negatively affects wildlife health (Scheulhammer and others, 2007), although its effects on mussels have not been studied extensively. Studies that have included effects of mercury have dealt primarily with the glochidia stage and used direct rather than dietary exposure to mercury (Naimo, 1995; Valenti and others, 2005; Faria and others, 2010).

Agricultural Activities

Agriculture and grazing within a watershed can increase the nutrient loading within a watershed (Arbuckle and Downing, 2001). Water demands for irrigation are often greater in watersheds that also support heavy agriculture. This is particularly true in the Great Plains, where irrigation practices can reduce the surface streamflows (Ferguson and Maxwell, 2012) crucial for mussel survival. Soil management practices in watersheds with heavy agriculture can increase erosion, leading to increases in suspended sediment and turbidity, which may have negative effects on mussel populations (Box and Mossa, 1999). For example, increased turbidity can reduce the ability of juvenile mussels to settle and mature (Osterling and others, 2010). Pesticides used in agriculture also can be harmful to mussels (Keller and Ruessler, 1997b; Robillard and others, 2003).

General Anthopogenic Effects

Dam construction and impoundments have a great effect on mussel populations living in lotic systems. Dams affect flow regimes and water temperatures, available habitat, habitat connectivity, and water quality. The majority of rivers and streams in the United States are severely affected by dams and other impoundments (Nilsson and others, 2005). Perhaps the most significant effect of dams and impoundments on downstream mussel populations in streams and rivers is altered streamflow patterns (Magilligan and Nislow, 2005). Many mussel species require flow rates to be within a specific, predictable range for survival and reproduction (Galbraith and Vaughn, 2011). Dams often dampen high flows in the spring and reduce overall flow in the summer and can reduce the heterogeneity of flows, all of which can be important to mussel populations (Magilligan and Nislow, 2005). Reduced flows during summer months can cause drastic increases in water temperature. Dams also drastically fragment mussel habitat, as neither the mussels nor their fish hosts are able to move past them (Nilsson and others, 2005). This likely drastically reduces gene flow between populations and the ability for recolonization of upstream habitat by larger, more stable downstream populations. Dams also alter upstream habitat, especially when they create large reservoirs or lakes. Many mussel species that need flowing water are not able to survive in lentic systems; thus, large portions of formerly suitable habitat can become uninhabitable if impounded. Furthermore, some invasive, non-unionid species, such as zebra mussel (*Dreissena polymorpha*), are more prevalent in lentic bodies than lotic bodies (Benson and others, 2014).

Channelization and bridge construction also can have negative effects on mussel populations. Suitable habitat is often destroyed by the process of channelization and may remain unsuitable for mussel recolonization because of altered flow regime, depth, and sediment stability or quality (Kesler and others, 2001; McGregor and Garner, 2004). Construction of bridges over rivers can destroy mussel habitat, either directly, by destroying mussel beds during construction, or indirectly, through downstream habitat alteration (Fulton and others, 2010).

Domestic and agricultural water use is a concern for conservation of mussel species. Both user groups are consuming water faster than it can be replenished (Baron and others, 2002; Richter and others, 2003). Furthermore, when demand for water is high, as in a drought, water is often held in reservoirs for human use, exacerbating the effects of drought on mussels downstream (Milly and others, 2005). Additionally, the water demands associated with hydraulic fracturing can be quite high and place additional strain on water resources (Nicot and Scanlon, 2012; Murray, 2013). All of these factors reduce the water available for wildlife, such as mussels. If dewatering results in the drying out of riverbeds, mussels are extirpated, and it can take some time for mussels to recolonize the area because of their limited mobility.

Wastewater can be a source of stress to mussels. Water treatment plants often discharge water with high concentrations of nitrogen and phosphorus, leading to the eutrophication of aquatic systems (Carpenter and others, 1998). Eutrophication can cause oxygen depletion (Cornett and Rigler, 1979) and shifts in algal communities (Arbuckle and Downing, 2001), the primary food source of most mussels. Effluent from wastewater treatment is also often high in ammonia, which can be particularly problematic to unionids (Augspurger and others, 2003). Indeed, the glochidia of some species can be negatively affected by unionized ammonia concentrations as low as 0.284 mg/L (0.00000238 ounces per cup [US]) (Goudreau and others, 1993). As a result, the U.S. Environmental Protection Agency's water-quality standards for ammonia have recently been adjusted to account for mussel intolerances of ammonia (U.S. Environmental Protection Agency, 2013). Aquatic ecosystems are also especially sensitive to toxins in runoff, such as heavy metals (mining, industrial waste) (Rainbow, 2002). These toxins can easily accumulate from across a watershed and become concentrated in associated aquatic ecosystems.

Invasive Species

Several invasive species are known to affect freshwater mussel populations. Among the more prevalent are the zebra mussel and Asian clam (*Corbicula fluminea*). These bivalve species are only distantly related to unionid mussels and are dissimilar from the native species in many ways. The zebra mussel was introduced into the Great Lakes in 1988 and since then has spread rapidly and has been found in lakes within the SGP. It affects native species negatively through competition for food and habitat resources, rapid reproduction, and attaching to native species, which can suffocate them (Ricciardi and others, 1998). Zebra mussels are typically found in lakes or large slow-flowing rivers, so they do not threaten mussels that tend to occur in smaller, faster flowing rivers and streams, such as the yellow sandshell (Benson and others, 2014). The Asian clam can be found in lakes, rivers, and streams throughout much of the SGP. It has been very successful in many systems since it was introduced in the United States during the 1950s. These clams are typically very numerous where they do occur, and they tend to infest larger, more permanent bodies of water (Foster and others, 2014). They have been shown to alter community composition and to compete with and negatively affect native mussels (Parker and others, 1998; Ricciardi and others, 1998; Strayer, 1999a; Baker and Levinton, 2003). It also should be noted that species introductions are occurring with increasing frequency, and it is likely that more invasive species could threaten native mussels in the future. Their slow reproduction, immobility, and need for fish hosts could make them more sensitive to and less likely to recover from new invasions.

Disease

Very little is known about the role that disease plays in affecting mussel populations, though it undoubtedly plays a role. Various parasites are known to infest mussels, often at very high rates, as up to 70–100 percent of studied populations were infested (Saarinen and Taskinen, 2004; Taskinen and Saarinen, 2006). Nonetheless, population, community, and ecological ramifications of these and other diseases remain unknown.

Climate Change

Mussels are likely to be heavily affected by climate change, which will exacerbate the existing stresses on mussels from anthropogenic dewatering of aquatic habitats. Projected increases in water temperature due to reduced or no flow will severely affect some species, especially extremes of 25–35 °C and greater. Recent work has shown that some mussels have a higher tolerance to increased temperatures than others, which influences community composition at elevated temperatures (Spooner and Vaughn, 2012). Climate change also is likely to increase the frequency and severity of drought (Intergovernmental Panel on Climate Change, 2007) within the SGP, resulting in low-flow or no-flow conditions (Milly and others, 2005).

Fish Hosts and Change Agents

The CAs discussed herein are only those that have direct effects on mussels. It is important to keep in mind, however, that CAs likely to have negative effects on populations of their fish hosts could be just as devastating to mussel populations. Host specialists may be more affected than host generalists. Known host species occurring within the SGP, especially those of host specialists such as the Tampico pearlymussel and the pondhorn, should be considered in any conservation plan.

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting the freshwater mussel species assemblage is illustrated in figure 17. Key ecological attributes and CAs identified in the Rapid Ecoregional Assessment are enumerated in tables 20 and 21.

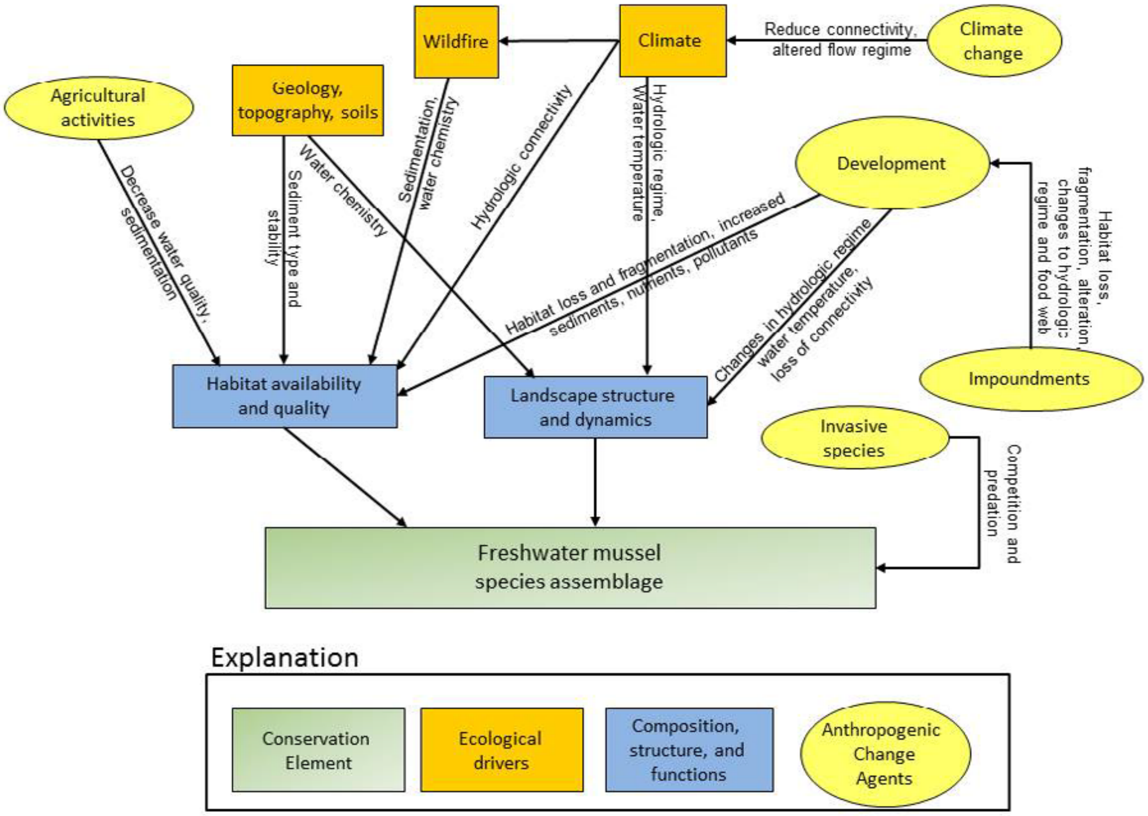

Figure 17. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for the freshwater mussel species assemblage in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of the populations and habtiats of the freshwater mussel species assemblage are shown in orange rectangles (see also table 20); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 21).

Table 20. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for the freshwater mussel species assemblage.

Attribute	Variables
Amount and distribution	Location of mussel beds (presence/absence at the watershed level).
Landscape structure	Size and spatial distribution of suitable habitat (hydrologic connectivity).
Landscape dynamics	Hydrologic connectivity (seasonal fluctuations), flow regime, sediment stability, water temperature.
Associated species management of concern	Multiple fish species serve as hosts (see table 19).

Table 21. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for the freshwater mussel species assemblage.

Attribute	Variables
Development (energy and infrastructure)	Habitat alteration (pollutants from produced waters, increased sedimentation).
Development (agricultural activities)	Habitat alteration (nutrient loading, increased turbidity, suspended sediment).
Development (other human effects)	Habitat alteration (impoundments/reservoirs [direct loss of habitat, change in flow regime, increased water temperature], channelization [direct loss of habitat], human water consumption [loss of connectivity], wastewater discharge [decrease in oxygen, shifts in algal community (food)]), habitat fragmentation (isolation, reduction in gene flow).
Invasive species	Habitat alteration (increased competition for food and habitat resources), predation.
Climate change	Drought effects on habitat condition (connectivity [changes in flow conditions], increase in water temperature), mussel assemblage species composition (temperature threshold varies by species).

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Chapter 11. Arkansas River Shiner

By James J. Roberts

Key Ecological Attributes

Distribution and Ecology

The Arkansas River shiner (*Notropis girardi*) (ARS) is a small-bodied member of the minnow family (Cyprinidae) native to the upper Arkansas River basin, which spans five states (Arkansas, Kansas, New Mexico, Oklahoma, and Texas) in the SGP ecoregion (fig. 18). Currently the ARS is found in only 20 percent of its original habitat (U.S. Fish and Wildlife Service, 1998). The most robust remaining ARS populations are primarily in the Canadian River (New Mexico, Oklahoma, and Texas) and bounded by river impoundments. In addition, small ARS populations are found in Kansas waters of the Cimarron River (Wilde, 2002). These remaining native ARS populations are found only in a combined stream total of 880 km (Larson, 1991; Pigg, 1991). One self-sustaining, wild population occurs outside the species' native range in the Pecos River (New Mexico) (Bestgen and others, 1989; Hoagstrom and Brooks, 2005), likely the result of anglers releasing baitfish (Bestgen and others, 1989; Osborne and others, 2013). Large reductions in the distribution of ARS populations are a major reason for this species being listing as threatened under the Endangered Species Act (U.S. Fish and Wildlife Service, 1998). Genetics of the non-indigenous population in the Pecos River appear comparable to the genetics of native populations (Osborne and others, 2013); thus, this population could help serve as a source or brood stock for future conservation activities and reintroduction plans.

The total length of ARS rarely exceeds 65 mm. The species' life span was 3 years in laboratory settings, but wild individuals rarely exceeded the age of 2 years (Wilde, 2002). The ARS is a broadcast spawner, meaning the eggs and milt (gametes) are released into the water column over a specific area but are not generally associated with a particular type of substrate or rock size and type. The species also spawns fractionally, whereby individuals release gametes multiple times over the course of the breeding season (spring and summer), which may help to avoid total seasonal spawning loss to a single event. Individuals reach maturity at 1 year of age, and because the ARS is a fractional and broadcast spawner, individuals can reproduce multiple times between May and August (Bonner and Wilde, 2000). The ARS eggs are nonadhesive and semibuoyant, characteristics that allow them to remain in the water column and be distributed downstream by river currents. To reproduce successfully, the ARS requires long, unfragmented rivers with periods of high flow because their fertilized eggs hatch within 2 days and larvae require another 1–2 days (that is, 3–4 days posthatch) until they can swim out of the main current and into low-flow areas (Moore, 1944). An approximate river length of 217 km or greater is required for these reproductive processes to effectively occur and sustain ARS populations (Perkin and Gido, 2011); however, simply ensuring the minimal amount of stream length for successful reproduction does not ensure high-quality ARS habitat. For example, the probability of ARS presence is greatest in fragments of 375–780 km (Worthington and others, 2014).

The ARS is found in main channels of braided, shallow (15–25 cm deep), wide rivers with sandy bottoms and slow currents (25–40 m^3/s) (Polivka, 1999; Bonner and Wilde, 2000). The species seems to tolerate warm temperatures, and the probability of ARS occurrence

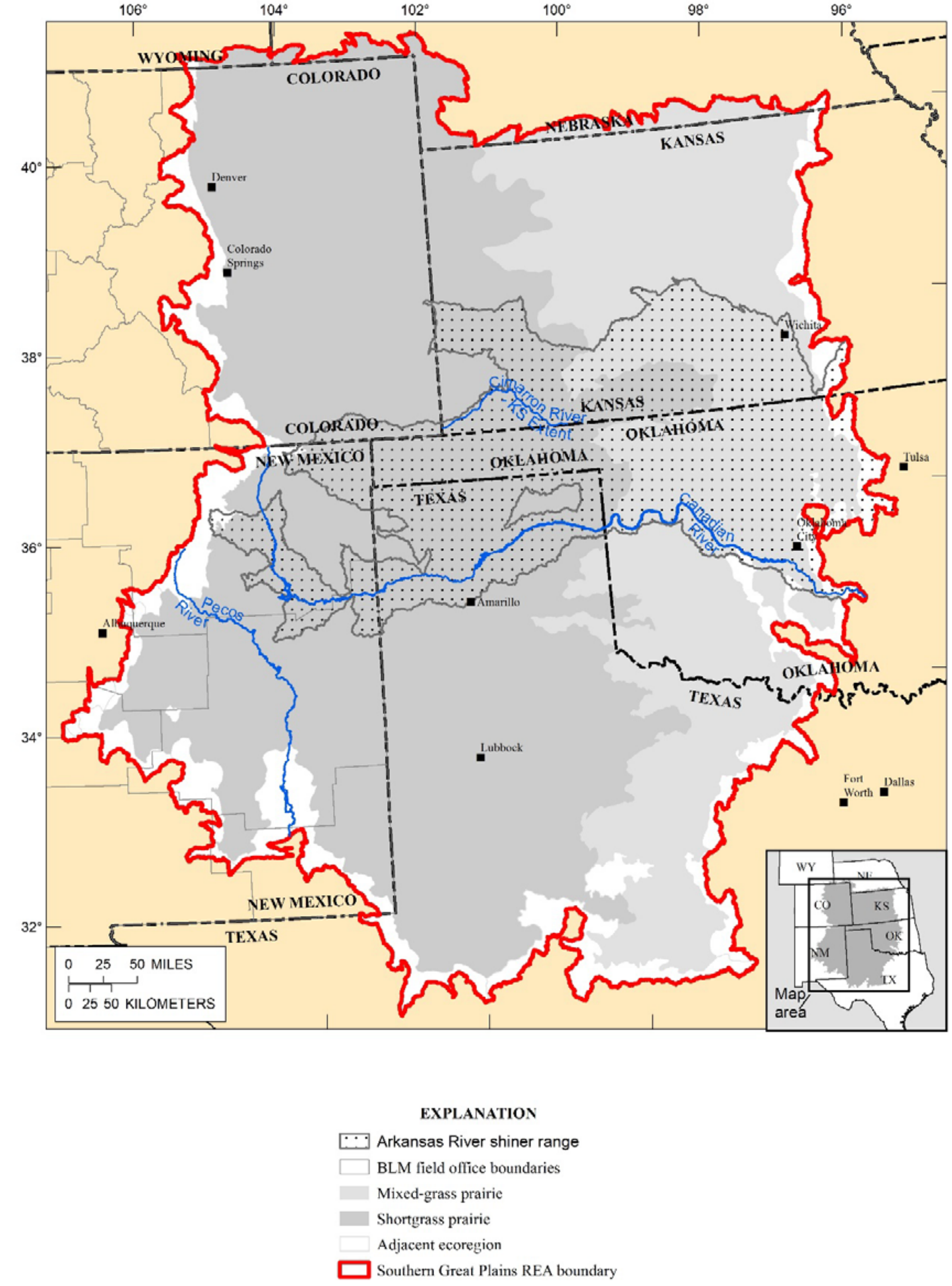

Figure 18. Approximate range of Arkansas River shiner (*Notropis girardi*) in the Southern Great Plains ecoregion (data source: U.S. Geological Survey nonindigenous aquatic species database). (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (BLM, Bureau of Land Management; REA, Rapid Ecoregional Assessment)

increased as temperature increased (until air temperature reached 23 °C) and remained constant at warmer air temperatures (Worthington and others, 2014). The ARS is an opportunistic forager, with most prey items being invertebrates (both terrestrial and aquatic), but plants, algae, and detritus are consumed as well (Wilde and others, 2001). The aquatic invertebrates commonly consumed include larval stoneflies (Plecoptera), caddisflies (Trichoptera), and damselflies (Odonata). Terrestrial invertebrates consumed were primarily beetles (Coleoptera) and true flies (Diptera). Specifically, the ARS appears to forage within benthic substrate and is found to have more prey biomass in the digestive tract during the spring and summer than in fall and winter (Wilde, 2001). The turbid conditions typically found in streams inhabited by the ARS are reflected by the presence of detritus, sand/silt, and plant materials in its diet, likely as bycatch when insects are consumed. Internal (neural structure) and external (sensory laden barbels) morphological adaptations are common in small-bodied minnows of Great Plains streams and rivers, including the ARS (Huber and Rhylander, 1992), to allow for location of prey in such turbid environments.

Landscape Structure and Dynamics

Under the Endangered Species Act listing for the ARS, the U.S. Fish and Wildlife Service has identified two critical habitats, which include sections of the Canadian and Cimarron Rivers (U.S. Fish and Wildlife Service, 2005). Within these habitats, the length of river fragments inhabited by declining (n = 4) and stable (n = 1) populations is 240 (mean) and 462 km, respectively, whereas the mean length of river fragments in which recently extirpated populations (n = 15) had been found was 160 km (Perkin and Gido, 2011). The ARS requires long, unimpeded river fragments primarily because it is a pelagic spawner (that is, they require longitudinal and lateral connectivity for proper in-suspension egg development, downstream dispersal, and movement of newly hatched larvae into side- and backwater habitats) (Wilde, 2002). Where pools have become isolated via low river flows, there are fewer successful reproduction events (Durham and Wilde, 2006).

Drought and withdrawals of groundwater and surface water could hinder successful reproduction events for members of the pelagic-spawning fish guild (that is, a group of fish species that produce similar, semibuoyant eggs that require suspension and distribution via river flows for successful reproduction). Models for predicting presence of ARS populations indicated that any type of channel discharge is more likely to support ARS than isolated pools (or no flow at all), suggesting that there is a threshold streamflow below which the ARS will not occur (Durham and Wilde, 2006). In other words, the magnitude of streamflow seems to be less important to the ARS than the presence of flowing water in river channels. Therefore, prolonged periods of drought could create perilous conditions for the ARS if streamflows are significantly reduced.

Associated Species of Management Concern

The ARS is a member of the Great Plains fishes pelagic spawning guild, which includes two other species of conservation concern, the plains minnow (*Hybognathus placitus*) and the peppered chub (*Macrhybopsis tetranema*), that occur within the ARS' native range. The peppered chub, which is more similar to the ARS than the plains minnow, is currently extirpated from 90 percent of its historical habitat. Specifically, the chub is only found in two disparate basins within the Arkansas River drainage (Luttrell and others, 1999). Diet data for the peppered chub suggest that its foraging ecology is very similar to that of the ARS. Peppered chub diets were approximately equal parts terrestrial and aquatic insects (Wilde and others, 2001), and prey biomass in their stomachs was greatest in June. The most common prey items were Diptera, which made up 29 percent of peppered chub diets on average (Wilde and others, 2001). Also like the ARS, peppered chub digestive tracts contained large proportions of detritus, plant material, and sand/silt (Wilde and others, 2001). There have been no similar, detailed analyses of the plains minnow's diet. Diet similarities between the ARS and peppered chub, however, provide insight as to how similar stressors could contribute to declines in multiple species of the pelagic-spawning guild of cyprinids in the SGP.

Habitat fragmentation is also the major threat to the plains minnow and the peppered chub (Perkin and Gido, 2011). A conservation goal for achieving persistence of these pelagic-spawning species is to ensure a minimum length of continuous, free-flowing river fragments: 115 km for the plains minnow and 205 km for the peppered chub (Perkin and Gido, 2011). Because the minimum river fragment required is greatest for the ARS (217 km), conservation goals of maintaining or reestablishing river-fragment length for the ARS also meet the needs of other SGP fishes of conservation concern.

Change Agents

The ARS literature does not address specific effects of energy and infrastructure, introduced insects and disease, or altered fire regime on this species.

Development

Agricultural Activities

The ARS is native to highly variable and dynamic streams, thus they can tolerate extreme physiochemical conditions (Wilde, 2002). In the SGP, however, the Arkansas River has been highly modified by agricultural cultivation, including reservoir and dam construction, groundwater pumping, and stream channelization. More than 50 reservoirs have been constructed on rivers in the Arkansas River drainage during the last century (Limbird, 1993). The resulting impoundments fragment these river systems and decrease the connected river miles available for fish that depend on longitudinally and laterally continuous, fluvial riverine habitats for successful hatching and dispersal of eggs and larvae (Perkin and Gido, 2011). Movements of eggs and larvae from river to reservoir habitats also expose these early life stages to novel abiotic (absence of flow, increased water clarity, and anoxic sediments) and biotic (visual lacustrine predators) conditions, both of which can increase mortality rates of ARS eggs and larvae (Platania and Altenbach, 1998; Dudley and Platania, 2007). In addition to impeding their downstream movements and raising their mortality rates, dams also can impede upstream movements, both spawning migrations and movements related to recolonization (Bonner, 2000). Information specific to the magnitude of these ARS movements is limited, but other members of the same pelagic-spawning guild have been found to move more than 50 km upstream during spawning season (Bestgen and others, 2010).

Drainages inhabited by the ARS are underlain by the Ogallala aquifer, which is used to irrigate some of the most productive agricultural fields in North America. Mining this aquifer, however, has already reduced its levels and threatens the constant flow of water in some streams (in the sense of Falke and others, 2010). Pumping groundwater and the associated reductions in streamflow also threaten the longitudinal connectivity of Great Plains riverscapes (Falke and others, 2010). Changes in the natural flow regime and connectivity not only reduce the amount of streamflow, but they also dampen the flow variability of ARS streams, resulting in population declines of the ARS and other similar plains stream fishes (Bonner and Wilde, 2002; Falke and others, 2010; Perkin and Gido, 2011). Excessive pesticide runoff also is suspected of having negative effects on populations of cyprinid species (Collins and others, 1995), but there have been no studies that confirm these suspicions.

Invasive Species

At least two invasive fish species, the Red River shiner (*Notropis bairdi*) and the red shiner (*Cyprinella lutrensis*), threaten the ARS. The Red River shiner has been implicated as a potential contributing factor in population declines of the ARS (Luttrell and others, 1995). This nonnative species has been documented in the Cimarron River and was probably introduced by anglers who dumped their bait buckets in the river between 1964 and 1972 (Luttrell and others, 1995). Since then, the Red River shiner has largely replaced the ARS in the Cimarron River (Felley and Cothran, 1981). This invasion happened as streamflow was reduced in the Cimarron River, which has been described as the main cause of ARS decline (Cross and others, 1983; Larson, 1991), but the additional stress from competition with the Red River shiner (an ecologically similar species) likely contributed to ARS population declines in the Cimarron River. The red shiner, which will prey on juvenile fishes (Gido and others, 1999), is another nonnative cyprinid found in high abundance in ARS habitat. A third nonnative species, the mosquitofish (*Gambusia affinis*), poses a risk to the ARS in several Oklahoma rivers, including the South Canadian River (Pigg and others, 1999). Mosquitofish are known to prey on the eggs and larvae and injure the adults of native fishes in the American Southwest (Minckley and Deacon, 1991), and it stands to reason that they could affect ARS populations in similar ways. Therefore, managers seeking to conserve ARS populations would need to incorporate measures in their conservation plans for precluding introductions of nonnative fish in ARS habitats.

Climate Change

Rivers of the SGP ecoregion are extremely variable and are characterized by harsh physiochemical conditions (Dodds and others, 2004), but the region’s native fauna, including the ARS, can tolerate these conditions (Wilde, 2002; Perkin and Gido, 2011). However, the cumulative influences of stream fragmentation, dewatering, channelization, and nonnative species may overwhelm the tolerance limits of the region’s fauna. If climate change further affects the already highly altered hydrology of SGP streams and rivers, ARS populations will be at even greater risk of additional extirpations. Climate change–driven alterations in average streamflow and timing of peak flow events are especially likely to exacerbate population declines of ARS, although potential mechanisms need further study (Worthington and others, 2014). Although historically the SGP ecoregion was characterized by periods of extreme drought (Dodds and others, 2004), any changes in hydroclimate, including increased severity, extent, and frequency of drought could further alter the flow regimes of prairie streams and rivers. Because ARS presence is negatively influenced when streamflows are diminished enough to disrupt a river’s flow continuity and isolate pools (Durham and Wilde, 2006; Taylor, 2010), climatic conditions that increase the prevalence of no-flow conditions could lead to further extirpation of ARS populations. Rises in stream temperature are less likely to exacerbate ARS declines, however, as probabilities of ARS occurrence increase up and level off at 23 °C (mean air temperature for the wettest quarter of the year) (Worthington and others, 2014).

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting the ARS is illustrated in figure 19. Key ecological attributes and CAs identified in the Rapid Ecoregional Assessment are enumerated in tables 22 and 23.

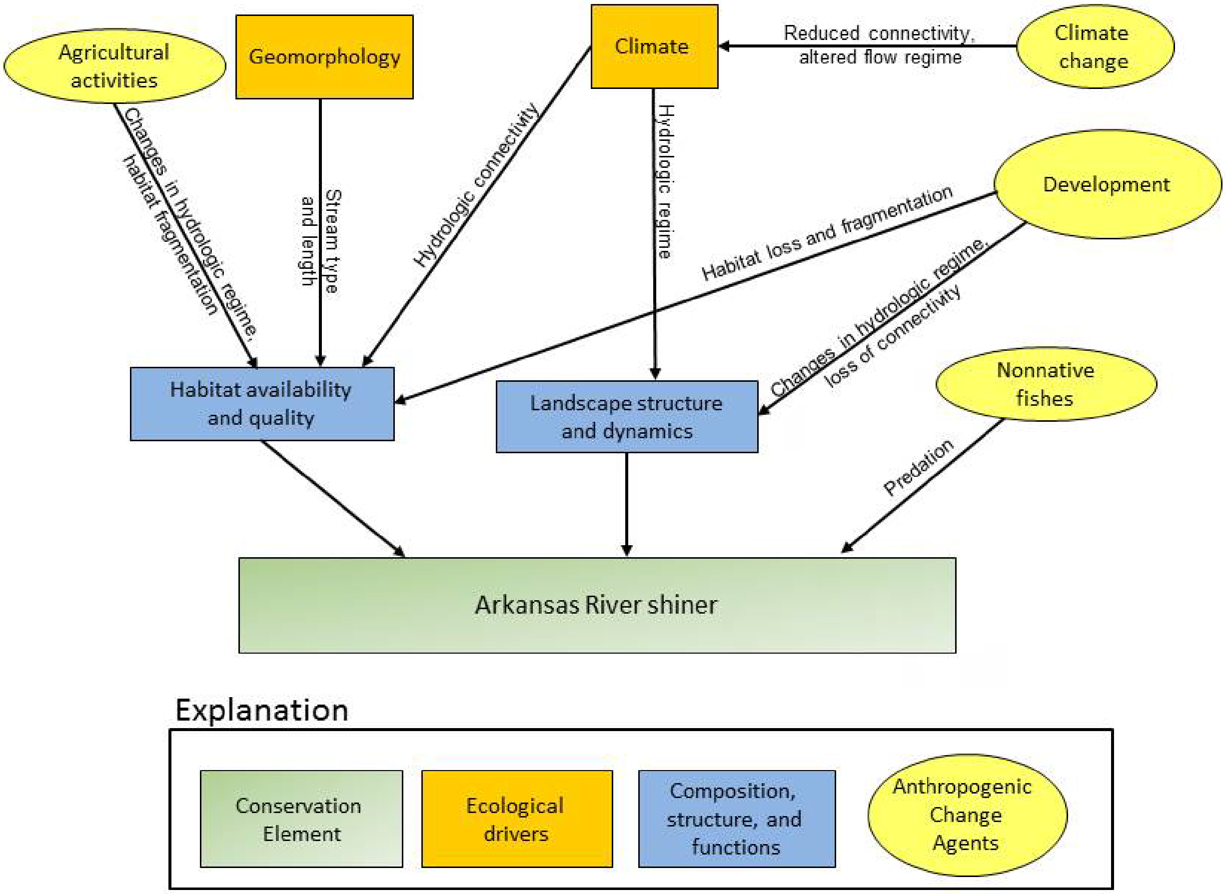

Figure 19. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for the Arkansas River shiner (*Notropis girardi*) in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of Arkansas River shiner populations and habtiats are shown in orange rectangles (see also table 22); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 23).

Table 22. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for the Arkansas River shiner (*Notropis girardi*).

Attribute	Variables
Amount and distribution	Inhabited stream length.
Landscape structure	Length and spatial distribution of inhabited stream length (hydrologic connectivity) and potential unimpeded stream fragments.
Landscape dynamics	Longitudinal and lateral hydrologic connectivity, flow regime, water depth and temperature.
Associated species management of concern	Member of Great Plains pelagic spawning guild, which includes plains minnow and peppered chub.

Table 23. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for the Arkansas River shiner (*Notropis girardi*).

Attribute	Variables
Development (agricultural activities)	Habitat alteration (altered hydrologic regime due to impoundments and reservoirs, direct loss of habitat due to channelization, loss of hydrologic variability from groundwater pumping), habitat fragmentation (reduction in connectivity for adult and reproductive stage).
Invasive species	Habitat alteration (increased competition for food and habitat resources), predation.
Climate change	Drought effects on habitat condition (loss of connectivity due to altered hydrologic regime).

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Chapter 12. Burrowing Owl

By Cynthia P. Melcher

Key Ecological Attributes

Distribution and Ecology

The burrowing owl (*Athene cunicularia*) is a small, diurnal, ground-dwelling raptor of open country. Although there are 15–25 subspecies widely and patchily distributed across the Western Hemisphere (Poulin and others, 2011), herein we address the western burrowing owl (*A.c. hypugaea*; hereafter, burrowing owl), which breeds in mainland North America outside of Florida (however, we cite some studies conducted in Florida when no information on a given topic is available for western burrowing owl). Currently, the burrowing owl breeds in the western half of the United States (including parts of all the Bureau of Land Management Rapid Ecoregional Assessment project areas in the lower 48 states); the extreme southern tiers of the Canadian prairie provinces; and throughout northern and central Mexico (Poulin and others, 2011). In the SGP ecoregion, the burrowing owl is found primarily in the western half of the region (fig. 20), but historically it occurred throughout most of the eastern half as well (Klute and others, 2003). Today, the largest relatively contiguous area of the breeding range lies east of the Rocky Mountains in the shortgrass prairie and the western half of the mixed-grass prairie, from southern Canada to the Big Bend region of Texas; thus, where possible, we further focus on key ecological attributes and CAs for populations east of the Continental Divide.

Northern breeding populations of burrowing owls are considered migratory. From about the Kansas-Oklahoma border southward, however, some birds remain as year-round residents (Poulin and others, 2011), usually males. Migratory owls winter primarily in the southern United States from California to western Louisiana, much of Mexico, and scattered sites southward into Central America. Spring migration occurs from late winter to early spring, with owls returning to their northernmost breeding range in April and early May. Fall migration occurs from late summer through mid-autumn, peaking in early to late September (Klute and others, 2003; Poulin and others, 2011).

The burrowing owl is a semicolonial species that nests in well-drained (that is, where water typically does not accumulate) grasslands and prairies, shrub-steppe, deserts, and various human-altered habitats, including agricultural lands, vacant lots, fairgrounds, and similar open sites (Poulin and others, 2011). In the SGP, primary breeding habitat is shortgrass and mixed-grass prairie (see Chapters 4 and 5) and desert grasslands. Occupied habitat is characterized by very short vegetation, including moderately to heavily grazed sites, a significant component of bare ground, and often elevated perch sites, such as shrubs or fence posts, from which the birds may scan for prey and predators (Panella, 2013). A crucial component of breeding habitat is the presence of underground burrows created by fossorial mammals, especially prairie dog burrows, but also the burrows of ground squirrels, badgers, and other species (Poulin and others, 2011). The owls co-opt the burrows as nest sites, underscoring the importance of drainage and relatively stable soils in their habitats. They also use artificial burrows made for them in areas where owl recolonization is desired but natural burrows are lacking, and they will nest in culverts and other artificial belowground cavities (Poulin and others, 2011). Nonbreeding and migration habitats

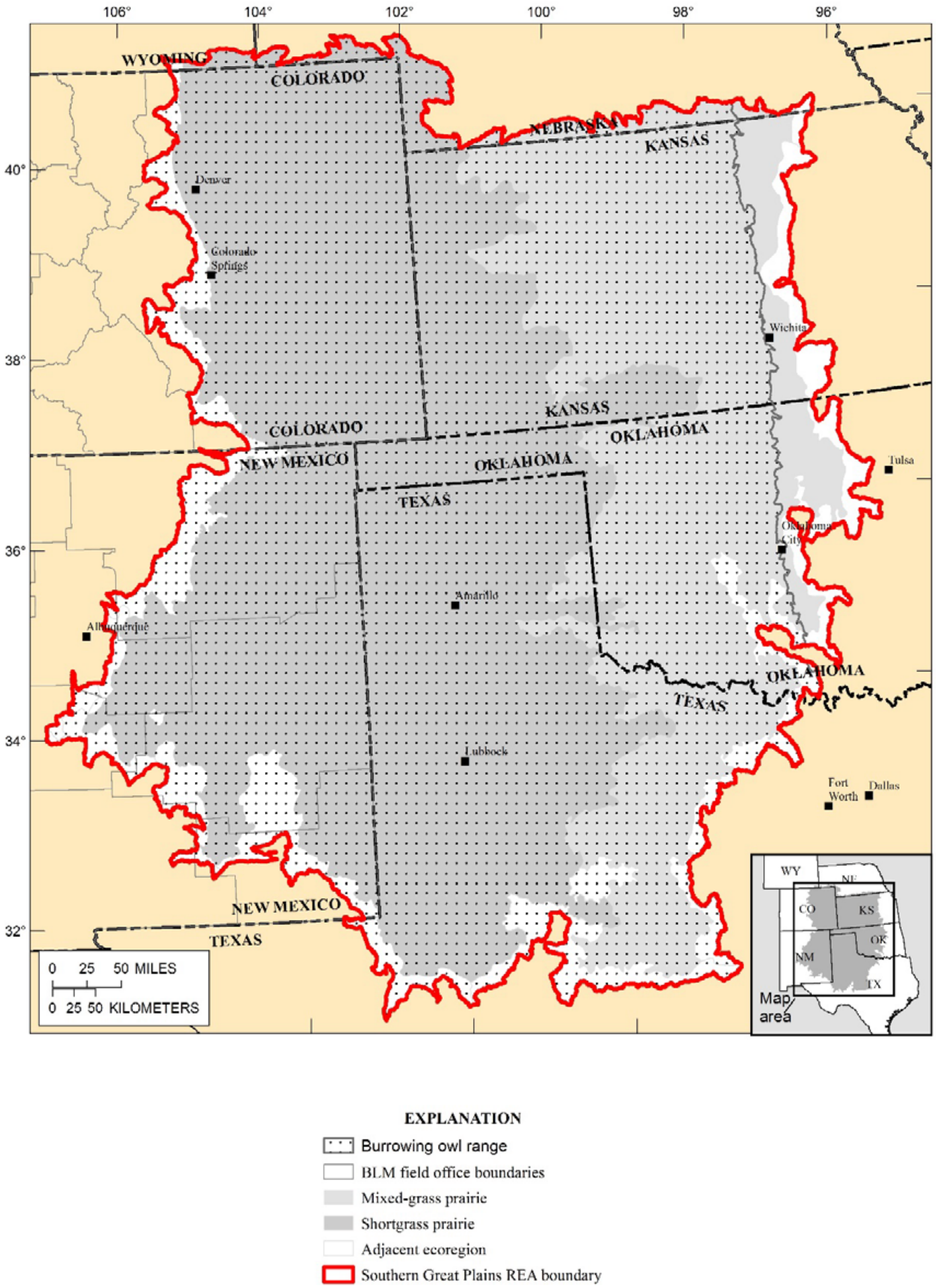

Figure 20. Current range of the western burrowing owl (*Athene cunicularia hypugaea*) in the Southern Great Plains ecoregion (data source: U.S. Geological Survey GAP Program). (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (BLM, Bureau of Land Management; REA, Rapid Ecoregional Assessment)

are similar to breeding habitats, but vegetation may be even more sparse than it is on the breeding grounds (Poulin and others, 2011). Nonbreeding owls often use burrows for roosting as well as nesting, but they also roost in clumps of vegetation. Depending on habitat, the American badger (*Taxidea taxus*) is an important nest predator of burrowing owls. Skunks (*Mephitis* spp.), weasels (*Mustela* spp.), and Virginia opossums (*Didelphis virginiana*) also consume eggs and (or) nestlings; American crows (*Corvus brachyrhynchos*) and a wide variety of raptors prey on prefledged owlets as well as adults (Poulin and others, 2011; Panella, 2013).

Burrowing owls are opportunistic feeders. They consume a wide variety of invertebrates (especially beetles, grasshoppers, crickets, scorpions, and ants), rodents, birds (horned larks [*Eremophila alpestris*] in particular), small herptiles, and crayfish (Cambaridae) (Conrey, 2010; Poulin and others, 2011). The owls forage primarily at dawn and dusk, but they will feed any time as circumstances dictate. Foraging owls typically search for prey while walking, running, or hopping along the ground, but they also hunt from elevated perches or while hovering midair, and they do some fly catching. More invertebrate prey are captured during daylight hours, usually by females, and more vertebrates are captured at dawn and dusk, usually by males (Poulin and others, 2011). Overall, the owls consume more invertebrates, but small mammals comprise most of the dietary biomass (Conrey, 2010; Poulin and others, 2011). Burrowing owls typically forage where vegetation is very short, such as prairie dog towns (Panella, 2013), but their diet also includes species more typical of taller vegetation, such as that found along road rights-of-way (explaining why these owls are often close to or on roads) or even shrub habitat (Conrey, 2010; Poulin and others, 2011). Dietary composition shifts seasonally, with more invertebrates being taken in the breeding season, although this can vary by region, prey dominance, and other factors. Food-supplementation experiments indicate that prey availability may limit egg and clutch sizes and hatching success, which may explain low productivity observed near developed areas where prey habitat has been diminished (Poulin and others, 2011).

Overall, many burrowing owl populations are suspected of declining, and the range has been contracting southward and westward for at least 30 years (Klute and others, 2003; Poulin and others, 2011). Indeed, the species has been extirpated from much of its original range in Canada and eastern portions of the Central Plains (Poulin and others, 2011). The burrowing owl has endangered, threatened, or conservation concern status in many states, and in Mexico and Canada it is listed as threatened and endangered, respectively (Klute and others, 2003; Poulin and others, 2011). Although it is not listed under the Endangered Species Act in the United States, the U.S. Fish and Wildlife Service considers the burrowing owl a species of conservation concern and has developed a status assessment and conservation plan for it (Klute and others, 2003). Important data gaps include a lack of coordinated long-term population monitoring and a poor understanding of mechanisms affecting populations (Poulin and others, 2011). Research is needed to better understand their metapopulation dynamics, particularly immigration/emigration and migration patterns (Klute and others, 2003).

Landscape Structure and Dynamics

Landscapes inhabited by burrowing owls are typically very open, treeless grasslands or shrub-steppe with minimal shrub cover (see Chapters 4 and 5). Nesting areas are characterized by gently sloping or fairly level but well-drained terrain, dry sandy-loam and silty-clay soils, short (if not sparse) vegetation, and often a significant component of bare ground (Poulin and others, 2011; Panella, 2013). In the northern Great Plains, however, greater owl occupancy and reproductive success were explained by habitat-occupancy models that included cropland and crested wheatgrass (Restani and others, 2008). Landscapes of very short vegetation interspersed with patches of taller vegetation may support greater prey densities and alternate prey species (Conrey, 2010), which could help the owls cope with variations in climate and population cycles of prey.

Spatial use of landscapes by burrowing owls occurs on a relatively small scale as compared to that of most raptors. Overall, between-nest distances vary widely from 14 to 900 m, with reported nesting densities varying from 7.1 to 16.4 per ha; this variation is probably due in part to the burrow types (colonial versus noncolonial fossorial animals) used in a given area (Poulin and others, 2011). Prairie dog colonies less than 35 ha in size and areas with greater burrow densities seem to be preferred over larger colonies and areas with lower burrow densities, possibly because clustering of breeding birds increases the chances of early predator detection and greater burrow density provides more escape options from predators (Poulin and others, 2011; Panella, 2013). Recorded home range sizes of adult male owls in Saskatchewan varied from 0.14 to 4.81 km^2, and in Oklahoma young of the year have been found up to 2.4 km from the nest during foraging bouts (Poulin and others, 2011). Dispersal distances from natal sites range widely from less than 0.5 km in Florida to more than 125 km in Saskatchewan, and generally females disperse from natal sites farther than males (Poulin and others, 2011). Burrowing owls exhibit some degree of nest-site fidelity, but the majority switch pastures or colonies in subsequent nesting seasons, usually within 30 km (often much less) of previously used sites.

Herbivory, weather events, climatic variability, and fire help to maintain burrowing owl habitat by setting back the seral stages of vegetation, discouraging encroachment of woody plants, and generating a mosaic of varying vegetation structure. Intensive grazing by ungulates, prairie dogs, and insects is an especially important dynamic for maintaining burrowing owl habitats, cycling nutrients through the ecosystem, and supporting plant diversity (thus prey diversity) across the species' range (Milchunas and others, 1998; Whicker and Detling, 1988; Poulin and others, 2011). Historically, American bison (*Bison bison*), elk (*Cervus elaphus*), deer (*Odocoileus* spp.), and pronghorn (*Antilocapra americana*) grazed across the SGP year-round; today, however, cattle and other livestock have largely replaced these animals (Milchunas and others, 1998). The black-tailed prairie dog (*Cynomys ludovicianus*) is another important aboveground year-round herbivore in the SGP; although they enter a state of shallow torpor when stressed by winter cold and food shortages, they are not obligate hibernators (Harlow and Frank, 2001). Herbivorous invertebrates that have significant effects on aboveground vegetation structure include short-horned grasshoppers (Caelifera), population outbreaks of which can result in denuded landscapes, and ants (Emerick, 1984; Weaver and others, 1996). Population outbreaks of grasshoppers often occur during drought, possibly because diseases that afflict grasshoppers are suppressed under arid conditions (Weaver and others, 1996) and grasshoppers lay their eggs in denuded soil (Knight, 1994). Important belowground herbivores include beetle grubs and nematodes (Rottman and Capinera, 1983; Weaver and others, 1996), as well as pocket gophers (for example, the plains pocket gopher [*Geomys bursarius*]), which not only consume significant amounts of belowground plant matter, but also create patchworks of surface disturbance (Martinsen and others, 1990).

Extreme, short-term weather events (such as tornados or hail) can immediately set back the vegetation seral stages or even remove vegetation, and they can cause direct mortality to owls and their prey. These events, however, are spotty and localized, thus creating patchworks of disturbed sites. Cycles of drought, however, have had far greater influence on the vegetation structure throughout the shortgrass and mixed-grass prairies (Axelrod, 1985). Many grasses in the SGP portion of the burrowing owl's range, including buffalograss and blue grama, are drought tolerant and can remain dormant for long periods until moisture returns; thus, short-term drought may favor these species and the owl's preferred habitat structure. Severe and prolonged drought, however, may alter germination success (Minnick and others, 1999), cause widespread plant mortality, greatly diminish plant recruitment, and result in significant declines in populations of invertebrate and small mammal owl prey. For example, prairie voles (*Microtus ochrogaster*), which are consumed by burrowing owls more than expected relative to their abundance (Conrey, 2010), were virtually wiped out around Hays, Kansas, during the Dust Bowl years and did not recover for several years after the drought (Tomanek and Hulett, 1970). On the other hand, mice are also an important component of burrowing owl diets (Conrey, 2010; Poulin and others, 2011), and the drought did not affect white-footed mouse (*Peromyscus leucopus*) populations (Tomanek and Hulett, 1970). To some extent, burrowing owls may track climate-driven changes in the relative abundances of their prey.

Reconstruction of historical fire regimes in the shortgrass and mixed-grass prairie is difficult because of the lack of woody vegetation and the absence of fire scars. Nonetheless, there is evidence that fire has been an important influence in shaping the structure and dynamics of the shortgrass and mixed-grass prairies (Ford and McPherson, 1997; Brockway and others, 2002; Clark and others, 2002). In a West Texas study, fire did not affect the viability of blue grama and buffalograss (Wright, 1974), two important components of burrowing owl habitat. Fire also recycles nutrients that promote vegetation diversity, which in turn promotes prey diversity. Indeed, some researchers have found that burrowing owl nesting densities were greater in fire breaks and other burned sites than in adjacent unburned sites (Panella, 2013).

Associated Species of Management Concern

Because fossorial mammals, including prairie dogs, badgers, ground squirrels (*Spermophilus* spp.), kangaroo rats (*Dipodomys* spp.), yellow-bellied marmots (*Marmota flaviventris*), foxes (Canidae), and coyotes (*Canis latrans*) create burrows used by nesting burrowing owls, local population declines or extirpations of these mammals are likely to result in declines or local extirpations of burrowing owls (Poulin and others, 2011). In the SGP, burrowing owls associate especially closely with the black-tailed prairie dog (*Cynomys ludovicianus*), which is listed as a Species of Greatest Conservation Need in all SGP states and is treated as a CE in the SGP Rapid Ecoregional Assessment (Chapter 20).

Change Agents

Development

Development of all kinds affects burrowing owls by degrading and fragmenting habitat or eliminating it altogether (Klute and others, 2003; Poulin and others, 2011; Panella, 2013). A major source of habitat fragmentation and owl mortality is the road building that accompanies all forms of development. Burrowing owls often are hit by moving vehicles along roads, which is exacerbated by the species' propensity for foraging along road rights-of-way and resting on gravel roads and their low-flying habits (Poulin and others, 2011; Panella, 2013). With residential and agricultural development, local densities of domestic pets also typically increase, which can cause direct and indirect mortality to burrowing owls (Klute and others, 2003), as dogs often collapse nesting burrows and cats prey on owls. In some cases, prey density has been shown to increase near developments, which in turn has attracted greater densities of burrowing owls, but eventually there is a development threshold beyond which the birds decline regardless of prey density (Poulin and others, 2011). The human activities associated with development disturb burrowing owls and have been found to result in reduced nesting productivity (Poulin and others, 2011), and recreational shooting is believed to cause stress and is known to cause direct mortality (Lantz and others, 2004).

Energy and Infrastructure

In the Great Plains, where energy development is accelerating (oil and gas drilling, wind farms), studies are lacking to evaluate effects of this development on burrowing owls. A study at the Altamont Pass Wind Resource facility in California, however, indicated that from 1998 to 2003 nearly one burrowing owl was killed per kilowatt capacity per year, as determined by owl carcasses found under wind turbines (the data excluded birds apparently killed by electrocution or collision with powerlines; turbines included a wide variety of older and newer types) (Smallwood and Thelander, 2008). The effect of overhead disturbance from wind turbines on owl behavior is unknown.

Infrastructure associated with energy development, roads and distribution systems in particular, create travel corridors used by mammalian predators. Powerlines and other vertical structures associated with energy development also create high perches in otherwise featureless terrain, thus conferring an advantage to aerial predators—crows, ravens, and hawks—that prey on burrowing owls (Klute and others, 2003; Poulin and others, 2011; Panella, 2013). Construction activity can have negative effects on the productivity of nearby owl colonies (Poulin and others, 2011), and although effects of sonic noises created by exploratory “thumper” trucks and drilling in owl habitat is unknown, owls rely heavily on their hearing to locate prey from dusk to dawn, and effects on their ears or hearing could diminish their ability to capture prey or hear warning sounds from other owls that predators have entered the area (Panella, 2013).

Agricultural Activities

Agricultural activities undoubtedly represent the greatest source of decline in burrowing owl populations across the Great Plains. Intensive row-cropping and associated cultivation practices, prairie dog control efforts (to diminish competition for livestock forage), pesticide use, entanglement with barbed wire fencing, and trampling of burrows by livestock all have negative effects on owl populations (Klute and others, 2003; Poulin and others, 2011; Panella, 2013). An estimated 20–99 percent of native prairie in states and provinces of the Great Plains has been converted to agriculture or other development (Samson and Knopf, 1994), easpecially in eastern parts of the region. Cultivation fragments, degrades, and eliminates habitat, in large part because of the elimination of the prairie dogs that create the burrows required by burrowing owls for nesting and roosting.

Landscape context associated with agricultural cultivation varies widely, however. In Canada, where burrowing owl populations are declining rapidly, the vast majority of nesting owls are found on the remaining intact native grassland, not on cultivated lands (Poulin and others, 2011). On the other hand, irrigated farmland in Idaho has been found to support burrowing owls, most likely because of greater prey densities associated with these lands (Poulin and others, 2011). In another study, nesting density, reuse of nesting burrows, and the number of young fledged per nesting attempt were greater in agricultural settings than in urban settings, but rates of recruitment and return of breeding adults were lower in the agricultural lands, indicating that agricultural lands may represent habitat sinks (Conway and others, 2006). Berardelli and others (2010) also found that owl reproductive success was greater in the agricultural settings, but the number of young fledged per successful nest and overall reproductive success was greater in urban settings where nesting densities were lower; in agricultural areas, greater reproductive success was associated with greater nesting density. Other studies have indicated that burrowing owls were more likely to be present (Orth and Kennedy, 2001; Restani and others, 2008) and experience greater reproductive success (Restani and others, 2008) in landscapes characterized by patchworks of shortgrass and agricultural lands. More research is needed across the owl's range to elucidate the mechanisms behind effects of agricultural lands on owl habitat use and its ultimate effects on population dynamics.

Agricultural economics and changes to the Farm Bill recently resulted in significant declines in lands enrolled in the CRP. The CRP has buffered many wildlife and plant species from negative effects of intensive agriculture and soil erosion, but studies of CRP benefits to burrowing owls are lacking. The seed mixes originally used on CRP lands were usually nonnative, cool-season grasses, which provide a vegetation structure not usually suitable for nesting burrowing owls, although there is at least some evidence that burrowing owls may benefit from higher prey densities if their shortgrass nesting habitat is juxtaposed with taller grass foraging habitats (such as wheat fields or CRP) (Poulin and others, 2011). Fallow agricultural lands also attract burrowing owls (U.S. Fish and Wildlife Service, 2014), as their habitats often include bare ground, but whether these habitats serve as source or sink habitats is unclear.

The black-tailed prairie dog has been extirpated from a large proportion of its historic range, including significant portions of the SGP (Proctor and others, 2006), largely a result of cultivation and intentional eradication efforts through shooting, poisoning, and trapping (Poulin and others, 2011). When prairie dogs are eradicated, their burrows eventually fall into disrepair and eventually collapse. In Nebraska, nesting burrowing owls declined after prairie dog control efforts, although the effect was not immediate because it took a few years for the burrows to deteriorate (Desmond and others, 2000). Although burrowing owls are killed by shooters during prairie dog control efforts (Klute and others, 2003; Panella, 2013), it is unclear whether the shooters cannot distinguish them from prairie dogs during prairie dog control efforts or they target the owls intentionally.

Pesticides affect owls directly through poisoning and indirectly through reduced prey populations. Carbofuran, a particularly toxic carbamate used to control insects in a wide variety of crops, diminishes burrowing owl survival and nesting success when the chemical is sprayed over nest burrows (Poulin and others, 2011). Owls that consume rodents poisoned by strychnine-coated grain experience sublethal effects, including lower body weights (Poulin and others, 2011). Burrowing owl eggs also are found to contain levels of DDE (dichlorodiphenyldichloroethylene, a breakdown product of the organochloride pesticide DDT [dichlorodiphenyltrichloroethane]) that cause reproductive failure in other species, although similar effects have not been reported for burrowing owls (Poulin and others, 2011).

Altered Fire Regime

Because it has been difficult to reconstruct historical fire regimes for the Central Plains grasslands, it remains uncertain how altered the fire regime is and how it may have affected burrowing owls. Overall, however, fire is believed to have been an important influence in shaping prairie systems, and fire suppression has allowed the encroachment of woody vegetation (Klute and others, 2003), which would not benefit burrowing owls and could contribute to their decline. Furthermore, there is evidence that owls readily use postburn sites (Panella, 2013). More information is needed, however, on how fire affects burrowing owl habitat occupancy and population dynamics.

Invasive Species

Invasive, exotic plants are not identified as a major source of concern in several burrowing owl assessments (Klute and others, 2003; Lantz and others, 2004; Poulin and others, 2011; Panella, 2013). In some regions, however, encroachment of woody vegetation in grasslands is a concern, particularly mesquite (*Prosopis* spp.), creosote bush (*Larrea tridentata*), and juniper (*Juniperus* spp.) (Klute and others, 2003). There are some reports of cheatgrass growing on burrowing owl nest sites, but generally west of the SGP in shrub-steppe and desert habitats, and it was indicated that owls occupied sites with more cheatgrass cover than was found in unoccupied sites. Because burrowing owls will use burn sites, it is not clear whether increasing fire frequency and size due to cheatgrass is a concern. However, the effects of cheatgrass-fueled fire at a landscape scale and effects of increased fire frequency on burrowing owl prey populations need further inquiry.

Insects and Disease

The most important disease affecting burrowing owls is the indirect effects of sylvatic plague (*Yersinia pestis*). This nonnative pathogen is very lethal to prairie dogs (U.S. Fish and Wildlife Service, 2004, 2009). Epizootics of this disease often kill off entire colonies of prairie dogs, which results in the loss of burrowing owl nesting habitat; after a few years, the unmaintained burrows collapse and the owls also disappear (Klute and others, 2003).

Climate Change

Effects of climate change on burrowing owls could be both direct and indirect. If precipitation events in the breeding range become more intense and slower moving, then flooding and (or) collapse of nest burrows could become a greater source of nesting failure (Poulin and others, 2011). If climate change intensifies and prolongs cycles of drought, associated changes in agricultural economics and changing land uses could have as yet unpredictable effects on burrowing owls. Although some animal species are predicted to expand their ranges northward as the climate warms, “northward” (Canada) for burrowing owls is currently a region of sharp population decline; thus, the potential for northward expansion is dubious. Many factors, including development of genetically altered crops that tolerate drought and economics of water use are not only driving farmers to switch from irrigated to dryland farming, but are also encouraging sod busting (Howell, 2001; Johnson, 2005), especially in the southern prairies of Canada.

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting the western burrowing owl is illustrated in figure 21. Ecological attributes and CAs identified in the Rapid Ecoregional Assessment are enumerated in tables 24 and 25.

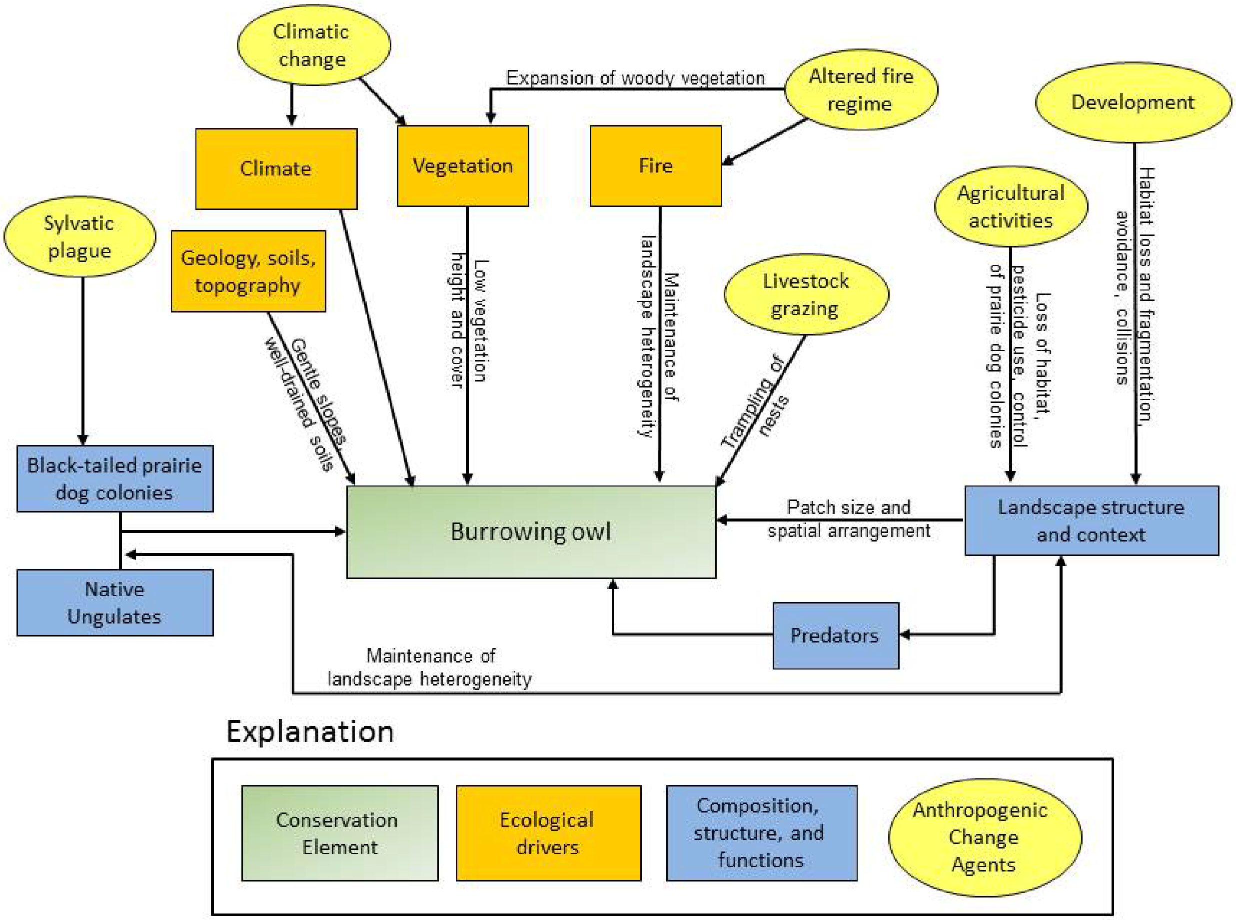

Figure 21. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for the western burrowing owl (*Athene cunicularia hypugaea*) in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of western burrowing owl populations and habtiats are shown in orange rectangles (see also table 24); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 25).

Table 24. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for the western burrowing owl (*Athene cunicularia hypugaea*).

Attribute	Variables
Amount and distribution	Burrowing owl distribution.
Landscape structure	Size and spatial distribution of habitat, nesting density.
Landscape dynamics	Habitat productivity (food resources), climatic variability, predator dynamics.
Associated species management of concern	Black-tailed prairie dog, swift fox, and pronghorn.

Table 25. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for the western burrowing owl (*Athene cunicularia hypugaea*).

Attribute	Variables
Development (energy and infrastructure)	Habitat alteration (direct loss of habitat), habitat fragmentation, avoidance, collisions (infrastructure), predation.
Development (agricultural activities)	Habitat alteration (direct loss of habitat due to cultivation [-], irrigated farmland (increase in prey base) [+]), livestock grazing (trampling of burrows), pesticide use, control efforts to black-tailed prairie dog colonies.
Development (other human effects)	Habitat alteration (direct loss of habitat), habitat fragmentation, collisions (roads), predation by domestic cats.
Altered fire regime	Fire suppression results in habitat alteration (decrease in landscape heterogeneity, expansion of woody vegetation).
Invasive species	Habitat alteration (expansion of invasive woody species).
Introduced species and disease	Presence and effect of plague on black-tailed prairie dog population.
Climate change	Indirect effects from changing land use likely to have largest effect on this species.

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Chapter 13. Ferruginous Hawk

By T. Luke George

Key Ecological Attributes

Distribution and Ecology

The ferruginous hawk (*Buteo regalis*) is the largest hawk in North America and the only buteo that is endemic to shrub-steppe grasslands. Ferruginous hawks breed from southern Canada to northern New Mexico and Arizona and from eastern Oregon, Washington, and northeastern California to western North and South Dakota and Nebraska (Bechard and Schmutz, 1995), encompassing almost the entire SGP ecoregion (fig. 22). Their wintering distribution stretches from northeast Utah, extreme southern Wyoming, and southwestern Nebraska south to western Texas, central Mexico, and northern Baja California. Their populations generally have been stable or increasing across most of their breeding range over the past several decades (Bechard and Schmutz, 1995), but declines have been observed in the periphery of their range and in northern and central Utah (Olendorff, 1993; Stepinsky and others, 2002). Analyses of Breeding Bird Survey data indicate that ferruginous hawk numbers have been stable in the shortgrass prairie and central mixed-grass prairie regions, both over the long term (1966–2011) and more recently (2001–1011) (Sauer and others, 2011). The ferruginous hawk was petitioned for listing under the Endangered Species Act in 1991 but was denied (U.S. Fish and Wildlife Service, 1992). Within the SGP ecoregion, the ferruginous hawk is listed as a species of management concern by the U.S. Fish and Wildlife Service (Region 6) as a sensitive species by the BLM (Travsky and Beauvais, 2005), as a species of special concern in Colorado, and as a species of conservation concern in Oklahoma.

Ferruginous hawks breed in open grasslands, sagebrush (*Artemisia* spp.) steppe, saltbush-greasewood (*Atriplex* spp.–*Sarcobatus vermiculatus*) shrublands, and on the periphery of piñon-juniper woodlands and other forests (Bechard and Schmutz, 1995). Within the SGP, they are found in shortgrass and mixed-grass prairie and areas with a mixture of grasslands and shrubs (Bechard and Schmutz, 1995). Ferruginous hawks generally choose elevated nest sites, such as boulders, creek banks, knolls, low cliffs, buttes, trees, large shrubs, utility structures, and haystacks, but they will nest on the ground when elevated sites are not available. Nests on the ground are generally located far from human activities and on elevated landforms within grassland areas (Blair and Schitoskey, 1982; Gilmer and Stewart, 1983; Preston, 1998), such as slopes, knolls, and ridge crests. In the winter, ferruginous hawks are generally found in grasslands, especially where prairie dogs (*Cynomys* spp.) are abundant, or near cultivated fields inhabited by pocket gophers (Geomyidae) (Bechard and Schmutz, 1995).

Ferruginous hawks feed primarily on lagomorphs and fossorial mammals. West of the Continental Divide, their diet mainly consists of jackrabbits (*Lepus* spp.) and cottontails (*Sylvilagus* spp.); east of the divide they primarily feed on ground squirrels (*Spermophilus* spp.), prairie dogs, and pocket gophers (*Thomomys* spp.) (Olendorff, 1993). There are few studies of ferruginous hawk diet within the SGP. In the Pawnee National Grassland of northeastern Colorado, breeding ferruginous hawks predominantly fed on thirteen-lined ground squirrels (*Ictidomys tridecemlineatus*); jackrabbits, cottontails, and pocket gophers were consumed less

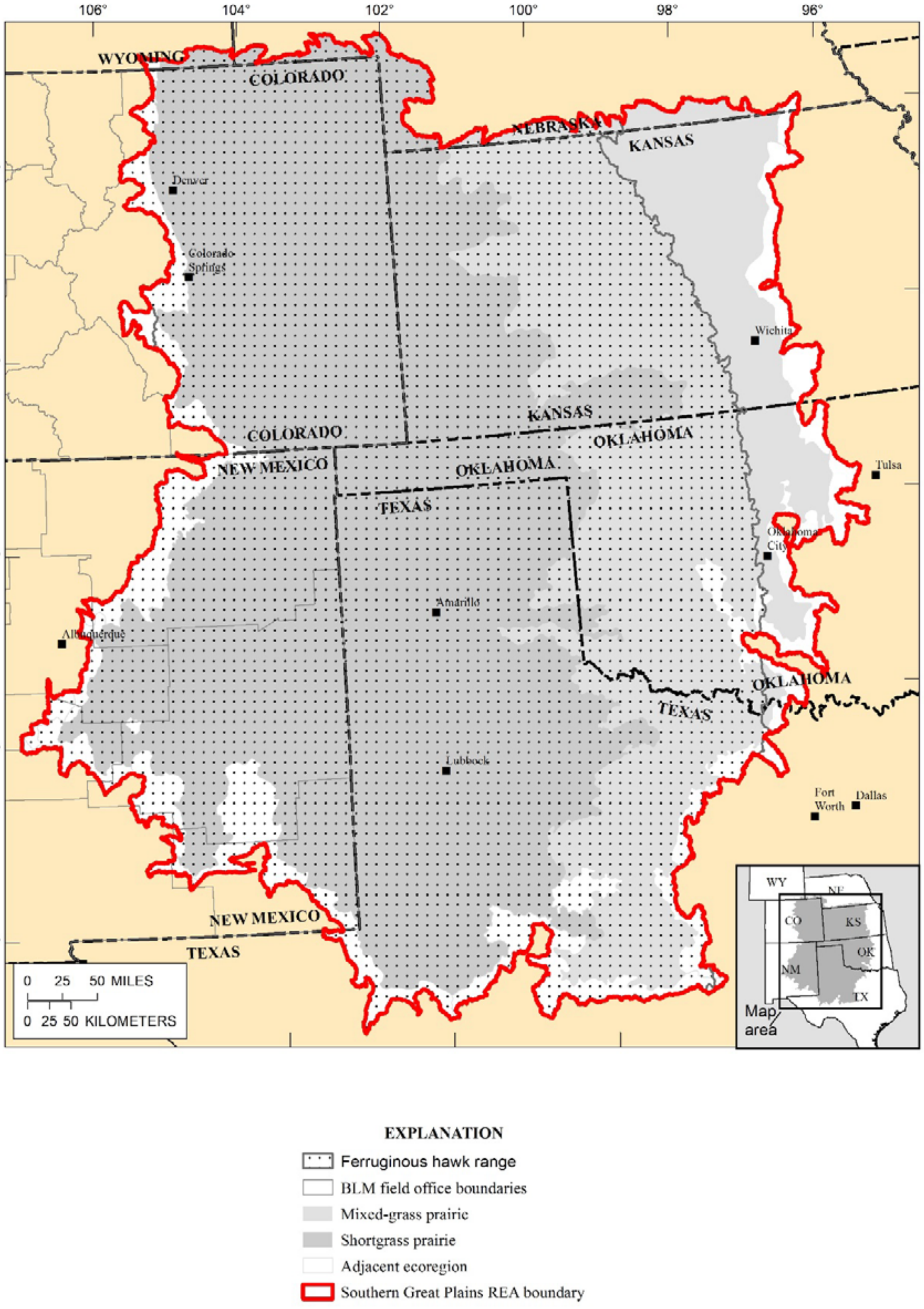

Figure 22. Current range of ferruginous hawk (*Buteo regalis*) in the Southern Great Plains ecoregion (data source: U.S. Geological Survey National GAP Analysis Program). (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (BLM, Bureau of Land Management; REA, Rapid Ecoregional Assessment)

frequently (Olendorff, 1973). In the panhandle of Oklahoma and adjacent areas of New Mexico and Texas, black-tailed prairie dogs (*Cynomys ludovicianus*) (29.9 percent), black-tailed jackrabbits (*Lepus californicus*) (15.8 percent), cottontails (*Sylvilagus* spp.) (14.4 percent), and yellow-faced and plains pocket gophers (*Cratogeomys castanops* and *Geomys bursarius*, respectively) (10.9 percent) accounted for 71 percent of the prey biomass delivered to ferruginous hawk nestlings (Giovanni and others, 2007). In the Estancia Valley of eastern New Mexico, Gunnison's prairie dogs (*Cynomys gunnisoni*) represented the most important prey biomass (Keeley, 2009), and mean productivity of ferruginous hawk nests close (less than 2 km) to prairie dog towns was greater than that of nests farther away (Cook and others, 2003). In the winter, prairie dogs are more important in the diet of ferruginous hawks because ground squirrels hibernate and are unavailable during that season. During migration and in winter, ferruginous hawks often associate with prairie dog towns (Cully, 1991; Bak and others, 2001), and their numbers decline locally following plague outbreaks that reduce prairie dog abundance (Cully, 1991; Seery and Matiatos, 2000). Ferruginous hawks are capable of moving long distances during the nonbreeding season (Watson, 2003), thus it is likely that local changes in abundance during the migratory and nonbreeding season are a function of the birds tracking changes in prey availability rather than a reflection of population-wide changes in abundance. On a smaller scale, presence of prairie dog towns was an important predictor of ferruginous hawk presence in winter along 1.6-km-long roadside transects at the Rocky Mountain Arsenal National Wildlife Refuge (NWR) near Denver, Colorado (Preston and Beane, 1996).

Unlike most other buteos, ferruginous hawks generally hunt on or close to the ground. A frequent hunting tactic is to perch on the ground near an active ground squirrel, pocket gopher, or prairie dog burrow and strike when the prey is near the surface. They also use still hunting, which entails observing prey from a perch followed by a short (less than 100 m) pursuit flight, and occasionally they will hunt from the air (Bechard and Schmutz, 1995).

Breeding Bird Survey data indicate that ferruginous hawk populations have been stable or increasing throughout most of their range and across much of the SGP over the past 47 years, except in eastern New Mexico, western Texas, and the Oklahoma panhandle, where declines have been observed (Sauer and others, 2011). Despite the long-term stability over most of their range, the number of breeding pairs at a location can change dramatically within a few years (Woffinden and Murphy, 1989; Olendorff, 1993). Large swings in breeding density of ferruginous hawks at a given location have been associated with changes in prey availability, and earlier studies suggested that hawks may exhibit nomadic behavior and move to areas with greater prey availability (Bechard and Schmutz, 1995). Recent studies using satellite telemetry, however, indicate that during years with low prey availability, birds may return to previous breeding sites but do not attempt to breed (Watson, 2003). Therefore, changes in the number of breeding pairs of ferruginous hawks at a given location over a short period (3–5 years) should be interpreted cautiously, as it may not indicate long-term population changes.

Landscape Structure and Dynamics

Nesting ferruginous hawks require large areas with little disturbance. Home-range size estimated using radiotelemetry varies from a mean of 5.9 km^2 in Utah (Bechard and Schmutz, 1995) to 79 km^2 in south-central Washington (Leary and others, 1998). No estimates of breeding home-range size within the SGP have been published. In the Pawnee National Grassland, 10 pairs of ferruginous hawks occupied a 1,072-km^2 study area, or 1 pair/109 km^2, which is an upper bound of home-range size (Olendorff, 1973). Human disturbance near nest sites causes nesting birds to flush from their nest and may lead to nest abandonment (White and Thurow, 1985; Keeley and Bechard, 2011; see "Development" under "Change Agents," below). Keeley and Bechard (2011) recommended restricting human activities within 650 m of nests to reduce disturbance to nesting pairs.

Because ferruginous hawks breed in shrub-steppe and grassland habitats, fires that cause a temporary conversion of shrub-steppe to grassland habitats likely will have little effect on ferruginous hawk distribution. Fire may indirectly affect ferruginous hawk distribution and abundance, however, through its effects on the abundance of prey species (Olendorff, 1993).

Associated Species of Management Concern

The golden eagle (*Aquila chrysaetos*) hunts in habitat similar to that used by foraging ferruginous hawks, and there is substantial overlap in their diet. Golden eagles, however, generally require tall cliffs for nest sites (Kochert and others, 2002). Home-range size of nesting golden eagles (20–33 km^2) is within the range of that used by ferruginous hawks; therefore, measures that protect foraging habitat for ferruginous hawks also should benefit golden eagles. Golden eagle populations are stable in western North America (Milsap and others, 2013), but because of their iconic status, they are protected under the Bald and Golden Eagle Protection Act (16 United States Code 668–668d). Other species that associate with ferruginous hawk habitat and respond in similar ways to threats, management, and conservation activities include Swainson's hawk (*Buteo swainsoni*), American kestrel (*Falco sparverius*), mourning dove (*Zenaida macroura*), great horned owl (*Bubo virginianus*), and loggerhead shrike (*Lanius ludovicianus*) (Colorado Partners in Flight, 2000).

Change Agents

Development

Ferruginous hawks appear to be more sensitive than other buteos to human disturbance, possibly as a result of their propensity to nest and hunt on or near the ground (Olendorff, 1993). Bechard and others (1990) found that ferruginous hawks nested farther from human habitation than either red-tailed hawks (*Buteo jamaicensis*) or Swainson's hawks (*Buteo swainsoni*). More direct evidence of their sensitivity to human activities comes from disturbance studies. When disturbances similar to those associated with development or recreational activities, such as walking towards the nest, driving towards the nest, running a generator near the nest, or firing a 0.22-caliber rifle while approaching the nest, were randomly applied to 24 of 62 active ferruginous hawk nests, 33 percent of the disturbed nests were deserted and those that remained active fledged fewer young than undisturbed nests (White and Thurow, 1985). Furthermore, when a pair deserted a nest following disturbance, territory occupancy was much lower (52 percent) the following year than undisturbed nests (93 percent), suggesting that the disturbance may reduce nesting densities in subsequent years.

The response of ferruginous hawks to human disturbance also appears to be influenced by landscape context. When ferruginous hawk nests were approached by a single observer walking towards the nest, the mean distance between the observer and the nest when the adult birds flushed was 486 m at rural nests and 340 m at nests in an exurban environment. If observers remained at least 650 m from the nest, adult birds flushed only 5 percent of the time; thus, 650 m was recommended as the buffer distance between human activities and ferruginous hawk nests (Keeley and Bechard, 2011). During the winter, however, ferruginous hawks appear to tolerate high amounts of human activity within their home ranges. There was no difference in home-range size, the number of perches used per day, or prey acquisition rates between ferruginous hawks wintering in a relatively undisturbed area at the Rocky Mountain Arsenal National Wildlife Refuge and birds that used adjacent Denver suburbs with high levels of human activity (Plumpton and Andersen, 1998).

Ferruginous hawks are vulnerable to recreational shooting and are subject to both direct mortality and indirect effects of lead poisoning from ingesting lead-contaminated carcasses. Shootings of ferruginous hawks have been reported in the literature (Harmata, 1981), although a decrease in the rate of birds recovered (through the bird band recovery system managed by the U.S. Fish and Wildlife Service) since the mid-1940s suggests that the number of hawks killed by shooting has declined (Houston and Bechard, 1984). Mortality of ferruginous hawks from shootings may still be a problem in some areas, but there are no recent summaries to indicate this. Recreational shooting of prairie dogs is a common activity throughout the western United States, and carcasses are often scavenged by raptors and other animals when they are not retrieved (Stephens and others, 2008). When lead ammunition is used, lead fragments remain in the carcasses (Knopper and others, 2006; Stephens and others, 2008) and raptors that consume them may be susceptible to lead poisoning (Fisher and others, 2006). Lead was detected in the blood of nestling ferruginous hawks and golden eagles in northeast Wyoming where recreational shooting of prairie dogs was common, but the lead levels were sublethal (Stephens and others, 2008). More studies are needed to evaluate the potential for lead poisoning among ferruginous hawks resulting from the recreational shooting of prairie dogs.

Although there is some evidence that ferruginous hawks may select nest sites farther from primary roads than other buteos (Bechard and others, 1990), elsewhere no effect on nest productivity was detected among ferruginous hawks nesting near (within 500 m) an interstate highway (Gilmer and Stewart, 1983). In north-central Montana, highly productive ferruginous hawk nests were closer to unimproved dirt roads than low-productivity or unproductive nests (Zelenak and Rotella, 1997). The authors suggested that this relationship likely was driven by the increased abundance of Richardson’s ground squirrels (*Urocitellus richardsonii*) along the edges of unimproved roads, but they also suggested that this relationship should be interpreted cautiously and may be limited to locations with similar prey populations and low human development.

Energy and Infrastructure

Studies of oil and gas development on nest-site selection by ferruginous hawks have yielded conflicting results, as ferruginous hawk nest locations have been found to be positively (Keough and Conover, 2012) and negatively associated (Smith and others, 2010) with oil and gas development. In addition, studies show inconsistent relationships between nest productivity and proximity to oil and gas development. In Utah, nest productivity was negatively associated with proximity to oil and gas wells (Keough, 2006), but in central Wyoming and north-central Montana, there were no directional relationships (Zelenak and Rotella, 1997; Smith and others, 2010). Ferruginous hawks have occasionally used infrastructure associated with oil and gas wells as nesting platforms, but those nests generally failed, probably due to disturbance (Smith and others, 2010). There are ongoing studies examining movement patterns and resource selection of ferruginous hawks in response to energy development in Wyoming, but results will not be available until after 2014 (John R. Squires, research wildlife biologist, U.S. Forest Service Rocky Mountain Research Station, Missoula, Mont., oral commun., March 2014).

Little information is available on the effects of wind energy development on ferruginous hawks, although they have been killed by wind turbines in California (Smallwood and Thelander, 2008), Oregon and Washington (Johnson and Erickson, 2011), and Wyoming (Johnson and others, 2000). Moreover, observers have noted that ferruginous hawks often fly close to wind turbines and at heights that make them vulnerable to spinning blades (Johnson and others, 2000; Smallwood and others, 2009). Although it is clear that ferruginous hawks are vulnerable to mortality from wind turbines, it is not clear what the mortality rates are or how it may affect population dynamics, and there is no published information on effects of wind development on nest placement or nesting success. Like other large raptors, ferruginous hawks are vulnerable to colliding with powerlines associated with all types of energy development. Indeed, occasionally they are killed by colliding with utility lines and electrocuted by perching on utility structures (Olendorff, 1993).

Agricultural Activities

For a given region, the relationship between ferruginous hawk nesting densities and the proportion of cropland has been found to be either negative (Gilmer and Stewart, 1983; Schmutz, 1984) or curvilinear with a slight density increase at low levels of cultivation (0−30 percent) followed by a density decline at higher levels (Schmutz, 1989). These relationships are probably a reflection of how cultivation affects the abundance and availability of the Richardson’s ground squirrel, the primary prey species of ferruginous hawks in the region studied (Schmutz, 1989). In another study, high-productivity nests were closer to croplands than low-productivity or unsuccessful nests, and Richardson’s ground squirrels were more numerous along the edges of cropfields (approximately 20 percent of the landscape) than they were in undisturbed grasslands (Zelenak and Rotella, 1997). This further supports the hypothesis that low levels of cultivation within a relatively undisturbed landscape may be beneficial to ferruginous hawks when their primary prey are Richardson’s ground squirrels.

Rodent-control programs may cause declines of ferruginous hawk numbers through direct poisoning and indirectly through reductions in prey populations. Two kinds of baits are currently approved for rodent control: those containing zinc phosphide and those containing anticoagulants, sometimes referred to as second-generation anticoagulant rodenticides (SGARs). Rozol, an SGAR commonly used to control prairie dogs, is permitted for use in all of the SGP states (EPA Reg. No. 7173-286). The SGARs are acutely toxic to raptors and have long half-lives, which increases the potential for bioaccumulation (Thomas and others, 2001). Indeed, SGARs have been implicated in the deaths of hundreds of raptors in the United States and Canada, including golden eagles and red-tailed hawks, but the U.S. Environmental Protection Agency has not documented ferruginous hawks dying from ingestion of SGARs (American Bird Conservancy, 2013). Ferruginous hawks, however, nest farther from human habitation than other buteos (Bechard and others, 1990), so birds that may be sickened by rodenticides are less likely to be encountered. Furthermore, ferruginous hawks are highly responsive to changes in prey abundance (Schmutz and Hungle, 1989), and reductions in prey due to rodent-control programs could lead to local declines in ferruginous hawks as well.

The effects of livestock grazing on ferruginous hawks are complex and may differ in different contexts (Olendorff, 1993). Compared to highly cultivated or urbanized areas, grazing lands provide much better nesting and foraging habitat for ferruginous hawks. In addition, restriction of human access on private lands is probably beneficial to nesting ferruginous hawks. Stocking rates, however, may have indirect effects on ferruginous hawks by influencing the abundance of their prey species (Olendorff, 1993).

Altered Fire Regime

Fire suppression and the subsequent expansion of quaking aspen (*Populus tremuloides*) woodlands into the prairie were identified as key factors in the disappearance of ferruginous hawks from the northern portion of their breeding range in Alberta (Schmutz, 1984). In the SGP, fire suppression could result in the expansion of juniper (*Juniperus* spp.) (Romme and others, 2009) and mesquite (*Prosopis* spp.) woodlands (Hagen and others, 2004), which could lead to a loss of ferruginous hawk breeding habitat (Olendorff, 1993). Eastern redcedar (*Juniperus virginiana*) also has expanded in the absence of fire in the SGP (Engle and others, 1996).

Invasive Species

Proliferation of cheatgrass was suggested as a possible cause of decline and local extirpation of a ferruginous hawk population in western Utah (Woffinden and Murphy, 1989), but more studies are needed to determine whether this is occurring in other locations, including the SGP. It has been suggested that cheatgrass (*Bromus tectorum*) may have negative effects on black-tailed jackrabbits (*Lepus californicus*), an important ferruginous hawk prey species in some regions (Olendorff, 1993). Expansion of eastern redcedar also could have negative effects on ferruginous hawk breeding habitat by changing the structure and function of small mammal habitat, thereby altering the hawk's prey base (Horncastle and others, 2005).

Disease

Unlike the statistics for other buteos, there are few documented cases of ferruginous hawk mortality resulting from infection by West Nile virus (WNv). In Colorado, there has been one documented case of WNv in a ferruginous hawk (Nemeth and others, 2007), but WNv does not appear to pose a serious threat to the species throughout its range (Collins and Reynolds, 2005). It is unclear, however, whether the lack of recorded cases is because the ferruginous hawks are not particularly susceptible to WNv or because these birds avoid humans and are less likely to be detected when sick or dead.

Climate Change

Climate model projections suggest the potential for temperature increases and changes in precipitation patterns for the SGP region, which could have negative effects on ferruginous hawk productivity. Ferruginous hawk nestlings often must endure very high temperatures because their nests are on or near the ground, where radiant heating can result in high ambient temperatures and shade is often unavailable (Tomback, 1981). Furthermore, nestling ferruginous hawks subjected to food deprivation exhibited higher and more varied body temperatures than those that were not food stressed (Tomback, 1981). If precipitation changes lead to decreases in food availability and extreme heat events increase as some climate models project (Karl and others, 2009), heat prostration could result in increased nestling mortality.

Rapid Ecoregional Assessment Components

A general conceptual model for the key ecological attributes and CAs affecting the ferruginous hawk is illustrated in figure 23. Key ecological attributes and CAs identified in the REA are enumerated in tables 26 and 27.

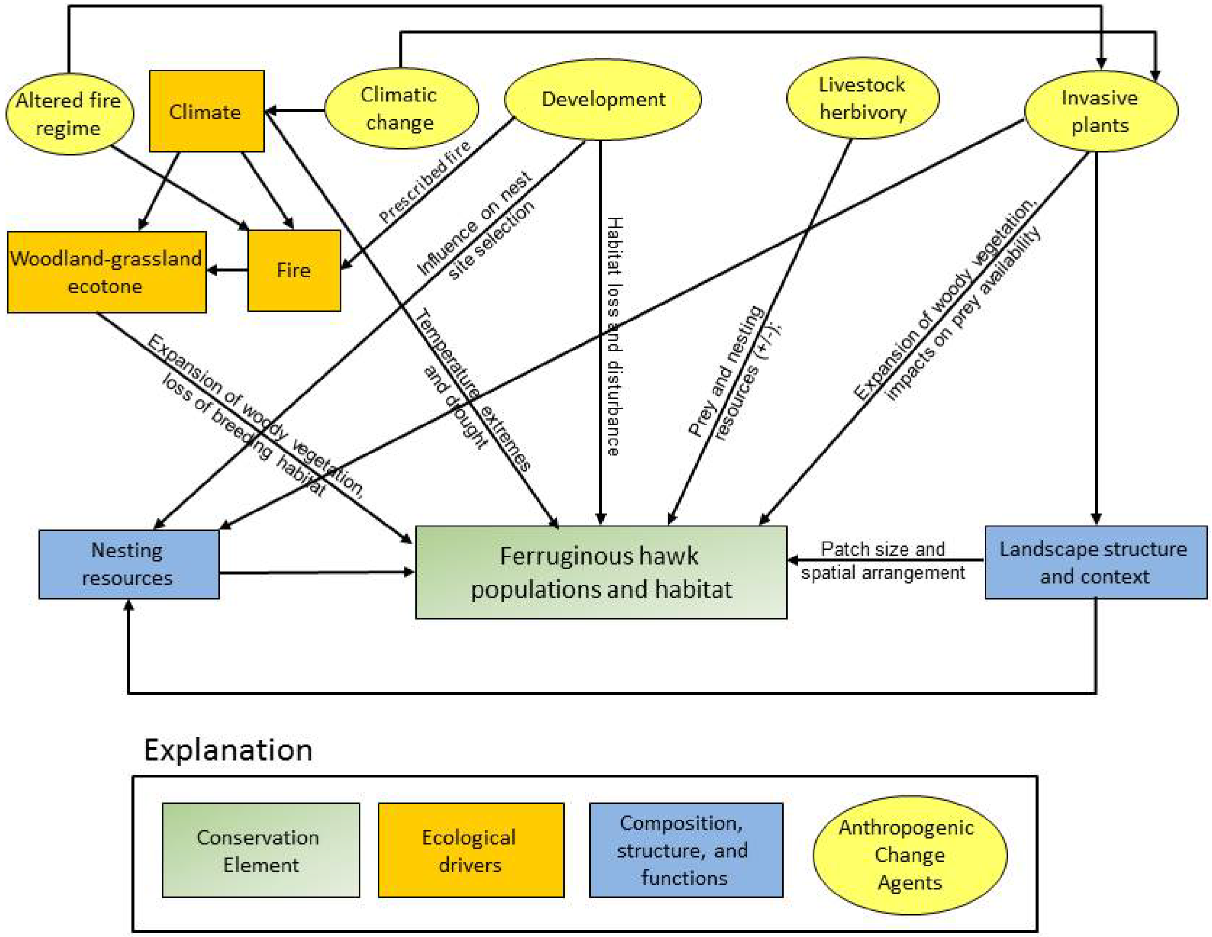

Figure 23. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for the ferruginous hawk (*Buteo regalis*) in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of ferruginous hawk populations and habtiats are shown in orange rectangles (see also table 26); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 27).

Table 26. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for the ferruginous hawk (*Buteo regalis*).

Attribute	Variables
Amount and distribution	Ferruginous hawk distribution (nest occurrence records).
Landscape structure	Size and spatial distribution of habitat, availability and type of nest locations, proximity of nest sites to human disturbance.
Landscape dynamics	Habitat productivity (prey availability), drought, woodland-grassland ecotone dynamics.
Associated species management of concern	Golden eagle, Swainson's hawk, American kestrel, mourning dove, great horned owl, and loggerhead shrike.

Table 27. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for the ferruginous hawk (*Buteo regalis*).

Attribute	Variables
Development (energy and infrastructure)	Habitat alteration (direct loss and disturbance of habitat), mortality (wind turbines).
Development (agricultural activities)	Habitat alteration (fluctuation in availability of prey [±]); rodent control (loss of prey, direct mortality, portal for poisons and lead to enter wildlife food chain).
Development (human effects)	Human influence on nest site selection and nest abandonment, habitat alteration (direct loss and disturbance of habitat, effects on prey availability), recreational shooting effects (direct mortality, ingesting of lead in carcasses).
Altered fire regime	Fire suppression results in habitat alteration (expansion of woody species, decrease in breeding habitat).
Invasive species	Habitat alteration (expansion in woody plants [juniper and mesquite], effects of cheatgrass on prey availability).
Introduced species and disease	Potential (unknown) presence and effect of West Nile virus on population.
Climate change	Drought effect on habitat condition (reduction in amount and quality of food resources due to drought), extreme heat events increasing on nestling mortality.

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Chapter 14. Interior Least Tern

By Linda C. Zeigenfuss

Key Ecological Attributes

Distribution and Ecology

The least tern (*Sternula antillarum*) is the smallest of the North American terns. These migratory, piscivorous birds breed throughout the coastal beaches and interior rivers of North America (Thompson and others, 1997). They winter along the Gulf Coast of the southern United States and Mexico, the Pacific Coast of Mexico, and the coasts of Central and South America. The interior least tern (*S. a. athalassos*) is the subspecies that nests and breeds on inland waters of the continental United States. The interior least tern (hereafter ILT) has been variously described as one of five subspecies of least tern (Thompson and others, 1997), but genetic studies have proven inconclusive in determining genetic distinctions between the interior population and other subspecies of *S. antillarum* (Whittier and others, 2006; Draheim and others, 2010, 2012). The ILT is separated geographically from populations of California least tern (*S. a. browni*), and they are separated from coastal populations of least tern (*S. a. antillarum*) by unique physiological and ecological factors related to nesting habitat.

The ILT's range in the Great Plains extends from Montana and North Dakota south through South Dakota, Nebraska, eastern Colorado and New Mexico, Kansas, Oklahoma, and Texas. In the SGP ecoregion, the ILT's range encompasses many areas along major rivers and reservoirs across the SGP. Known populations within the SGP ecoregion (fig. 24) are found along reservoirs and river sandbars associated with the Arkansas River in Colorado, Kansas, and Oklahoma; Quivira NWR in Kansas and Salt Plains NWR in Oklahoma; the Cimarron River in Kansas and Oklahoma; the Canadian and Red Rivers in Oklahoma and Texas; reservoirs along the Conchos River in Texas; and the Bitter Lake NWR and reservoirs associated with the Pecos River in New Mexico and Texas (Lott and others, 2013).

Least terns experienced rapid declines in the late 1800s and early 1900s because of egg collection and harvesting of feathers for the millinery trade until they became protected under the Migratory Bird Treaty Act of 1916 (Thompson and others, 1997). Loss of nesting habitats because of river channelization and dam building led to increasing habitat fragmentation and declines in ILT populations. In 1985 when the overall ILT population was estimated at 1,970, the ILT was listed as a federally endangered species (U.S. Fish and Wildlife Service, 1985, 2013). Recovery plans were established in 1990, and by 2005 the number of adult ILTs was estimated to be 17,500 (Lott, 2006). It is unknown, however, whether this increase was more attributable to habitat recovery efforts (habitat protection, water flow management, sandbar creation) or to improved population surveys, including greater rangewide coverage and increased efficiency in survey methods and (or) immigration from coastal least tern populations (Kirsch and Sidle, 1999; Lott, 2006; U.S. Fish and Wildlife Service, 2013). Since that time, no comprehensive, rangewide survey has been conducted, but in 2012, adult population estimates based on partial range surveys indicated a population size of more than 13,855 adults (U.S. Fish and Wildlife Service, 2013). A recent review of the ILT's current status suggested that population recovery goals have been met and recommended delisting, pending completion of metapopulation modeling and

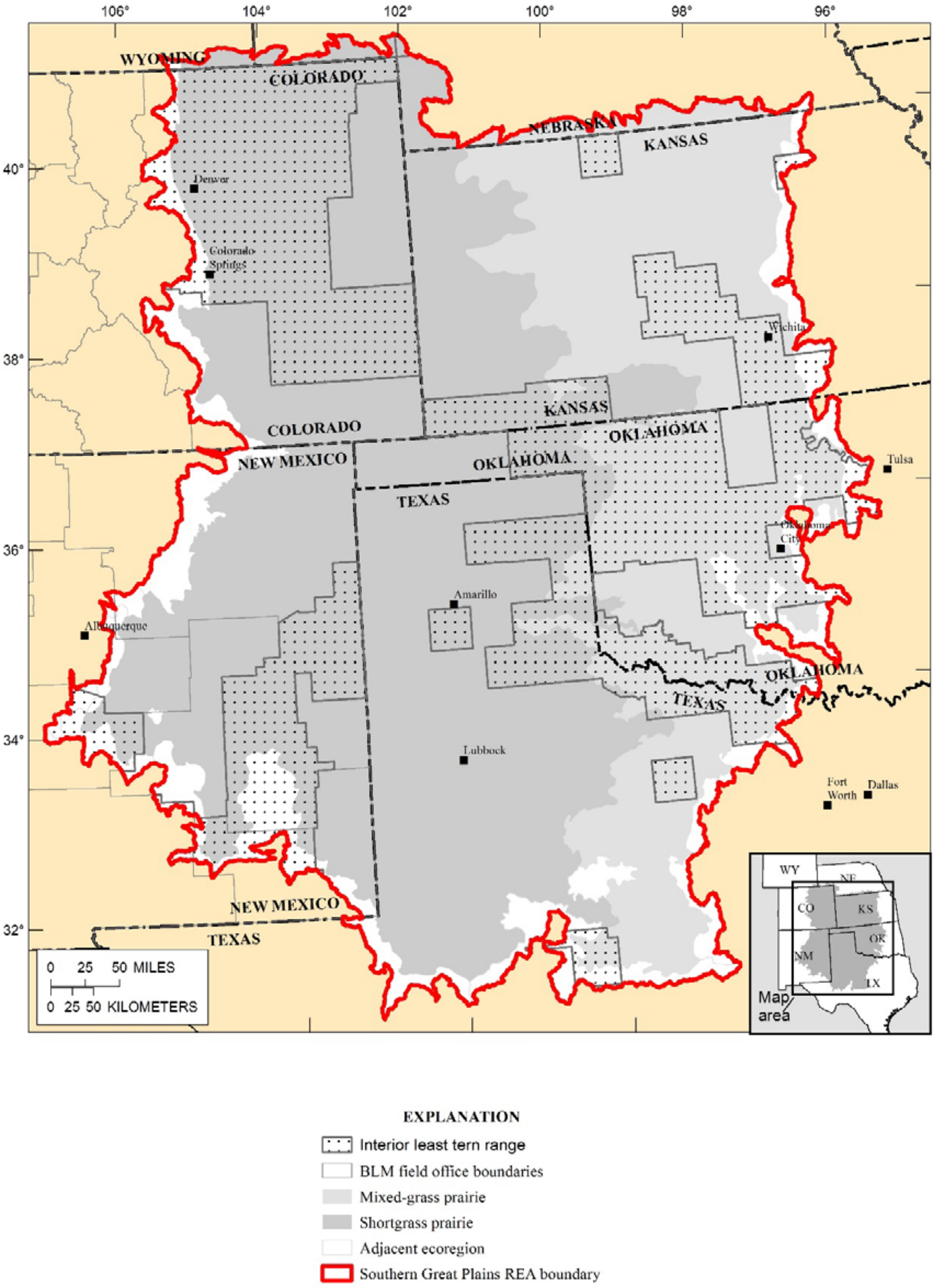

Figure 24. Current range of the interior least tern (*Sternula antillarum athalassos*) in the Southern Great Plains ecoregion (data source: U.S. Fish and Wildlife Service Species Profile). (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (BLM, Bureau of Land Management; REA, Rapid Ecoregional Assessment)

development of conservation agreements and plans for postlisting monitoring and management (U.S. Fish and Wildlife Service, 2013).

The ILT primarily feeds on small, slender-bodied fish species, such as gizzard shad (*Dorosoma* spp.) and shiners and minnows (Cyprinidae), as well as fingerlings of larger species, such as freshwater drum (*Aplodinotus grunniens*), and bass and sunfish (Centrarchidae) (Stucker, 2012). Least tern also occasionally feed on aquatic invertebrates (Thompson and others, 1997). Feeding habitats include various shallow-water habitats along rivers, streams, marshes, ponds, sand pits, reservoirs, and dike fields (Thompson and others, 1997). They prefer to feed in clear waters less than 1 m deep and where densities of forage fish are greatest (Thompson and others, 1997).

The long-lived ILT typically begins to breed at 2−3 years of age, and individuals are known to have nested successfully at up to 20 years of age (Thompson and others, 1997). The birds nest in colonies along sandbars and shorelines of rivers, reservoirs (Thompson and others, 1997; Lott and others, 2013), and alkali flats (Schweitzer and Leslie, 2000; Winton and Leslie, 2003), but they also will nest on gravel rooftops (Gore and Kinnison, 1991; Butcher and others, 2007; Forys and others, 2013) and in sand and gravel pits (Marcus and others, 2007; Jenniges and Plettner, 2008). Colony size has been observed to range from 0.2 to 4.4 ha (South Dakota; Schwalbach, 1988), and from 2 to 468 nests (Mississippi River Valley; Renken and Smith, 1995). Colonies are typically located in open areas, preferably less than 1 km but no more than 10 km from foraging habitat (Whitman, 1988; Sherfy and others, 2012; Lott and others, 2013). Nests are simple scrapes on unvegetated substrates characterized by coarse substrate (particle size), preferably in areas characterized by sparse or short (less than 25 cm high) vegetation and large debris that can provide shade for chicks (Smith and Renken, 1991; Stucker, 2012). The ILT prefers island habitats for nesting (midchannel gravel or sand bars) but also uses beaches, sand banks, and point bars (Lott and others, 2013). Nest-site conditions necessary for successful reproduction include (1) sites that are not inundated during the incubation and rearing seasons, (2) less than 30 percent vegetation cover and a distance of at least 150 m from shrubs and trees more 2 m high (they can harbor predators), and (3) availability of prey fish within 10 km to support chick growth until fledging (Lott and Wiley, 2012; Lott and others, 2013). These terns appear to prefer channels greater than 300 m wide (Jorgensen and others, 2012), but this constraint is likely related to distance from large trees, as they have been found to use sandbars in narrower channels when large trees are not present along river banks (Lott and others, 2013).

Because the ILT is a colonial nester that uses ephemeral habitats, social factors likely play a role in nest-site selection. For example, at abandoned coastal least tern colonies, more terns landed where decoy birds were used than where no decoys were used (Kotliar and Burger, 1984). Similarly in Illinois and Missouri, decoys and playback vocalizations were used successfully to attract ILTs to breed in artificial habitats where previously no breeding pairs had been observed (Ward and others, 2011). Least terns often abandon sites with marginal or declining habitat quality and rapidly colonize new sites, thus selecting higher reproductive success over site fidelity (Burger, 1984). Predation may lead to rapid colony abandonment, as predators often continue to return when they have located and successfully preyed upon a colony (Ward and others, 2011). This strategy of rapid colonization and abandonment also may allow least terns to keep a step ahead of predators and consequently increase their reproductive success (Ward and others, 2011). Not only does the ILT show little fidelity to previously used colony sites, but there is also a lack of fidelity to natal colony sites (Renken and Smith, 1995).

In the SGP, nesting and egg laying typically occur in May. Clutch size is typically 2−3 eggs (Thompson and others, 1997), and the eggs are incubated for approximately three weeks. The birds can renest, sometimes repeatedly, if eggs or chicks are lost early in the breeding season (Massey and Fancher, 1989). Chicks are precocial and leave the nest within two days but remain nearby until they fledge, which occurs at three to four weeks of age. The adults continue to feed their young for several weeks after fledging, and chicks typically disperse from their natal sites within three weeks of fledging (Thompson and others, 1997).

The eggs, chicks, and adults of ILTs are preyed on by a variety of avian and mammalian species. Documented predators include gulls (*Larus* spp.), herons and night-herons (*Ardea* spp. and *Nycticorax* spp., respectively), shorebirds, raptors, crows (*Corvus* spp.), various passerine birds, canids, raccoons (*Procyon lotor*), skunks (*Mephitis* spp.), Virginia opossums (*Didelphis virginiana*), feral hogs (*Sus scrofa*), catfish (*Ictalurus* spp.), Old World rats (*Rattus* spp.), and domesticated and feral dogs and cats (Schulenberg and Ptacek, 1984; Thompson and others, 1997). Eggs and chicks also incur mortality from a variety of other sources, including flood, exposure to extreme temperatures, burial from wind-deposited sand, pesticides, trampling, and nest-site disturbance.

Landscape Structure and Dynamics

The nesting and foraging habitats used by ILTs are ephemeral and dynamic, in large part because they are subject to flooding under natural river flow regimes. The flood events that may inundate nesting sites and cause widespread nest failure at a colony during in any given year are nonetheless important for replenishing the sand and gravel bar habitats that ILTs depend on. Scouring floods both remove encroaching vegetation and transport and deposit coarse substrates, thus constantly creating new sand bars and expanding and replenishing existing ones. The ILT is naturally adapted to take advantage of this dynamic environment and readily exploit suitable habitats throughout its range. The lack of strict fidelity to nesting or natal sites and the ability to disperse and forage over long distances (Lott and others, 2013) and renest within a breeding season allow these birds to take advantage of suitable habitat as it becomes available. Moreover, the diversity of prey fish species that they consume and their ability to travel large distances to forage (Sherfy and others, 2012) enhance their ability to survive fluctuations in prey availability.

Although the ILT is limited by habitat availability, including the loss of traditional habitats due to damming along the Mississippi and Missouri rivers, its adaptability to anthropogenic habitats has increased the quantity of nesting habitat and allowed it to increase its range longitudinally under less than ideal conditions. These artificial sites include reservoirs, industrial sites, gravel rooftops, and sand and gravel pits (U.S. Fish and Wildlife Service, 2013). Under regulated regimes of river flow, construction and maintenance of dikes and sandbars, along with vegetation management, these sites provide suitable nesting habitat that the ILT has used successfully (Stucker and others, 2013).

Demographic connectivity between subpopulations of ILTs appears to be high along major rivers in the Great Plains, as individuals dispersing from these subpopulations colonize available intervening habitat (Lott and others, 2013). Despite habitat fragmentation, ILT populations have been stable or expanding since they were federally listed as endangered in 1985, and their rapid colonization of new or previously unoccupied sites suggests a metapopulation dynamic; that is, populations are close enough to allow dispersing individuals to readily colonize unused habitat patches (Lott and others, 2013). Therefore, ILT-management activities based on a landscape scale are more likely to be successful than actions at the scale of patches or river segments, as populations may rapidly abandon one area while colonizing a new site nearby.

Although some ILT colonies appear isolated (sites in New Mexico and southeastern Colorado), their long-term persistence (more than 25 years) indicates that they can persist even where habitats are highly fragmented and long distances (more than 300 km) separate occupied habitats (Lott, 2006; Lott and others, 2013). This is due, in part, to the ILT's ability to move over long distances to forage and disperse. Actions to encourage ILT colony persistence since their listing in 1985 have included managing flows on regulated river stretches to replenish and maintain habitats and avoid flooding during the nesting season, construction and maintenance of sandbars and dikes from dredged material, and vegetation management. Several ILT populations in the SGP occur in close proximity to Gulf Coast populations. Although Gulf Coast populations are considered coastal least terns (*S. a. antillarum*), little genetic or phenotypic variation has been found between the coastal and interior subspecies (Whittier and others, 2006; Draheim and others, 2010, 2012). As such, it is likely that the degree of demographic connectivity between these populations is high, and individuals from both populations may be equally likely to colonize empty habitats between them (Lott and others, 2013).

Least terns spend the nonbreeding season in Central and South America, but little specific information is known about where interior populations go or what habitats they use. They follow major river basins to the Mississippi River and on to the Gulf of Mexico, where they mix with other least tern populations. In winter, the species is observed along coastal waters, bays, estuaries, and the mouths of rivers (Thompson and others, 1997), but without the use of special bands or other identification tools, it is not possible to distinguish ILTs from other populations on their wintering grounds.

Associated Species of Management Concern

The nesting habitat requirements of the federally endangered piping plover (*Charadrius melodus*) is similar to that of the ILT, and the two species' ranges overlap in the northern Great Plains (U.S. Fish and Wildlife Service, 2009). Indeed, monitoring and management programs in these areas often target both species because of the similarity of their habitat needs (Aron, 2005; Brown and others, 2011; Sherfy and others, 2011). In the SGP, both ILTs and piping plovers nest along reservoirs in southeast Colorado. Snowy plovers (*Charadrius nivosus*) (Chapter 18) also nest on sandy areas devoid of vegetation in saline/alkaline wetlands and along shorelines and sandbars (Page and others, 2009). The inland population of snowy plovers is listed as a species of conservation concern in Colorado, New Mexico, Oklahoma, and Texas; it is designated as threatened in Kansas; and the Bureau of Land Management and the U.S. Forest Service list it as a sensitive species in the SGP states.

Change Agents

Development

Energy and Infrastructure

Threats to the ILT from wind energy development have not been specifically documented, but they are assumed to be similar to threat identified for other migratory birds. Loss and others (2013) estimate that over a quarter of a million birds have fatal collisions with wind turbines in the contiguous United States each year. These authors found collision mortality rates to be lower in the Great Plains than other parts of the country and determined that tower height was positively correlated with mortality rates (Loss and others, 2013). Because ILTs migrate through the Gulf of Mexico, they are subject to hazards from both land-based and offshore wind energy facilities (Kuvlesky and others, 2007). Although mortality rates associated with wind turbines reported by Loss and others (2013) were lower than those in some previous reports and lower than rates reported for other anthropogenic sources of mortality (collisions with buildings and communication towers, feral and pet cats), they emphasized the importance of assessing species-specific and location-specific risks when siting wind energy developments.

Agricultural Activities

Because ILTs nest along river bars and reservoir shores, they are not likely to be directly affected by agricultural operations. The demand for water from reservoirs for agriculture, however, may affect nesting habitat. When water is released or drawn from reservoirs during high-demand seasons or drought, there may be rapid changes in water levels that can affect nesting habitat or food availability. Runoff from irrigated and pesticide-treated fields also may affect food supplies, leading to bioaccumulation of pesticides in predators such as the ILT. Mercury, selenium, and organic pollutants were found at low to moderate levels in ILTs that were tested for contaminants (Fannin and Eamoil, 1993; Ruelle, 1993; Allen and others, 1998; Sanchez and Caldwell, 2008). Although researchers concluded that these compounds are not being ingested at levels high enough to affect reproductive success or adult survival, they were unable to determine whether toxic residues were causing egg mortality (Fannin and Eamoil, 1993). The existence of moderate levels of contaminants in some populations may be a cause for concern in those populations. Nests of ILTs in the northern Great Plains have been trampled by grazing livestock (Schulenberg and Ptacek, 1984; Aron, 2005), although the percentage of nests lost was very low (less than 1 percent of more than 4,000 monitored nests) (Aron, 2005).

Dams and Diversions

Natural river flow regimes create and maintain ILT habitat, but flows altered by channelization and impoundments present the greatest threats to this species (U.S. Fish and Wildlife Service, 2013). Damming of major rivers has flooded miles of shoreline and sand and gravel bars along the Mississippi, Missouri, and Ohio Rivers and their tributaries. Below dams, river flows are regulated, disrupting the natural erosion and deposition processes that create and maintain sand and gravel bars where ILTs nest (U.S. Fish and Wildlife Service, 1985). Natural scouring floods also remove vegetation, which increases ILT habitat quality. Water releases from dams that occur during the nesting season can inundate nests and cause nesting failure and mortality of unfledged young.

Recreation

Fishing, boating, swimming, and other recreational activities along shorelines in the vicinity of ILT colonies can lead to nest trampling and disturbance by recreators and their pets (Aron, 2005). Human disturbance can cause adult terns to flush from nests, leaving eggs exposed to hot or cold temperatures for long periods of time and sometimes leading to nest abandonment. Off-highway vehicles can crush nests and chicks (Smith and Renken, 1993).

Invasive Species

Nesting habitat of the ILT is susceptible to vegetation encroachment, particularly from fast-growing plant species that colonize disturbed areas. Two nonnative plants that can be particularly troublesome in ILT habitats are tamarisk or saltcedar (*Tamarix* spp.) and kochia (*Kochia* spp.). Tamarisk is an invasive woody plant that thrives in arid and semiarid riparian habitats, and it is particularly tolerant of the saline conditions found on alkaline flats where ILTs often nest in the SGP. Tamarisk invasion has reduced ILT nesting habitat in Kansas and Oklahoma, and nests near stands of tamarisk experienced greater rates of mammalian predation (Schulenberg and Ptacek, 1984; Koenen and others, 1996; Schweitzer and Leslie, 1999; Winton and Leslie, 2003). It appears that kochia is not as large a threat to ILT nesting habitat as tamarisk, but it has been found in some sites (Anteau and others, 2012), including a nesting site in New Mexico where it was removed when habitat improvement actions were applied (Ahlers and Moore, 2012).

Red imported fire ants (*Solenopsis invicta*) are native to South America and were introduced to the United States in the 1930s in Mobile, Alabama (Allen and others, 2004). Since that time they have spread throughout the southeastern states and currently are found in Texas, Oklahoma, and New Mexico. Fire ants have been documented as predators of several ground-nesting avian species (Allen and others, 2004), and they were found to harass and possibly prey on least tern colonies in Mississippi (Lockley, 1995) and Georgia (Krogh and Schweitzer, 1999). Live chicks were observed with ant stings and multiple fire ants clinging to them, and dead chicks and pipped eggs were covered with ants, although the cause of death in these cases was not conclusive. Climate warming could lead to northern expansion in the distribution of these ants (Morrison and others, 2005).

Introduced Insects and Disease

Little is known about disease in the ILT; however, it does not appear to be a major factor in natural mortality. Indeed, disease was not listed as a mortality factor for any individual (adult or chick) in a multiyear study of ILT on the Salt Plains National Wildlife Refuge in northwestern Oklahoma (Schweitzer and Leslie, 2000). There are, however, confirmed West Nile virus infections in least terns (Centers for Disease Control and Prevention, 2013).

Climate Change

Climate change has the potential to affect populations of ILT directly through altered river flow regimes. Extreme high-flow events may result in loss of annual recruitment, although on undammed rivers these events can provide the scouring flows necessary to remove encroaching vegetation and create or renew sandbars that provide high quality ILT nesting habitats (Sidle and others, 1992). On the other hand, extreme drought events may lower water levels (through reduced precipitation and increased anthropogenic demands on water resources) to the point that nesting islands become connected to the mainland, which would lead to an increased threat of predation on ILT colonies. Lower water levels also expose areas of unvegetated shoreline, potentially exposing new suitable nesting areas, which could lead to increased expansion of nesting efforts and potentially increased local recruitment rates (U.S. Fish and Wildlife Service, 2013). Increasing summer temperatures, however, may lead to catastrophic recruitment failure due to chick exposure to heat, particularly for roof-nesting birds (Watterson, 2009); however, the ILT is dispersed along wide latitudinal and longitudinal gradients of climate conditions, thus it is unlikely to experience rangewide catastrophic recruitment failure due to high summer temperatures (U.S. Fish and Wildlife Service, 2013).

Rapid Ecoregional Assessment Components

A general conceptual model for the key ecological attributes and CAs affecting the interior least tern is illustrated in figure 25. Key ecological attributes and CAs identified in the REA are enumerated in tables 28 and 29.

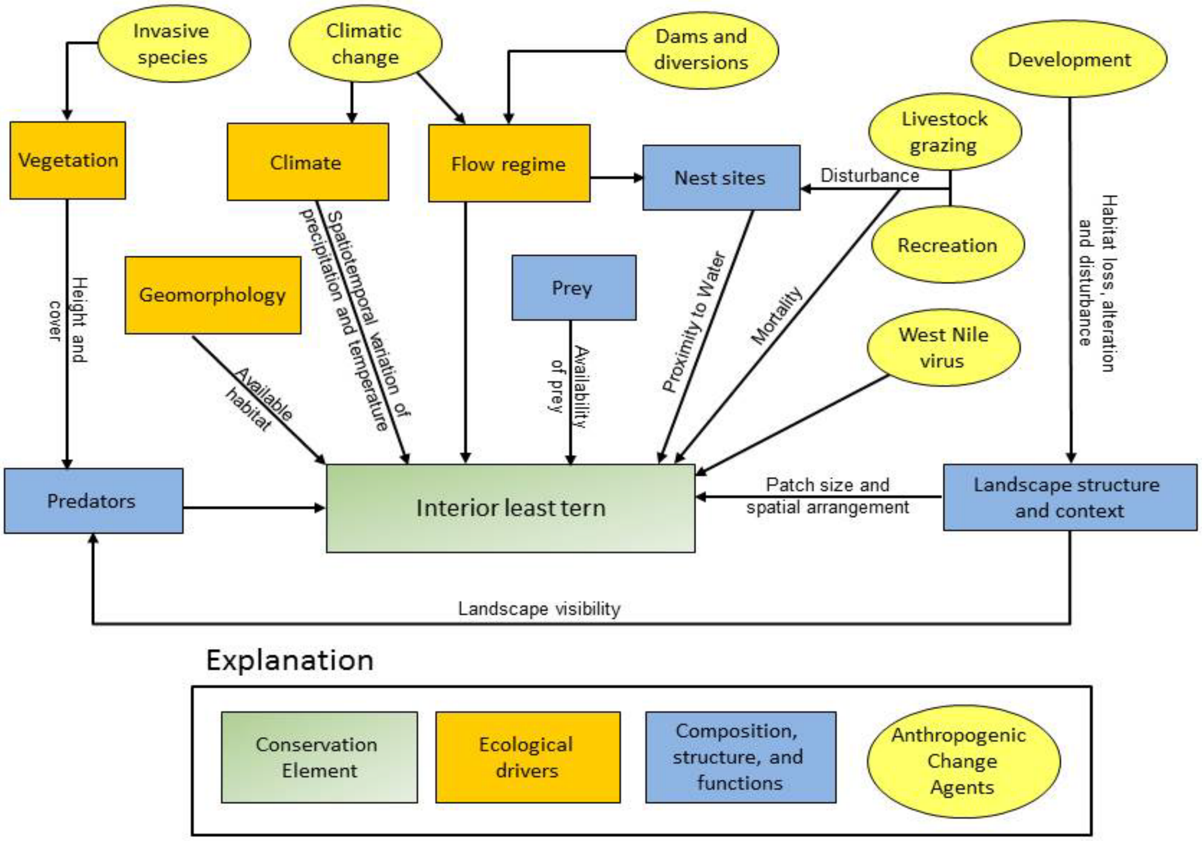

Figure 25. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for the interior least tern (*Sternula antillarum athalassos*) in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of interior least tern populations and habtiats are shown in orange rectangles (see also table 28); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 29).

Table 28. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for the interior least tern (*Sternula antillarum athalassos*).

Attribute	Variables
Amount and distribution	Amount of interior least tern habitat and distribution.
Landscape structure	Size and spatial distribution of available habitat (length of exposed shoreline), forage availability.
Landscape dynamics	Flow regime, extreme flood events.
Associated species management of concern	Piping plover (threatened) and snowy plover (high conservation concern) have similar nesting habitat requirements.

Table 29. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for the interior least tern (*Sternula antillarum athalassos*).

Attribute	Variables
Development (energy and infrastructure)	Collisions with wind turbines (mortality).
Development (agricultural activities)	Habitat alteration (demand for water affects nesting habitat), pollutants from runoff (bioaccumulation in prey base), livestock grazing (trampling of nests).
Development (dams and diversions)	Habitat alteration (impoundments/reservoirs [change in flow regime], channelization [direct loss of habitat], loss of nesting habitat [upstream of reservoir = flooding; downstream of reservoir = scouring]).
Recreation	Nest disturbance/trampling.
Invasive species	Predation (red imported fire ants), habitat loss/alteration (*Tamarix* spp. invasion).
Insects and disease	Presence and effect of West Nile virus on population.
Climate change	Habitat alteration (changes to flow regime [drought effects, extreme high flow event]), extreme temperatures (mortality of chicks).

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Chapter 15. Lesser Prairie-Chicken

By Cynthia P. Melcher

Key Ecological Attributes

Distribution and Ecology

The lesser prairie-chicken's (*Tympanuchus pallidicinctus*) current range is encompassed entirely by the SGP ecoregion (fig. 26). Occupied areas are limited to extreme southeastern Colorado, Kansas south of Interstate 70 and west of Great Bend, the eastern panhandle and adjacent counties of northwestern Oklahoma, the Texas panhandle, and southeastern New Mexico (Hagan and Giesen, 2005; U.S. Fish and Wildlife Service, 2008). Originally, the species reportedly ranged from northeastern Colorado and the Nebraska Sandhills south to central Texas and the northern Trans-Pecos boundary and the western two-thirds of Kansas (Hagan and Giesen, 2005). Although the lesser prairie-chicken (hereafter, LPC) has recently expanded its current range farther north in western Kansas in areas where it had occurred historically (as well as in areas it probably had not occurred historically), it presently occupies only about 8 percent of its historic range and populations are highly fragmented.

Initially, market hunting in the late 1890s and overhunting in subsequent years reduced LPC populations, but the main factor currently driving their decline is habitat loss, fragmentation, and degradation due to changing land uses, exacerbated by drought (Hagan and Giesen, 2005). The most recent comprehensive LPC survey revealed an estimated 50 percent population decline from 34,440 birds in 2012 to 17,616 in 2013, most likely driven by the severe drought in 2012 (McDonald and others, 2013). In early 2014, after nearly two decades since initially being petitioned for listing, a final ruling was made to list the LPC as a threatened species under the Endangered Species Act (U.S. Fish and Wildlife Service, 2014).

A geographic information system (GIS)–based analysis of habitat use in Oklahoma, Texas, and New Mexico indicated that landscapes inhabited by LPCs are characterized by mosaics of several habitat types. Native dwarf shrublands and mixed grasslands compose the majority of LPC habit. Typically, LPCs use vegetation types that grow in sandy soils, particularly communities dominated by shinnery oak or sand sagebrush and interspersed with grasses, particularly bluestem species (*Andropogon* spp.) but also sand dropseed (*Sporobolus cryptandrus*), sideoats grama (*Bouteloua curtipendula*), threeawn species (*Aristida* spp.), and blue grama (*B. gracilis*) (Woodward and others, 2001; Hagan and Giesen, 2005). Variation in dwarf shrub communities across the LPC's range may contribute to regional differences in habitat associations (Timmer and others, 2014); in Colorado and Kansas the birds use predominantly sand sagebrush–bluestem types, and in southern portions of their range they use predominantly shinnery oak–mixed-grass types (Hagan and Giesen, 2005). In northwestern Kansas, however, where the LPC range is expanding somewhat, there are no shrub components; there, the birds are using mosaics of shortgrass and lands enrolled in the Conservation Reserve Program that are planted with dense mixes of native, warm-season, tall and mid-height grasses (Van Pelt, 2013). Overall, LPCs use habitats with varying proportions of shrub, grass, and forb cover, depending on their life stage (for example, nesting and brood rearing) and the quality of the vegetation structure (Hagen and others, 2004).

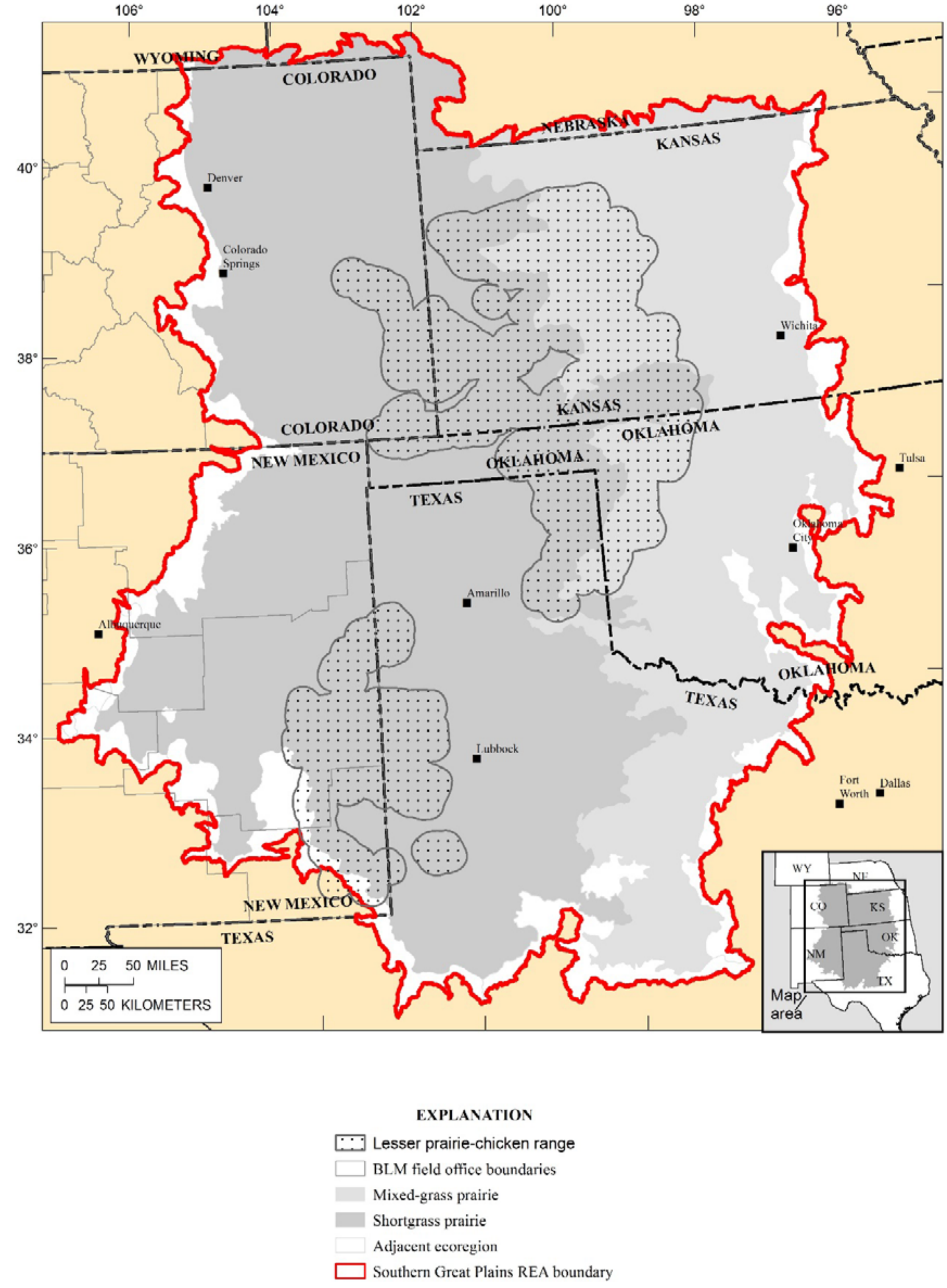

Figure 26. Current range of lesser prairie-chicken (*Tympanuchus pallidicinctus*) in the Southern Great Plains ecoregion (data source: Van Pelt, 2013). (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (BLM, Bureau of Land Management; REA, Rapid Ecoregional Assessment)

Escape and thermal cover is a crucial aspect of LPC habitat, without which the birds are more vulnerable to predation and negative effects of temperature extremes. Canopies of sand sagebrush, shinnery oak, and relatively tall, dense grasses provide LPCs with escape cover from ground and aerial predators during all life stages. Sand sagebrush–bluestem and shinnery oak habitats also provide essential forage items consumed by LPCs, including mast (seeds, acorns), buds, and leaves of shrubs, forbs, and (or) grasses. Native shrubland-grassland habitats provide important invertebrate prey, including short- and long-horned grasshoppers (Acrididae and Tettigoniidae, respectively), beetles (Coleoptera), and treehoppers (Membracidae), as well as insect galls (Riley and Davis, 1993; Jamison and others, 2002). Juveniles consume primarily invertebrates, whereas adults consume both invertebrates and vegetable matter in spring and summer and predominantly vegetable matter in fall and winter (Hagan and Giesen, 2005). Although LPCs will drink water from stock ponds and other waterbodies, particularly during drought, it is believed that they do not require free water because their historical range was not limited by water availability (Hagan and Giesen, 2005).

The focal point of the LPC's home range is the lek. Fidelity to lek sites promotes long-term lek use, provided that disturbances at or near the lek are minimal (Hagan and Giesen, 2005). LPCs may establish new lek sites where vegetation is sparse or eliminated by disturbances such as fire. Adult birds generally restrict their activities to within 4.8 km of the lek (Hagan and Giesen, 2005). Severe winter weather or prolonged drought, however, may prompt LPCs to make longer movements to foraging locations such as small-grain fields (Hagen and others, 2004; Hagan and Giesen, 2005). In Kansas, most LPC movements were less than 30 km, but of those that moved farther, most did not return to original capture sites, suggesting that they had dispersed (Hagen, 2003).

Breeding season for the LPC extends from approximately March 1 through June 30, with male lek attendance generally peaking in mid-March through May (Van Pelt, 2013). The number of males attending a lek is influenced by habitat quality and local population density and is one indicator of local population size (Jamison and others, 2002). Lek sites are usually minimally vegetated and often somewhat higher in elevation than the surrounding terrain (Hagan and Giesen, 2005). The birds exhibit strong fidelity to both their lek sites and their home ranges, which include the leks and suitable adjacent foraging, roosting, loafing, and nesting habitats (Hagan and Giesen, 2005).

Generally, LPC hens nest within 3 km of the lek where they mated (Hagan and Giesen, 2005). Nests sites, often located on northeast hillsides with less than 6 percent slope, typically have greater vertical (43–81 cm) and horizontal cover than that of surrounding areas. Successful nests are found more often where shrub and grass cover are tall and grass density is relatively high (Hagen and others, 2004). More specifically, nests located under a dwarf shrub or a tall clump of bunchgrass with at least 70–80 percent litter cover on the adjacent surface are more successful than those placed elsewhere (Hagan and Giesen, 2005). The primary cause of nest failure is egg predation by small and midsized mammals and snakes (Pitman and others, 2006).

Ideal brood habitat also has relatively tall vegetation with high vertical density. Overall, brood habitats are characterized by 25–35 percent canopy cover of shrubs, forbs, and grasses, and grasshopper biomass (important brood prey) is greatest where shrub density is low enough to permit significant forb cover (Hagen and others, 2005). Brood habitats with approximately 25 percent grass, forb, and (or) shrub cover provide escape cover (Hagen and others, 2004; Hagen and others, 2005). Typically, LPCs roost on the ground where vertical cover is less than 1 m but taller than that of surrounding foraging sites. Roosting sites may include CRP fields and draws, ridges, or knolls (Doerr and Guthery, 1983; Giesen, 2000; Hagan and Giesen, 2005).

Landscape Structure and Dynamics

Although the LPC is nonmigratory (Hagan and Giesen, 2005), it is considered a landscape species because of its large home range (up to 3,000 ha), which encompasses multiple habitat types, including a mosaic of foraging, nesting, and lekking habitats (Woodward and others, 2001). Silvy and others (2004) have indicated that the quality and quantity of native habitat and connectivity of native habitat patches (that is, a contiguous area of one vegetation type) are necessary for the persistence of prairie-chicken species (*Tympanuchus* spp.). Accordingly, current LPC management guidelines specify the need to maintain, protect, and restore large areas of shinnery oak–tallgrass or sand sagebrush habitat, particularly areas larger than 2,000 ha with interpatch spacing of less than 30 km (Hagen and others, 2004).

Although fire has been important in shaping SGP ecosystems, presettlement fire regimes and spatial patterns in LPC range are poorly understood. Furthermore, they likely varied significantly across the SGP region and habitat types. Nonetheless, it is believed that fire helped to control the expansion of woody plants and create mosaics of various other habitat types in various seral stages (Boyd and others, 2013). Because LPC lek sites are usually on sparsely vegetated or bare ground (Sell, 1979; Hagan and Giesen, 2005; Van Pelt, 2013), fire likely helps to maintain existing lek sites (Hagen and others, 2004), and new lek sites have formed on burn sites (Van Pelt, 2013). Fire also may enhance populations of invertebrates that LPCs prey on by reducing native shrub densities and temporarily enhancing conditions for forbs and grasses. In Oklahoma, cool- and warm-season forbs, which are important grasshopper forage, increased 1–2 years postfire in shinnery oak habitat; grass and shrub cover, however, were reduced and there was no production of shinnery oak leaf buds, acorns, or catkins (Boyd and others, 2013). If fire is pervasive enough in a given landscape to decrease or eliminate significant amounts of nesting and brood-rearing habitat, escape and thermal cover, and production of shinnery oak acorns, LPC survival and population growth also may be diminished (Fuhlendorf and others, 2002; Hagen and others, 2004; Boyd and others, 2013). Overall, periodic, small fires during fall or winter in various LPC habitat types scattered across the landscape at the home range scale would help to ensure that brood habitat is available (for example, high densities of forbs and grasshoppers) while maintaining adequate shrub canopy cover (Boyd and others, 2013).

Soils and precipitation strongly affect the landscape structure and dynamics within the LPC's range. Sand sagebrush typically grows on light, sandy soils with low silt and clay content (U.S. Forest Service, 2014), whereas shinnery oak typically grows on rolling sandhills on dunes of deep sandy loams (U.S. Forest Service, 2005); as clay content increases, shinnery oak dominance gives way to other vegetation types. Precipitation patterns influence the predominant grass types in habitats occupied by LPCs, with short grasses dominating the drier western portion of the LPC's range and tall to mixed grasses dominating the moister central and eastern portions of their range (Woodward and others, 2001; Hagan and Giesen, 2005). The timing of precipitation, however, determines which plant species will be more productive in a given year (Fields and others, 2006). Severe or prolonged drought may result in vegetation dieback that diminishes nesting, thermal, and escape cover, and it diminishes the abundance and productivity of important LPC foods, such as forbs and insects, which are critical for brood survival (Jamison and others, 2002; Fields and others, 2006). Drought also may lead to smaller clutch sizes, fewer nesting attempts, and diminished nest success, particularly when drought is accompanied by above-average temperatures during incubation (Grisham and others, 2013).

Undoubtedly, wild herbivores, including ungulates and invertebrates, also helped to shape the landscape structure and dynamics of LPC habitats. The extent to which native herbivores represented costs or benefits to LPC populations is poorly understood and requires further study; however, it is likely that the American bison (*Bison bison*) and other herbivores helped to maintain landscape heterogeneity that resulted in a mosaic of vegetation density and height and patches of shrubs and grasses interspersed with bare ground (Patten and others, 2005). It is also likely that ungulate trampling helped to open or maintain sites suitable for lekking activities.

Associated Species of Management Concern

The dunes sagebrush lizard (*Sceloporus arenicolus*), which occurs in the southeastern region of the SGP, also depends on shinnery oak habitat. More specifically, it forages for its insect prey shinnery oak shrubs in the sandy habitats of small blowouts, and the roots of oak shrubs provide the underground structure that helps to support the lizard’s burrows (Degenhardt and Jones, 1972). Therefore, the LPC could serve as an umbrella species for the lizard, as shinnery oak habitat is crucial to LPCs in the southern portions of their range. Both species are threatened by loss of shinnery oak habitat due to energy development and misapplications of rangeland management practices, including widespread applications of herbicides (followed by fire) to eliminate shinnery oak (U.S Fish and Wildlife Service, 2010; Van Pelt, 2013).

Change Agents

Habitat loss and degradation are the greatest threats to LPC population viability, including the rate of habitat change and the total area changed (Woodward and others, 2001). Specifically, agricultural conversion, reverting CRP lands to croplands, misapplications of livestock grazing and range management practices, and energy development (fossil fuels and wind, including the associated infrastructure) are significant contributors to habitat degradation and loss (Hagen and others, 2004). Fire suppression, which leads to expansion of woody plants, and widespread use of herbicides to eliminate shinnery oak for improving livestock range further degrade or fragment LPC habitats. Combined, these factors can reduce patch size to less than 500–2,000 ha (Hagen and others, 2004), diminish habitat heterogeneity, promote greater predator densities in remaining patches of suitable habitat, and hinder recolonization as remaining patches become more isolated. These factors also make LPC populations more susceptible to the effects of catastrophic events, such as severe drought (U.S. Fish and Wildlife Service, 2008). Moreover, the birds’ strong site fidelity may hamper or preclude them from dispersing when their habitats are altered or converted.

Development

Energy and Infrastructure

Energy development, including fossil fuels and wind, has been implicated in the fragmentation and degradation of LPC habitat, largely because the birds avoid areas affected by energy development (Hagen and others, 2011). Indeed, accelerating rates of wind turbine development in the LPC's range is an important reason for the species' recent listing under the Endangered Species Act (U.S. Fish and Wildlife Service, 2014). Overall, LPCs avoid anthropogenic features and activities associated with energy exploration, development, and extraction, effectively eliminating and fragmenting otherwise suitable habitat (Hagen and others, 2004; Jarnevich and Laubhan, 2011); avoidance distances may be influenced by topography and intensity of activity and noise (Hagen, 2003; Pruett and others, 2009; Dusang, 2011; Hagen and others, 2011). Hens generally avoid nesting within 80 m to 1 km of well pads, buildings, improved roads, and transmission lines (Pitman and others, 2005), and collisions with powerlines and fences may be a source of LPC mortality in some areas (Wolfe and others, 2007). Although LPC leks are often found on sites disturbed by anthropogenic activities, including abandoned well pads and unimproved roads (Sell, 1979; Hagan and Giesen, 2005), they also may abandon leks due to noise and activity associated with energy development or increased predator activity associated with energy infrastructure (Hagen and others, 2004).

Agricultural Activities

Effects of agriculture on LPCs are mixed. Conversion of native grassland to cropland has been implicated in rangewide declines of LPCs (Hagen and others, 2004). Overall, LPC populations decline where agricultural cultivation of any type exceeds more than 37 percent of the landscape (Hagen and others, 2004; Hagan and Giesen, 2005). Generally, lekking males and nesting hens avoid cropland edges and disturbances associated with cultivation, including center pivot irrigators (Crawford and Bolen, 1976). Hens rarely use croplands for nesting and only occasionally for brood rearing (Fields and others, 2006). Small-grain crops, however, such as grain sorghum and wheat, provide important foraging habitat for the birds, especially in winter or during drought when native foods are depleted (Applegate and Riley, 1998). In fact, landscapes that encompass up to 37 percent small-grain cropland may support greater LPC densities than landscapes of 100 percent native habitat (Crawford and Bolen, 1976). Lekking males and prenesting hens also have been reported foraging in alfalfa fields (Hagen and others, 2004).

The enrollment of marginal croplands into the CRP program and minimum-till cultivation methods have improved conditions for LPCs in croplands (Hagan and Giesen, 2005); LPCs will use CRP habitats for lekking, nesting, brood rearing, and winter cover (Giesen, 2000; Fields and others, 2006), particularly where the seed mixes include native forbs and grasses (Silvy and others, 2004). However, many CRP grasslands have been disenrolled and put back into agricultural production. Various economic factors are driving the acceleration of disenrollment, which could negatively affect LPC habitat throughout its range (U.S. Fish and Wildlife Service, 2008).

Inappropriate livestock grazing practices, including a lack of (or too little) pasture rotation, have greatly degraded LPC habitat in portions of the LPC's range. Although short-duration grazing can improve LPC habitat quality, mismanaged livestock grazing can damage the macro- and microhabitat structure required for successful nesting and protection from predation and weather extremes (Hagen and others, 2004; Patten and others, 2005; Van Pelt, 2013). Trampling associated with intensive livestock grazing can damage or destroy LPC nests (Pitman and others, 2005). Although careful use of herbicides to create mosaics of shinnery oak and grass/forb cover can improve LPC habitat, the widespread practice of applying herbicides for eliminating shinnery oak to improve livestock grazing rangeland can be detrimental to LPC habitat by reducing crucial escape and thermal cover and an important source of fall and winter mast that LPCs feed on (Van Pelt, 2013). Finally, LPCs are known to collide with objects placed in their habitat, including livestock fencing, and although fences are considered relatively minor sources of mortality to the LPC population, they may be locally significant (Wolfe and others, 2007).

Altered Fire Regime

Effects of fire suppression, and the interactive effect of fire suppression and grazing, are poorly understood in the central prairies of North America (Engle and Bidwell, 2001). Overall, however, it is believed that fire suppression has led to increased densities of shrublands and significant expansion of both native and nonnative species, such as eastern redcedar (*Juniperus virginiana*) and mesquite (*Prosopis* spp.). In some regions, these expanding species have completely replaced the native dwarf shrubland–grassland habitats used by LPCs (Fuhlendorf and others, 2002; Hagen and others, 2004). Fire suppression can lead not only to increased shrub densities, it can allow fine fuels to accumulate, which may promote larger, hotter fires from which shrubs (that is, suitable LPC habitat) require more time to recover.

Invasive Species

Several nonnative and native species have attained elevated densities on the landscape because of anthropogenic activities, including cultivation and CAs discussed herein. Expansion of woody plant species, particularly eastern redcedar, osage orange (*Maclura pomifera*), and honey mesquite (*P. glandulosa*), is a concern in the LPC's range because the resulting woodlands fragment and even may replace native dwarf shrubland–grassland habitats (Hagen and others, 2004). An increase in tree-dominated landscapes was associated with declining LPC populations in Texas and Oklahoma (Fuhlendorf and others, 2002). In Oklahoma alone, the estimated annual expansion of the nonnative eastern redcedar is 113,000 ha (Pieper, 2005). Eastern redcedar and osage orange are a larger problem in the eastern SGP, whereas mesquite is more problematic in the southwestern SGP (Johnson and others, 2006). Historically, there also has been some concern about competition between LPCs and the nonnative ring-necked pheasant (*Phasianus colchicus*), but the rangewide LPC conservation plan indicates that pheasant competition is not a conservation issue for LPCs (Van Pelt, 2013).

Insects and Disease

No population-threatening insects or diseases have been reported for the LPC (Hagen and others, 2004). In Texas, however, a coronavirus infection (infectious bronchitis) was detected in LPCs, and 1.3 percent of the LPCs tested for West Nile virus have been positive (U.S. Fish and Wildlife Service, 2012). If these viruses were to become widespread, they could have significant negative effects on smaller, isolated populations of LPCs (Hagen and others, 2004).

Climate Change

Little is known about potential effects of climate change on LPC populations and species survival, but if temperatures increase and precipitation patterns change in the SGP region, LPC habitat and productivity could be affected negatively. Extreme heat events are projected to increase through the 21st century (Karl and others, 2009), and extreme heat increases adult and juvenile mortality rates, particularly where canopy cover is diminished by grazing, herbicide treatments, drought, and extensive wildfire (Patten and others, 2005). Above-average winter temperatures also are projected for 2050–2080 in portions of the SGP, and the La Niña–driven droughts usually associated with warmer winters lead to reduced survival rates of LPC nests, likely the result of reduced thermal cover and forage (Grisham and others, 2013). Although the efficacy of precipitation models being used to forecast future precipitation patterns is debated, the models project decreases in average rainfall throughout the SGP but with more intense rainfall when rain does occur (Karl and others, 2009). If intense rainfall occurs during the brood-rearing season, it can increase chick mortality (Fields and others, 2006). There also is some concern that increased temperatures, drought, late spring freezes, and storm timing/intensity could reduce the cover and distribution of shinnery oak (Van Pelt, 2013), which would diminish LPC thermal and escape cover, as well as an important fall-winter food source. Although the LPC is expanding into some of its historical range as well as some new areas in northwestern Kansas, where LPCs come into contact with the greater prairie-chicken (*Tympanuchus cupido*), there is no evidence that the greater prairie-chicken is having negative effects on LPCs. Furthermore, the rate of hybridization between LPCs and greater prairie-chickens is less than 1 percent and not currently considered a threat to LPC populations (Van Pelt, 2013). Further work on effects of climate change in the SGP is needed to better understand its potential effect on LPCs.

Rapid Ecoregional Assessment Components

A general conceptual model for the key ecological attributes and CAs affecting the lesser prairie-chicken is illustrated in figure 27. Key ecological attributes and CAs identified in the REA are enumerated in tables 30 and 31.

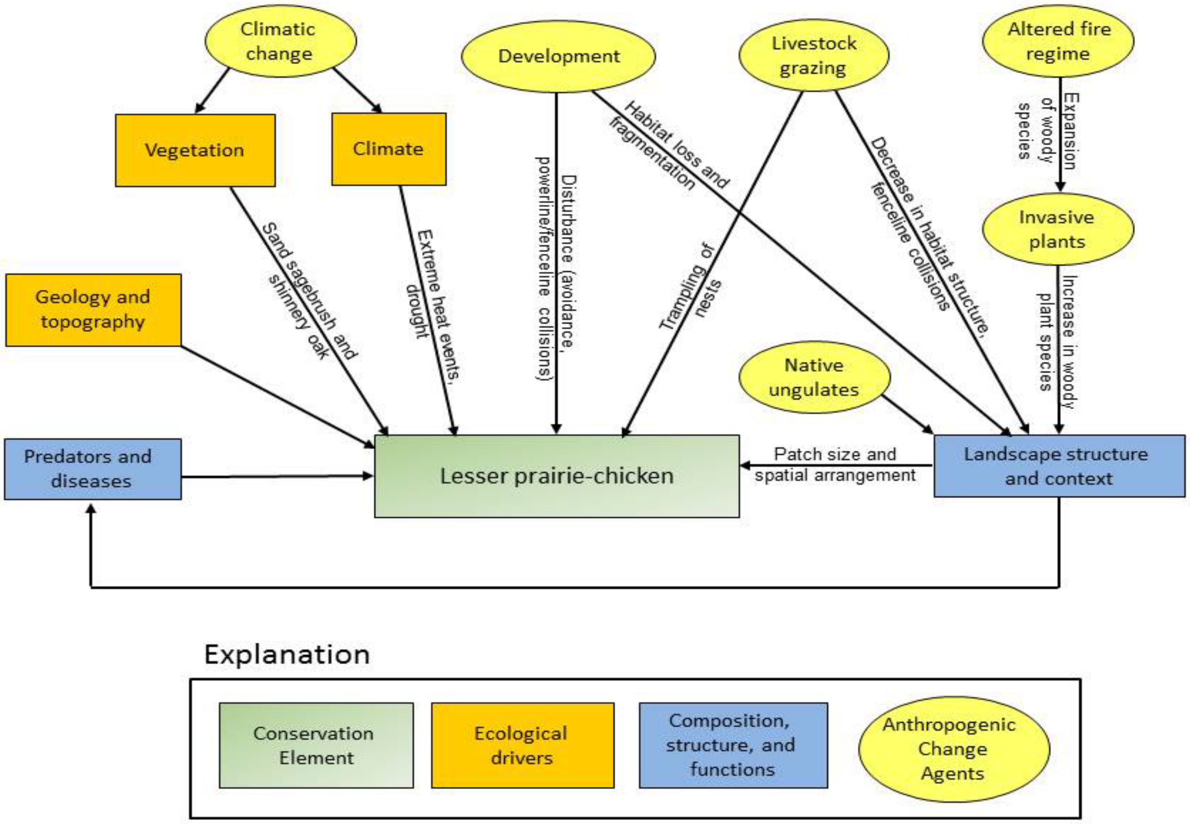

Figure 27. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for the lesser prairie-chicken (*Tympanuchus pallidicinctus*) in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of lesser prairie-chicken populations and habtiats are shown in orange rectangles (see also table 30); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 31).

Table 30. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for the lesser prairie-chicken (*Tympanuchus pallidicinctus*).

Attribute	Variables
Amount and distribution	Lesser prairie-chicken distribution, cover type distribution (nesting, brood-rearing, winter), forage quality, lek location.
Landscape structure	Size and spatial distribution of habitat (patch size [including sand sagebrush or shinnery oak], landscape heterogeneity, connectivity).
Landscape dynamics	Time since fire, fire size and occurrence, habitat productivity (for cover and food resources), drought, shrubland-grassland ecotone dynamics, food dynamics such as insect (prey) outbreaks (mast years).
Associated species management of concern	Dunes sagebrush lizard (due to shared habitat of shinnery oak).

Table 31. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for the lesser prairie-chicken (*Tympanuchus pallidicinctus*).

Attribute	Variables
Development (energy and infrastructure)	Habitat alteration (direct loss of habitat), habitat fragmentation and reduction in habitat connectivity, avoidance, collisions (powerlines).
Development (agricultural activities)	Habitat alteration (direct loss of habitat, decrease in habitat structure for nesting and cover, use of herbicide to reduce shrub cover and promote graminoids), habitat fragmentation and isolation (reduction in habitat connectivity), collisions (fencelines), habitat quality (small grain crops provide foraging and habitat), heavy grazing (diminish vertical escape and thermal cover, trampled nests).
Altered fire regime	Fire suppression results in habitat alteration (expansion of woody species, decrease in landscape heterogeneity), reduction in forage quality (decreased productivity of grasses and forbs).
Introduced insects and species	Habitat alteration (increase in woody plants, such as mesquite, eastern redcedar, and osage orange).
Insects and disease	Presence and effect on population (coronavirus infection—only detected in Texas at present, West Nile virus infection).
Climate change	Animal mortality (extreme heat events, reduced cover), habitat condition (reduction in amount and quality of food resources [insects and vegetation] due to drought).

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Chapter 16. Long-Billed Curlew

By Susan K. Skagen

Key Ecological Attributes

Distribution and Ecology

Long-billed curlews (*Numenius americanus*) (LBC) are patchily distributed across their current breeding range, which spans 16 U.S. states and three Canadian provinces. Endemic to the Great Plains, breeding LBCs inhabit shortgrass and mixed-grass prairies from eastern New Mexico north to south-central Saskatchewan, extending east into north-central Nebraska. The western edge of the range is outlined by the Cascade Range of northern California to southern British Columbia (Dugger and Dugger, 2002; Fellows and Jones, 2009). Originally, the LBC's range extended east to southern Manitoba and Wisconsin (Russell, 2006), but the historical breeding distribution has contracted by about 30 percent (Fellows and Jones, 2009). Within the SGP ecoregion, LBCs currently breed in eastern Colorado, eastern and occasionally northwestern New Mexico, and the very western edges of the Oklahoma and Texas panhandles (fig. 28). Their winter range falls almost entirely outside of the SGP, covering much of Mexico and extending along the coastal Baja peninsula, the Pacific coast and Imperial Valley of California, and the Gulf Coast of Texas (Dugger and Dugger, 2002).

Long-billed curlews were easily exploited by market hunting in the late 1800s and were vulnerable to habitat loss as native prairie landscapes were converted to agricultural fields, both drivers contributing to population declines across the past decades (Jones and others, 2008). A statistically rigorous rangewide survey conducted in 2004−2005 yielded an average estimate of 161,181 individuals across 16 western states and three Canadian provinces (Stanley and Skagen, 2007; Jones and others, 2008; Fellows and Jones, 2009), although prior population estimates based on peak counts in concentration areas were considerably lower. Based on the greater precision of the 2005 count, the official population size, as of 2012, has been designated at 140,000 (about 90 percent; range = 98,000–198,000; Andres and others, 2012). According to the North American Breeding Bird Survey, the surveywide trend estimate for the long-billed curlew from 1966 to 2012 suggests a stable population (0.20 percent per year), but it has been suggested that historical declines that began in the 1800s in the eastern extent of the range are slowly continuing (Dugger and Dugger, 2002). Trends within the SGP indicate significant population declines in Colorado (–4.8 percent per year, 95 percent confidence interval –7.7, –2.0), although not in New Mexico (–0.0 percent per year, 95 percent confidence interval –2.5, 2.3) (Sauer and others, 2014).

The LBC was designated as a Species of Greatest Conservation Need in the Comprehensive Wildlife Conservation Strategy plans of Colorado, New Mexico, and Oklahoma (Colorado Division of Wildlife, 2006; New Mexico Department of Game and Fish, 2006; Fellows and Jones, 2009). It was listed as highly imperiled in North America in early versions of shorebird conservation plans of both the United States and Canada (Donaldson and others, 2000; (B. Andres, U.S. Fish and Wildlife Service, oral commun., 2013). Despite recent population estimates that indicate a larger rangewide population size than earlier estimates, the status of concern continues to incorporate the range contraction of earlier decades.

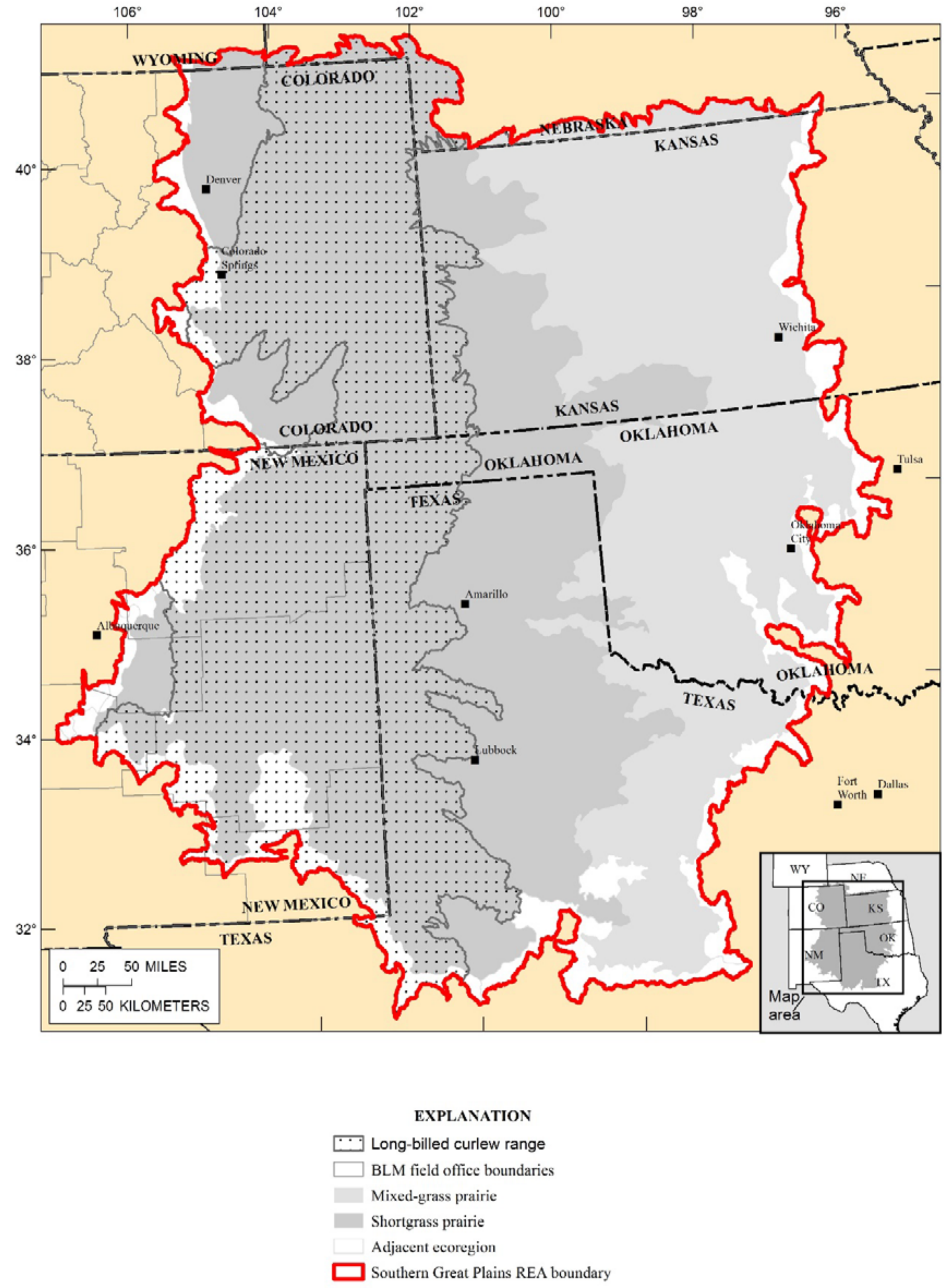

Figure 28. Current range of the long-billed curlew (*Numenius americanus*) in the Southern Great Plains ecoregion (data source: U.S. Geological Survey National GAP Analysis Program). (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (BLM, Bureau of Land Management; REA, Rapid Ecoregional Assessment)

The largest of North American shorebirds, the carnivorous LBC is also well adapted for capturing burrowing prey items, such as earthworms, shrimp, and crabs. They probe deeply in mud substrates for benthic invertebrates and marine crustaceans in tidal wintering areas where their diet is fairly specialized. On the breeding grounds, they use pecking and probing techniques to capture terrestrial insects, such as grasshoppers, carabid beetles (Carabidae), hemipterans, and lepidopteran larvae, as well as arachnids and small vertebrates (Redmond and Jenni, 1985; Dugger and Dugger, 2002). Predation on grassland bird eggs and nestlings by LBCs is not uncommon (Goater and Bush, 1986; Dugger and Dugger, 2002).

During the breeding season, LBCs forage and nest in native grasslands, pastures, haylands, and agricultural fields within flat or rolling topography (Dugger and Dugger, 2002; Dechant and others, 2003; Saalfeld and others, 2010). They tend to prefer habitats with low vertical profiles (4–15 cm) and avoid areas with high densities of shrubs, trees, and evergreen forests (Pampush and Anthony, 1993; Dugger and Dugger, 2002; Saalfeld and others, 2010). Long-billed curlews build nests in a variety of grassland plant communities across their range and likely select nest sites based on vegetation structure rather than preferences for specific plant communities (Jenni and others, 1981; Dugger and Dugger, 2002). Although nests in the northern part of their range are often found in grasslands dominated by cheatgrass (*Bromus tectorum*), in Colorado, Nebraska, and Wyoming they tend to nest in grasslands dominated by shortgrasses, such as buffalograss (*Bouteloua dactyloides*) and blue grama grass (*Bouteloua gracilis*), and medium-height grasses, such as needle-and-thread (*Hesperostipa comata*), sixweeks fescue (*Vulpia octoflora*), and several species of bluestem (*Andropogon* spp.). Some authors have generalized that LBCs nest "in the simplest, most open habitat available" (Dechant and others, 2003; Fellows and Jones, 2009). In Alberta, Canada, LBCs nested in active croplands of fall-seeded winter wheat and spring-seeded barley (Devries and others, 2010), and in the Great Basin, LBCs selected agricultural fields of wheat stubble, fallow fields, and cultivated hay fields dominated by timothy (*Phleum pratense*) and redtop (*Agrostis* spp.) for nesting and foraging (Cochrane and Anderson, 1987). Although habitat associations have been extensively documented throughout the breeding range of the LBC (Dechant and others, 2003; Fellows and Jones, 2009), the value of proximity to water has not been clearly defined. Habitat models by Saalfeld and others (2010), however, suggest the potential importance of emergent wetlands in the vicinity of nest sites.

Nesting is initiated in early to mid-April. Both sexes scrape a shallow depression on the ground, then the female lines the nest bowl with small pebbles, bark, grass, cattle and goose droppings, and (or) other materials (Dugger and Dugger, 2002). Clutch size is usually four eggs, occasionally three (Dugger and Dugger, 2002), and normally each pair raises only one brood per year. Brood rearing extends from mid-May to late July. Curlews exhibit a degree of site fidelity, occupying sites and territories year after year (Dechant and others, 2003). Reported nest success from a small number of studies across their distribution ranges from 20 percent to more than 65 percent (Dugger and Dugger, 2002). Egg and chick loss has been attributed to predation by mammals, including canids, badgers, and weasels, as well as raptors; to a lesser extent, nesting failure is caused when trampled by livestock and bison (Redmond and Jenni, 1986; Clarke, 2006). Mortality of recently hatched chicks (0–5 days old) can be high, especially if chicks are exposed to heat stress and parental feeding behavior is altered by human disturbance (Redmond and Jenni, 1986), although mortality among recently hatched chicks of precocial (feathered, able to walk and forage when hatched) species or recently fledged chicks of altricial (unfeathered, unable to leave the nest or feed themselves) species is common across a broad range of avian taxa.

In addition to providing breeding areas for LBCs, grassland habitats within the SGP also provide habitat for migrating curlews that breed in Montana and winter primarily in Mexico (Page and others, 2014). Fourteen southbound birds equipped with solar-powered satellite transmitters in breeding areas in Montana stopped to rest and feed in eastern Colorado for varying durations (1 day to 2 weeks) and in the western panhandle of Texas for as long as 1–3 months before making their final journeys to southern wintering areas. Median (and maximum) duration of stopovers of Montana breeders was 7.5 (108.3; $n = 37$) days during fall migration and 8.8 (39.8; $n = 37$) days in the spring. During migration stopovers, curlews use a variety of habitats, including shortgrass prairie, sparsely vegetated playas and shallow wetlands, and fallow or harvested agricultural fields. Migration across the SGP can occur from late June to late September in the fall and from late March to late April in the spring.

Landscape Structure and Dynamics

Throughout their breeding range, LBCs occur in flat or rolling topography covered by native grasslands, pastures, haylands, and agricultural fields (Dugger and Dugger, 2002). Although primarily associated with grassland habitats, in landscapes with little (0–5 percent) grass cover, LBC occurrence is more closely associated with cropland and hayland, avoiding shrublands and forests (Saalfeld and others, 2010). In suitable grassland habitats, density estimates range from 5 to 7 males per 100 ha (Redmond and Jenni, 1986) and are positively correlated with area of grassland (Dechant and others, 2003).

Wildfire, drought, and herbivory, including outbreaks of insects such as grasshoppers and locusts (Belovsky and others, 2000), are the ecological drivers that have set conditions for the short-statured grasslands preferred by curlews throughout their breeding range. Indeed, these disturbances can have positive effects on habitat suitability for LBCs (Dugger and Dugger, 2002), presumably as early as one year after disturbance. Although LBCs nest in short vegetation, they can forage in a range of vegetation heights 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 can mimic the grassland structure they prefer and attract pairs for nesting and brood rearing, and prairie dog (*Cynomys* spp.) colonies attract curlews for foraging.

Minimum block size recommendations for nesting and brood-rearing habitat have not been formalized, although a recommendation of three times the territory size has been recommended to account for an unoccupied buffer of 300–500 m around the edge of suitable habitat (Redmond and others, 1981; Dechant and others, 2003). Long-billed curlew territories surrounding nests average 14 ha in size and range from 6 to 20 ha. After hatching, adults and their broods leave the nesting site and move into more diverse habitats, some traveling as far as 6.5 km within a week. Home-range sizes, as reflected by movements during brood rearing, range from 0.5 to 4.8 km^2 (Clarke, 2006; Fellows and Jones, 2009), although home ranges are larger during drought.

Associated Species of Management Concern

This species is not considered a good indicator or umbrella species to represent other grassland birds or shorebirds, either because ranges do not overlap substantially or microhabitat needs differ (Fellows and Jones, 2009).

Change Agents

Major identified threats to LBC populations across their range include habitat loss and fragmentation, conversion of native grassland to agricultural lands, altered fire regimes, invasive plant species that alter grasslands, and energy and urban development (Jones and others, 2003; Fellows and Jones, 2009).

Development

Energy and Infrastructure

The potential effects of energy development, including oil, natural gas, and wind, on avifauna across the entire breeding range of the LBC has been explored for few species. Although not specifically addressed for the LBC, effects of energy development have been examined for lesser prairie-chicken in Kansas (Hagen and others, 2011) and for greater sage-grouse (*Centrocercus urophasianus*) and songbirds in Wyoming (Holloran, 2005; Gilbert and Chalfoun, 2011; Naugle and others, 2011). In general, birds tend to avoid infrastructure related to energy development. In Wyoming, drilling rigs and producing wells with their associated infrastructure and disturbance, including roads, traffic, and noise, generally were avoided by displaying male and nesting female greater sage-grouse. Similarly, lesser prairie-chickens, Brewer's sparrows (*Spizella breweri*), sage sparrows (*Amphispiza belli*), and vesper sparrows (*Pooecetes gramineus*) avoided energy-related infrastructure. Additional negative effects on avifauna, including LBCs, include potential collisions with powerlines and increased chick predation by raptors and corvids using power lines as perching structures (Naugle and others, 2011).

Development of wind power is occurring throughout the LBC breeding range and will undoubtedly increase its footprint in coming decades. Potential direct and indirect threats to curlews include strikes from rotor blades or collisions with towers, disturbance from human activity in the wind farms, and loss, fragmentation, and degradation of habitat associated with the development. A worldwide meta-analysis revealed the avian order Charadriiformes (including the genus *Numenius* [curlews]) as particularly vulnerable to negative effects of wind farms, especially when birds are aggregated in coastal areas (Stewart and others, 2007). In Europe, breeding abundance of the Eurasian curlew (*Numenius arquata*), which nests in habitats similar to those used by LBCs, decreased in the vicinity of a wind farm development, possibly because of avoidance by curlews (Pearce-Higgins and others, 2009).

Agricultural Activities

Long-billed curlews nest in both native and tame habitats of short stature, including crop fields and haylands, and tend to avoid tall vegetation, shrubs, and trees (Dugger and Dugger, 2002; Dechant and others, 2003). While rearing broods, curlews often move into taller vegetation that provides added shade and protection from predation. Haylands are used by LBCs across much of their range, but haying and mechanical disturbance can damage active nests (Cochran and Anderson, 1987; Dechant and others, 2003). During the nonbreeding season, curlews also forage in alfalfa, rice fields, and irrigated pastures. Between 2006 and 2011, increased demand for biofuels led to an accelerated conversion of native prairie to corn and soy production in the western corn belt (Wright and Wimberly, 2013). Although the corn belt is primarily east and north of the curlew's breeding range, the distributions do intersect within the SGP, suggesting that loss of grassland habitats through biofuel production may threaten curlew habitats within the SGP ecoregion.

Because grazing can produce the shorter grassland vegetation structure preferred by LBCs for nesting, it often is positively associated with bird presence. Domestic livestock can be used as "ecosystem engineers" within the context of heterogeneity-based management practices. By varying the intensity of grazing within and across pastures, providing supplemental food, and using patch burning techniques, managers can provide landscapes with varying grassland structure important to the full suite of prairie birds, especially for curlews across their entire breeding cycle (Derner and others, 2009). Very heavy grazing by cattle or bison, however, can result in destruction of eggs by trampling (Redmond and Jenni, 1986; Dugger and Dugger, 2002; Clarke, 2006).

General Anthropogenic Effects

Human disturbance due to excessive vehicle traffic, including off-road vehicles, and recreational use of breeding areas can cause nest abandonment and can disrupt essential brood-rearing behaviors of feeding and shading young (Fellows and Jones, 2009). Urbanization can encroach on grassland habitats, rendering them unsuitable for nesting and brood rearing through direct habitat loss, fragmentation, increased road density, and accompanying human disturbance.

Altered Fire Regimes

Along with drought and grazing, fire is considered a primary ecological driver responsible for maintaining grassland ecosystems and influencing the evolution of biota across the Great Plains (Knopf and Samson, 1997). Although fire regimes have been altered through suppression of wildfires and the advent of prescribed burning to enhance forage quality for cattle, the effects of these altered regimes on LBCs are only conjecture. Both wildfire and prescribed fire can temporarily alter grassland vegetation structure, resulting in short, sparse grass cover and increased habitat suitability for LBCs (Dugger and Dugger, 2002). Substantially shorter vegetation structure, however, can lead to increased predation on LBC eggs and chicks because of reduced concealment. Fire suppression that leads to greater vegetative structure in grasslands, including encroachment of shrubs and (or) woodlands, can reduce the overall amount of available habitat for LBCs.

Invasive Species

Because of their tall, dense structure, exotic crested wheatgrass (*Agropyron cristatum*) and knapweed (*Centaurea* spp.) can reduce habitat quality for nesting LBCs. In contrast, the shorter, sparser invasive cheatgrass (*Bromus tectorum*) appears to provide suitable nesting substrate (Dugger and Dugger, 2002), although its widespread distribution has altered the fire patterns and launched a series of ecosystem changes in the American West (Young and others, 1987). Across the sagebrush region, the easily ignited and fine-textured cheatgrass has replaced native bunchgrasses (the discontinuous cover of which did not carry fire through the ecosystem).

Insects and Disease

Aspergillosis, a pulmonary infection caused by fungi of the genus *Aspergillus*, was deemed responsible for nearly 15 percent of chick loss in 1978 (Redmond and Jenni, 1986). *Aspergillus* spores are naturally found in the environment and can cause disease symptoms worldwide in humans and in free-ranging and domesticated birds and mammals, although it is not contagious. No other diseases have been reported in LBCs, although ectoparasites and endoparasites, such as lice and intestinal helminths, cestodes, and trematodes, have been noted as uncommon (Dugger and Dugger, 2002). A potential but unstudied threat to curlew health is exposure to pesticides used for spraying grasshoppers throughout the breeding range (Fellows and Jones, 2009).

Climate Change

Climate predictions suggest higher temperatures in both summer and winter within the SGP ecoregion and across the entire breeding range of the LBC (Kirtman and others, 2013). Potential changes in precipitation are less clear, but decreases in rainfall are likely in more southern areas in the coming decades. Because the water cycle is strongly influenced by temperature, projected evaporation will increase, and soil moisture and relative humidity will decline. Future droughts within the range of those experienced in recent decades may favor LBC productivity. Fledgling production of LBCs has been greater in drought years and lower in abnormally wet years (Redmond and Jenni, 1986), possibly because of increased availability of short-statured grassland habitat for foraging and nesting. Additionally, in wetter years, avian predators may shift diets to curlew chicks because lush vegetation provides more cover for small mammals, their primary food resource (Redmond and Jenni, 1986). On the other hand, high temperatures coupled with a lack of shade can result in mortality of newly hatched birds when their thermoregulatory systems are not yet fully developed. Future work on the potential effects of climate change on the demography of LBCs is needed.

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting the long-billed curlew is illustrated in figure 29. Ecological attributes and CAs identified in the Rapid Ecoregional Assessment are enumerated in tables 32 and 33.

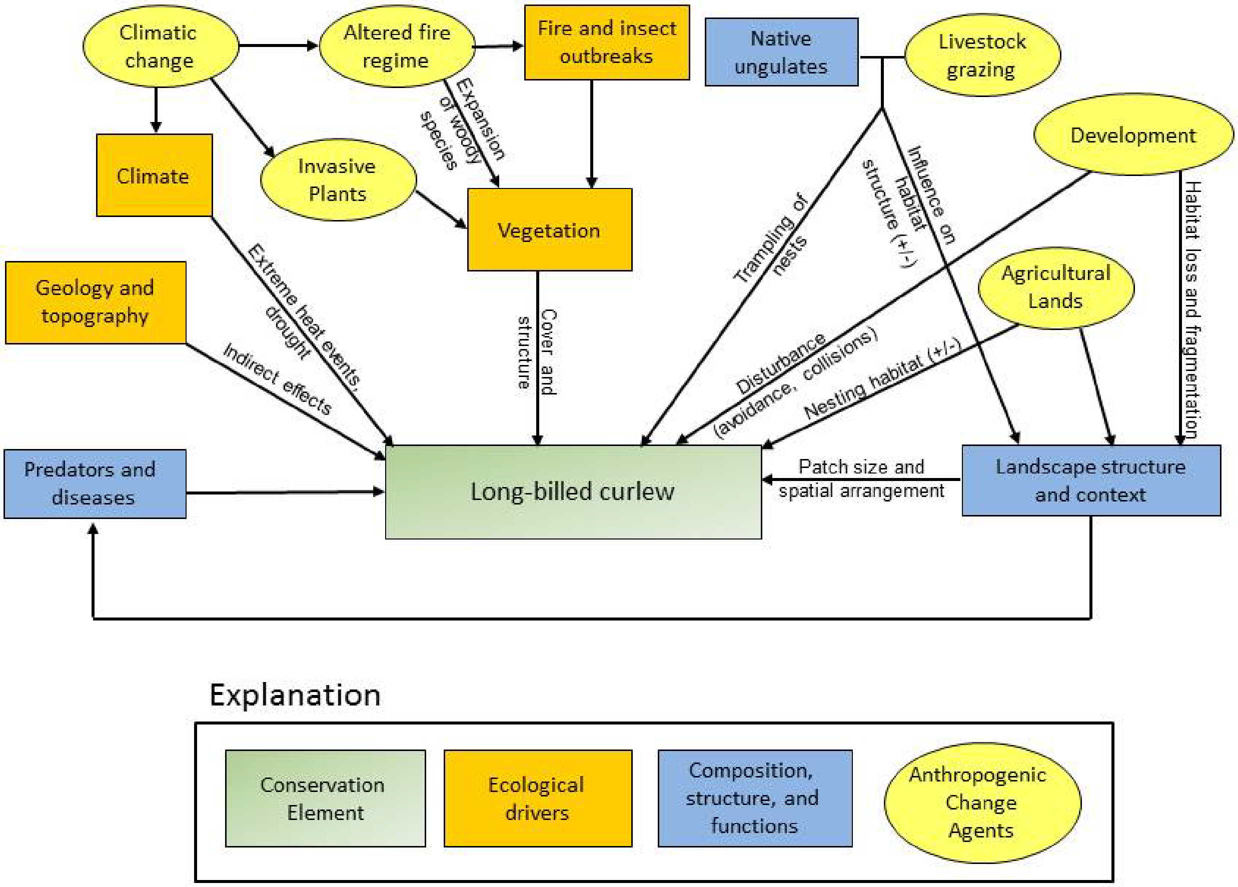

Figure 29. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for the long-billed curlew (*Numenius americanus*) in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of long-billed curlew populations and habtiats are shown in orange rectangles (see also table 32); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 33).

Table 32. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for the long-billed curlew (*Numenius americanus*).

Attribute	Variables
Amount and distribution	Long-billed curlew distribution.
Landscape structure	Size and spatial distribution of breeding and migratory habitat (patch size, connectivity).
Landscape dynamics	Habitat cover and productivity (food resources), drought, fire, insect outbreaks.

Table 33. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for the long-billed curlew (*Numenius americanus*).

Attribute	Variables
Development (energy and infrastructure)	Habitat alteration (direct loss of habitat), habitat fragmentation, avoidance, collisions (infrastructure), predation.
Development (agricultural activities)	Habitat alteration (direct loss of habitat due to cultivation of corn and soy [-], maintenance of nesting and brood-rearing habitat in haylands [+]), nest destruction via tilling (-), livestock grazing (varied intensity = increase heterogeneity [+], non-varying intensity = loss of habitat heterogeneity [-]).
Development (human effects)	Habitat alteration (direct loss and disturbance of breeding habitat), habitat fragmentation.
Altered fire regime	Fire suppression results in habitat alteration (decrease in landscape heterogeneity, expansion of woody species).
Invasive species	Habitat alteration (nonnative grasses that increase vegetation cover and height [-], short, sparse nonnative grasses [+] [but such grasses have cascading effects on altered fire regime]).
Climate change	Predicted increase in extreme heat events and drier conditions (this species could be affected less by these predicted conditions compared to other species).

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Chapter 17. Mountain Plover

By Colin Woolley

Key Ecological Attributes

Distribution and Ecology

The mountain plover (*Charadrius montanus*) is a migratory shorebird and native breeder on arid shortgrass prairie habitat in the western Great Plains. Current breeding range extends from southern Saskatchewan to New Mexico and northern Texas, with breeding records as far south as Nuevo Leon, Mexico (Knopf and Wunder, 2006). This range includes portions of four Rapid Ecoregional Assessment project areas: Northwestern Plains, Wyoming Basin, Southern Great Plains (fig. 30), and Chihuahuan Desert. Formerly, mountain plover breeding range extended farther east to include western portions of North Dakota, South Dakota, and Kansas (Knopf and Wunder, 2006).

Wintering range for this migratory species includes the Central Valley in California and extends east through southern Arizona and northern Mexico to south Texas. The majority of the population winters in California in the Sacramento, San Joaquin, and Imperial Valleys, with flocks of over 500 individuals documented on agricultural fields of alfalfa (*Medicago sativa*) and Bermudagrass (*Cynodon dactylon*) (Wunder and Knopf, 2003). Wintering plovers were formerly common on the coastal plains of California, but Christmas Bird Count data for years 1950–2000 indicate that wintering populations have shifted away from coastal plains and into the Imperial Valley (Hunting and others, 2001; Wunder and Knopf, 2003). Analysis of Christmas Bird Count data from 1980 to 1997 by Hunting and others (2001) found that 95 percent of North American plover sightings during winter were located in California. Plover distribution across their wintering range outside of California is poorly known (Knopf and Wunder, 2006).

Mountain plovers typically arrive on breeding grounds in late March through April. Within suitable habitat, plovers prefer to nest where there is little to no slope (Graul, 1975). In areas with rolling hills, they avoid nesting where slope exceeds five degrees (Graul, 1975). Knopf and Miller (1994) suggest 30 percent bare ground as a minimum habitat requirement for both breeding and wintering grounds. The nest itself is a scrape on the ground, typically lined with locally available leaf litter or small pebbles. Mountain plovers use a split-clutch mating system, in which the female lays typically three eggs in each of two nests (Graul, 1975; Knopf and Wunder, 2006). The first nest is incubated by the male while the female then lays and incubates a second nest. Care of broods is uniparental in this precocial species. Recently hatched broods typically leave the vicinity of the nest within 24 hours and stay with the adult until fledging after 33–34 days (Graul, 1975; Knopf and Wunder, 2006). Survival rates for nests tended by males are higher than for those tended by females (Dinsmore and others, 2002). Brood survival, however, is higher in female-tended broods (Dinsmore and Knopf, 2005). The contributions of each sex to overall productivity are likely similar (Dinsmore and Knopf, 2005).

Migration timings are well-documented, though the migration routes are still poorly understood. Hypotheses relating to migration routes are speculative, based on disparate and infrequent sightings during the periods of fall and spring migration (Knopf and Wunder, 2006).

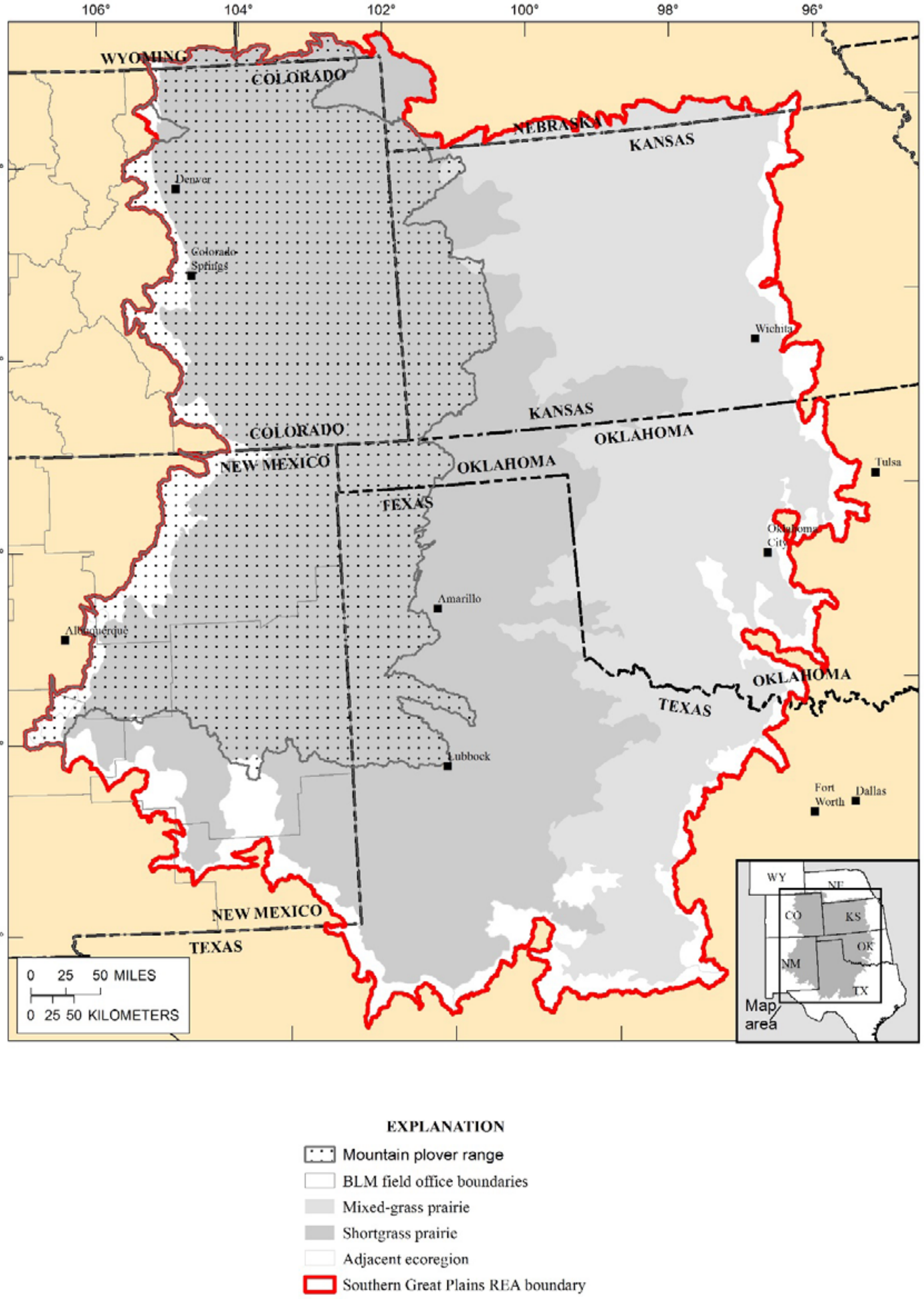

Figure 30. Current range (as of 2011) of mountain plover (*Charadrius montanus*) in the Southern Great Plains ecoregion (data source: U.S. Geological Survey National GAP Analysis Program). (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (BLM, Bureau of Land Management; REA, Rapid Ecoregional Assessment)

Postbreeding flocks begin to form on breeding grounds as early as late June and may persist as late as October before migrating (Knopf and Wunder, 2006). Fall migration routes may follow the east flank of the Rocky Mountains moving south, before heading west across New Mexico and Arizona and into California. Spring migration is likely the same route in reverse but could involve an alternate route over the Sierra Nevada and Great Basin region and across the Rocky Mountains to breeding grounds, though there is little evidence to support this route (Knopf and Wunder, 2006). Recent and ongoing studies using light level loggers and (or) global positioning system tags attached to migrating plovers will further develop our understanding of plover migration routes in the near future.

Mountain plovers exhibit high breeding site fidelity, with adults often returning to the same breeding grounds in consecutive years (Knopf and Wunder, 2006). Additionally, some plovers have been documented returning to breed at their natal sites (Dinsmore and others, 2003). These traits suggest the possibility of genetic differentiation among breeding locales. Despite this, a study of plover population genetics (Oyler-McCance and others, 2008) found overall high genetic diversity within and among sites, with no significant differentiation between breeding sites, which suggest that mountain plovers consist of one large gene pool rather than multiple disjointed breeding populations.

Diet consists primarily of terrestrial invertebrates, especially beetles (Coleoptera), ants (Hymenoptera) and crickets (Orthoptera) (Knopf and Wunder, 2006; Knopf, 2008). These prey are opportunistically foraged from the ground and consumed immediately (Knopf and Wunder, 2006). Chicks have been observed foraging for themselves just 24 hours after hatch (Graul, 1975).

Long-term population data gathered from Breeding Bird Surveys indicate that mountain plovers experienced a rangewide population decline averaging 3.15 percent per year from 1966 to 2012 (Sauer and others, 2014). The actual severity of the population decline is difficult to measure, however, because of the low detectability of mountain plovers and limitations of Breeding Bird Survey survey designs. Breeding Bird Survey survey routes are restricted to roads, which limits their application in estimating rangewide population densities (Dreitz and others, 2006). A reliable population estimate for mountain plovers has been difficult to determine. Even in suitable habitat, mountain plovers are relatively rare and often sparsely distributed. Additionally, much of their breeding range occurs on private land, and access can be difficult or impossible to obtain. Recent continental population estimates have varied from 11,000–14,000 (Plumb and others, 2005), to 15,000–20,000 (Tipton and others, 2009), to a minimum of 20,000 (U.S. Fish and Wildlife Service, 2011). The apparent increase in size of population estimates is likely due to limitations of earlier sampling methods and failures to account for smaller, localized breeding populations outside of known breeding areas rather than an actual increase in population numbers (U.S. Fish and Wildlife Service, 2011).

The mountain plover was proposed for listing as a threatened species under the Endangered Species Act in 1999, 2002, and again following litigation by conservation groups in 2009 (U.S. Fish and Wildlife Service, 2011). After a period of review and public comments, it was ultimately not listed, as there was insufficient evidence to suggest the species was “in danger of extinction, or likely to become endangered within the foreseeable future, throughout all or a significant portion of its range” (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).

Landscape Structure and Dynamics

Mountain plovers breed in sparsely vegetated areas and have been strongly associated with native grazers such as American bison (*Bison bison*), prairie dogs (*Cynomys* spp.), and pronghorn (*Antilocapra americana*) throughout their range (Knopf and Wunder, 2006; Goguen, 2012; Augustine and Baker, 2013). Eastern Colorado has long been considered a stronghold for breeding plovers (Graul and Webster, 1976; Knopf and Rupert, 1996; Knopf and Wunder, 2006). Shortgrass prairie in eastern Colorado is characterized primarily by two grasses, blue grama (*Bouteloua gracilis*) and buffalograss (*B. dactyloides*), often with sparsely dispersed pricklypear (*Opuntia* spp.) (Graul, 1975; Knopf and Wunder, 2006).

Park County, Colorado, supports a breeding population of mountain plovers estimated at 2,300 individuals (Wunder and others, 2003). Here, slimstem muhly (*Muhlenbergia filiculmis*) dominates the shortgrass prairie, and plovers can be found in areas with sparse shrub coverage of green rabbitbrush (*Chrysothamnus viscidiflorus*) and silver sagebrush (*Artemisia frigida*) (more commonly known as fringed sage or fringed sagewort) (Wunder and others, 2003). In the tablelands of Wyoming and Montana, plover breeding habitat contains sparsely distributed shrubs including greasewood (*Sarcobatus vermiculatus*), big sagebrush (*Artemisia tridentata*), pricklypear, and fourwing saltbush (*Atriplex canescens*) (Dinsmore and others, 2003; Plumb and others, 2005). Throughout their breeding range, mountain plovers are highly associated with recently disturbed areas (by grazers, fire, or agricultural practices) of low-growing vegetation with no or few shrubs (Knopf and Wunder, 2006). Given its preferred habitat, the common name of the mountain plover is considered a misnomer (Knopf and Miller, 1994); a more appropriate name would be “prairie” plover.

The role of fire in shaping grassland ecosystems has been well documented (see Brockway and others, 2002; Samson and others, 2004), though it is considered secondary to grazing as an ecological driver in the shortgrass prairie (Askins and others, 2007). Historically, fire influenced local grassland community composition and structure and increased heterogeneity in the shortgrass prairie (Brockway and others, 2002). However, fire likely had a lesser influence in shortgrass prairie relative to other grassland systems because of reduced fuel load limiting the ability of fire to spread. Blue grama and buffalograss maintain 90 percent of their biomass below the soil surface (Askins and others, 2007), while the above surface biomass was typically heavily grazed. Thus, the influence of fire in shortgrass prairie was very localized (Askins and others, 2007). Mountain plovers are quickly attracted to recently burned habitat on both wintering and breeding grounds, with birds seen moving into areas that are still smoldering (Knopf and Wunder, 2006). This phenomenon is probably explained by movements of local birds, but more study is needed to determine the provenance of these plovers. In a study of plover response to prescribed burns, Augustine (2011) found that plover densities on prairie dog colonies and recently burned rangeland were higher than on unburned rangeland.

Mountain plover will readily nest on agricultural fields, whether fallow or with low-growing crop (Knopf and Rupert, 1999; Shackford and others, 1999; Bly and others, 2008). The conservation implications of plovers nesting on agricultural fields are still being discussed; a review of this discussion is detailed below under “Agricultural Activities” in the “Change Agents” section. Home-range size during the brood-rearing period has been found to be similar across various habitat types. Dreitz and others (2005) found average home-range sizes of 131.6 ha (agricultural fields), 146.1 ha (rangeland), and 243.3 ha (prairie dog colonies) using 95 percent fixed-kernel estimates. Home-range size during other life history stages (such as during nest incubation or on wintering grounds) has not been studied.

Associated Species of Management Concern

Both mountain plovers and burrowing owls (Chapter 12) are strongly associated with black-tailed prairie dogs (Chapter 20) during the breeding season (Dinsmore and others, 2005; Augustine and others, 2008; Tipton and others, 2008; Augustine and Baker, 2013). Prairie dog colonies create large areas of grazed vegetation and bare ground, which benefit breeding plovers. Sylvatic plague has a strong influence on local black-tailed prairie dog distribution and abundance (Collinge and others, 2005), which in turn affects plovers. Mountain plover nesting density rapidly declines following a plague event on black-tailed prairie dog colonies (Augustine and others, 2008). Use of plover nesting habitat associated with prairie dog colonies closely follows the active portions of the colony. Burrowing owls also nest in higher density on active prairie dog colonies than on grasslands or agricultural fields (Tipton and others, 2009). In wet years, it is likely that plover use of prairie dog colonies increases in response to taller grass growth in the surrounding landscape (Andres and Stone, 2009). Prairie dog control programs are still implemented on both public and private land, and the resulting declines in prairie dog populations reduce availability of preferred mountain plover breeding habitat (Dinsmore and others, 2005). Management decisions that benefit black-tailed prairie dogs will likely benefit both mountain plovers and burrowing owls (Tipton and others, 2009).

Pronghorn and swift fox (*Vulpes velox*) (Chapter 22) are also species of management concern that use shortgrass prairie habitat. Swift foxes have been documented as a predator of mountain plover nests (Knopf and Wunder, 2006). In northern Colorado, plover productivity appears closely related to swift fox activity, with most reproductive losses due to fox predation (Knopf and Rupert, 1996). Swift fox reintroduction in other parts of the plover’s range may negatively affect plover reproduction (Knopf and Wunder, 2006).

Change Agents

Habitat loss and degradation on both the breeding and wintering grounds are thought of as the primary drivers of population declines in mountain plovers (Knopf, 1994; Knopf and Wunder, 2006). This includes conversion of shortgrass prairie to agricultural production, planting to taller nonnative grasses (for example, crested wheatgrass [*Agropyron cristatum*]) for livestock feed, and loss of wintering habitat due to development on the coastal plains of southern California. Additionally, landscape changes brought about through fire suppression and removal of native grazers such as bison and prairie dogs from the shortgrass prairie have reduced the availability of preferred plover breeding habitat.

Development

Energy and Infrastructure

A potential threat to mountain plovers is the habitat loss and fragmentation resulting from wind energy development and oil and gas production. The implications for mountain plovers are still poorly understood, but recent studies have suggested that mountain plovers may be little affected by oil and gas development (U.S. Fish and Wildlife Service, 2011). Mountain plovers fly low to the ground, thus collisions with wind turbines and utility lines are not expected to be a problem (Andres and Stone, 2009), and they are very tolerant of nearby heavy machinery such as tractors, road graders, and other large vehicles associated with agriculture or oil and gas development (Knopf and Wunder, 2006). While mountain plovers are probably locally displaced by active mining activities, they may actually benefit from the open, bare ground created by disturbance once active mining has ceased (U.S. Fish and Wildlife Service, 2011). Overall, more study is needed on the effect of energy development on plovers.

Agricultural Activities

Conversion of shortgrass prairie to agricultural crop production has greatly altered mountain plover habitat, especially in the eastern portion of their range (Knopf, 1994). Plovers are known to nest on agricultural fields, particularly fields that are fallow or with low-growing crop (Knopf and Rupert, 1999; Shackford and others, 1999). Plovers nesting on agricultural fields are subjected to loss of nests through tilling and other mechanical agricultural practices. This was thought to be a potential population sink (Knopf and Rupert, 1999) but has since been demonstrated to have little to no effect on overall plover nesting success (Dreitz and Knopf, 2007). Nest failure due to predation is much lower on agricultural fields compared to native rangeland, resulting in overall similar nest survival for both habitats (Dreitz and Knopf, 2007). Some nests on agricultural fields actually survive the tilling process (Dreitz and Knopf, 2007); if the eggs are unburied and intact, then the incubating adult will often return to the nest. Use of herbicides and pesticides on agricultural fields may negatively affect the invertebrate prey base of plovers on both breeding and wintering grounds, but this has not been well studied. Iko and others (2003) found no evidence of recent exposure to anticholinesterase pesticides in plovers wintering on agricultural fields in California.

Successful nest-marking programs have been implemented on private land in Colorado and Nebraska (Dreitz and Knopf, 2007; Bly and others, 2008), in which active nests are found and marked by biologists prior to mechanical treatment of a field. The landowner is then able to avoid tilling where the marked nest is located. Although these efforts are not expected to have a population-level effect for the plover, they have served as a model for interacting with private landowners and demonstrating landowner willingness to participate in species conservation efforts (Dreitz and Knopf, 2007).

Livestock grazing practices have led to greater homogeneity across shortgrass prairie habitat compared to grazing habits of native species such as bison, prairie dogs, and pronghorn (Derner and others, 2009). Grazing practices that increase heterogeneity (such as alternating between intense and light grazing or incorporating prescribed burns) will likely benefit mountain plovers (Derner and others, 2009). 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). Augustine and Derner (2012) found that heavy livestock grazing did not substitute adequately for the grazing services of black-tailed prairie dogs. In eastern Colorado, mountain plovers rarely occupy rangeland that lacks prairie dogs or recent fire (Augustine, 2011). Plover habitat preferences are at odds with grazing practices that maximize available livestock forage (Augustine and Derner, 2012). Thus, land managers must balance conservation goals with economic grazing concerns. Augustine and Derner (2012) suggest developing economic incentives that compensate for losses of livestock productivity related to prairie dogs as a way to support mountain plover habitat on private rangeland.

The Farm Service Agency of the U.S. Department of Agriculture administers the CRP, in which private land is pulled out of agricultural production and planted to both native and nonnative grasses to support wildlife habitat among other goals. Typically, however, CRP land is planted to taller, cool-season grass species that provide habitat for other breeding birds but are not used by mountain plovers (Knopf and Wunder, 2006). Management of CRP lands that incorporate prescribed burns may benefit mountain plovers by providing increased bare ground coverage in CRP-enrolled land.

Little is known about the effects of agriculture and livestock grazing on plover wintering grounds in northern Mexico.

Urbanization

Residential and commercial development in southern California over the last 150 years has replaced native coastal plains habitat on historical mountain plover wintering grounds. This is thought to be a contributing reason for the shift of wintering grounds onto agricultural fields of the Imperial, San Joaquin, and Sacramento Valleys (Wunder and Knopf, 2003). It is unclear, however, if plover preference for agricultural fields may also have played a role in this shift (Wunder and Knopf, 2003).

In shortgrass prairie of eastern Colorado, more native habitat is currently being converted to suburban development than to agricultural production (Andres and Stone, 2009). This urbanization, however, falls mainly to the west and north of mountain plover breeding distribution and thus is not considered a threat at this time (Andres and Stone, 2009). Plovers are highly tolerant of off-road vehicles (Knopf and Wunder, 2006) and will usually quietly flush from the nest to return later. An increase of recreational vehicle use near breeding grounds is not expected to be a concern, though eggs are subject to overheating if plovers are kept off the nest for prolonged periods of time (Graul, 1975).

Altered Fire Regime

Historically, fire increased heterogeneity in the shortgrass prairie by regulating the structure and community composition of localized areas (Brockway and others, 2002). Fire suppression, however, has nearly eliminated the influence of fire in shortgrass prairie. On public lands in the northern Great Plains, fire affects only about 0.5 percent of shortgrass prairie (Samson and others, 2004). The absence of fire has reduced landscape heterogeneity in the shortgrass prairie, although direct effects of fire suppression on mountain plover have not been analyzed.

Invasive Species

Nonnative grasses introduced in shortgrass prairie habitat tend to increase vegetation cover and height as well as soil moisture, all of which reduce habitat quality for mountain plovers (Andres and Stone, 2009). Sometimes CRP land is seeded with nonnative cool-season grasses such as crested wheatgrass (*Agropyron cristatum*), which precludes plover use (Knopf and Wunder, 2006). Overall, however, the effect of invasive plant species on mountain plovers has not been specifically studied (U.S. Fish and Wildlife Service, 2011). The effect of feral cats on mountain plover populations has not been studied, though plovers have been observed nesting within 200 m of houses with feral cats present.

Introduced Insects and Disease

Disease in mountain plovers has not been studied (Knopf and Wunder, 2006), but it does not appear to be a conservation concern. West Nile virus has not been documented in mountain plovers (Andres and Stone, 2009), though it has been detected in other bird species within the plover's range. Dinsmore (2013) found that mountain plover nest survival was negatively affected by exposure to insecticide treatments on the breeding grounds. In Montana, pyrethroid insecticides containing deltamethrin were applied to black-tailed prairie dog colonies to combat the spread of flea-borne sylvatic plague. Dinsmore (2013) found no evidence of direct toxicity to the plover, but nest survival in treatment areas declined likely as a result of reduced available food resources.

Climate Change

Current projections of climate change in shortgrass prairie predict an increase of extreme heat events and drier conditions in the summer (Karl and others, 2009; Patricola and Cook, 2012). The effect this may have on mountain plovers is still poorly understood but could favor creation of plover habitat by increasing bare ground and decreasing vegetation coverage (Dinsmore, 2008; Andres and Stone, 2009). Drought has long been considered an ecological driver of the shortgrass prairie ecosystem (Knopf, 1994), and mountain plover adult survival increases in drought conditions (Dinsmore, 2008).

Adult plovers actively shade eggs and chicks on hot days during the breeding season to keep them from overheating (Graul, 1975). Increasing daily temperatures could increase energetic demands on adults or lead to nest failure due to overheating. Overall, however, plover productivity appears to be higher in drought years (Wunder, 2007), suggesting again that plovers may benefit from hotter, drier weather predicted by climate change.

Climate change will also likely influence agricultural practices throughout the Great Plains which could change patterns of plover distribution, especially where they commonly nest on agricultural fields (Andres and Stone, 2009). These changes, however, are difficult to predict and require more study.

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting the mountain plover is illustrated in figure 31. Ecological attributes and CAs identified in the Rapid Ecoregional Assessment are enumerated in tables 34 and 35.

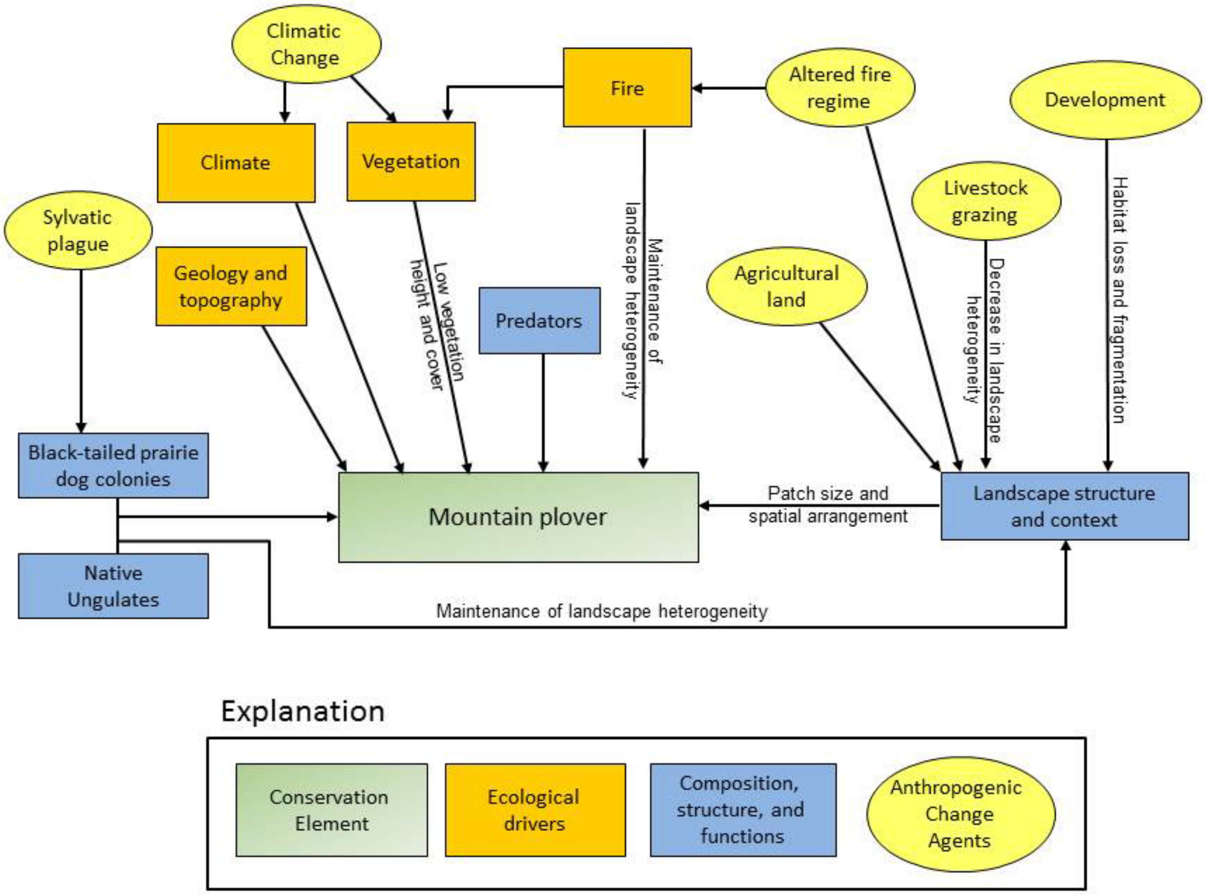

Figure 31. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for the mountain plover (*Charadrius montanus*) in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of mountain plover populations and habtiats are shown in orange rectangles (see also table 34); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 35).

Table 34. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for the mountain plover (*Charadrius montanus*).

Attribute	Variables
Amount and distribution	Mountain plover distribution.
Landscape structure	Size and spatial distribution of habitat (breeding and winter).
Landscape dynamics	Habitat productivity (food resources), drought, fire, predator dynamics.
Associated species management of concern	Black-tailed prairie dog, burrowing owl, swift fox, pronghorn.

Table 35. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for the mountain plover (*Charadrius montanus*).

Attribute	Variables
Development (energy and infrastructure)	Habitat alteration (direct loss of habitat), habitat fragmentation.
Development (agricultural activities)	Habitat alteration (direct loss of habitat due to cultivation [-], maintenance of nesting habitat [+]), nest destruction via tilling (-), livestock grazing (loss of habitat heterogeneity [-]), increase in Conservation Reserve Program lands (when planted with tall, cool-season grasses [-]).
Development (urbanization)	Habitat alteration (direct loss of habitat), habitat fragmentation.
Altered fire regime	Fire suppression results in habitat alteration (decrease in landscape heterogeneity, influence in shortgrass prairie very localized).
Invasive species	Habitat alteration (nonnative grasses increase vegetation cover and height [-], increase soil moisture [-]).
Introduced species and disease	Presence and effect of plague on black-tailed prairie dog population.
Climate change	Predicted increase in extreme heat events and drier conditions (this species could be affected less by these predicted conditions compared to other species).

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Chapter 18. Snowy Plover

By T. Luke George

Key Ecological Attributes

Distribution and Ecology

The snowy plover (*Charadrius nivosus*) is a small, pale-colored shorebird (family Charadriidae) with a broad breeding distribution that stretches from the west coast of the United States to the Caribbean islands. Despite its breeding distribution, the snowy plover is one of the rarest shorebirds in North America. A recent survey of the breeding population in continental North America yielded an estimate of 20,000−30,000 individuals (Thomas and others, 2012). The principal reason for the species' low numbers is its specialized breeding habitat requirements. These birds require large expanses of flat, unvegetated areas of sand or cobble adjacent to waterbodies that support productive arthropod populations (Page and others, 2009). Their breeding habitat, therefore, is limited to sandy coastal beaches, barrier islands, barren shores of inland saline lakes, ephemeral wetlands, and sand bars on large rivers. Within the SGP ecoregion, distribution of the snowy plover is closely tied to the distribution of playas and saline lakes, large reservoirs, and major rivers (fig. 32).

The SGP ecoregion includes one of the largest breeding colonies of snowy plovers in the world. In a comprehensive survey of snowy plover breeding populations in 2007 and 2008, more than 5,000 individuals were estimated to occur at Salt Plains NWR in Oklahoma, making it the second largest population surveyed and accounting for 20 percent of the entire breeding population in North America (Thomas and others, 2012). Other sites with large breeding populations (more than 200) within the SGP include Quivira NWR in Kansas and Cargill Salt Flat in Oklahoma. Small populations nest on the shorelines of saline lake, reservoirs, and some of the major rivers, bringing the total number for the SGP to 6,924 (with a 95 percent confidence interval of 5,297−8,560) (Thomas and others, 2012). The estimated number of breeding plovers in the region during a 2007–2008 survey (Thomas and others, 2012) was more than three times greater than the 2,105 birds estimated in 2006 (Morrison and others, 2006), although differences between the two survey methods alone could have accounted for this difference in estimates rather than any change in breeding population size (Thomas and others, 2012). The 2006 survey was based on a compilation of total birds observed at known sites, which likely underestimated the total population, whereas the latter survey was based on a combination of (1) complete surveys of small sites and (2) random sampling of suitable breeding habitats at large wetland complexes, adjusted for probability of detection and extrapolated across all unsurveyed wetland complexes.

Based on repeated surveys of known breeding locations, Morrison and others (2006) and Andres and others (2012) concluded that snowy plovers were declining across their range and in interior North America. The surveys compiled by Morrison and others (2006) and Andres and others (2012), however, showed little change in numbers over time, were not comprehensive, and did not include a correction for imperfect detection rates; thus, the conclusions are questionable. Snowy plovers are rarely detected on the Breeding Bird Survey, therefore estimates of population trends are not available from those data (Sauer and others, 2011). Sporadic surveys of populations in Kansas indicate that populations are relatively stable at saline lakes and

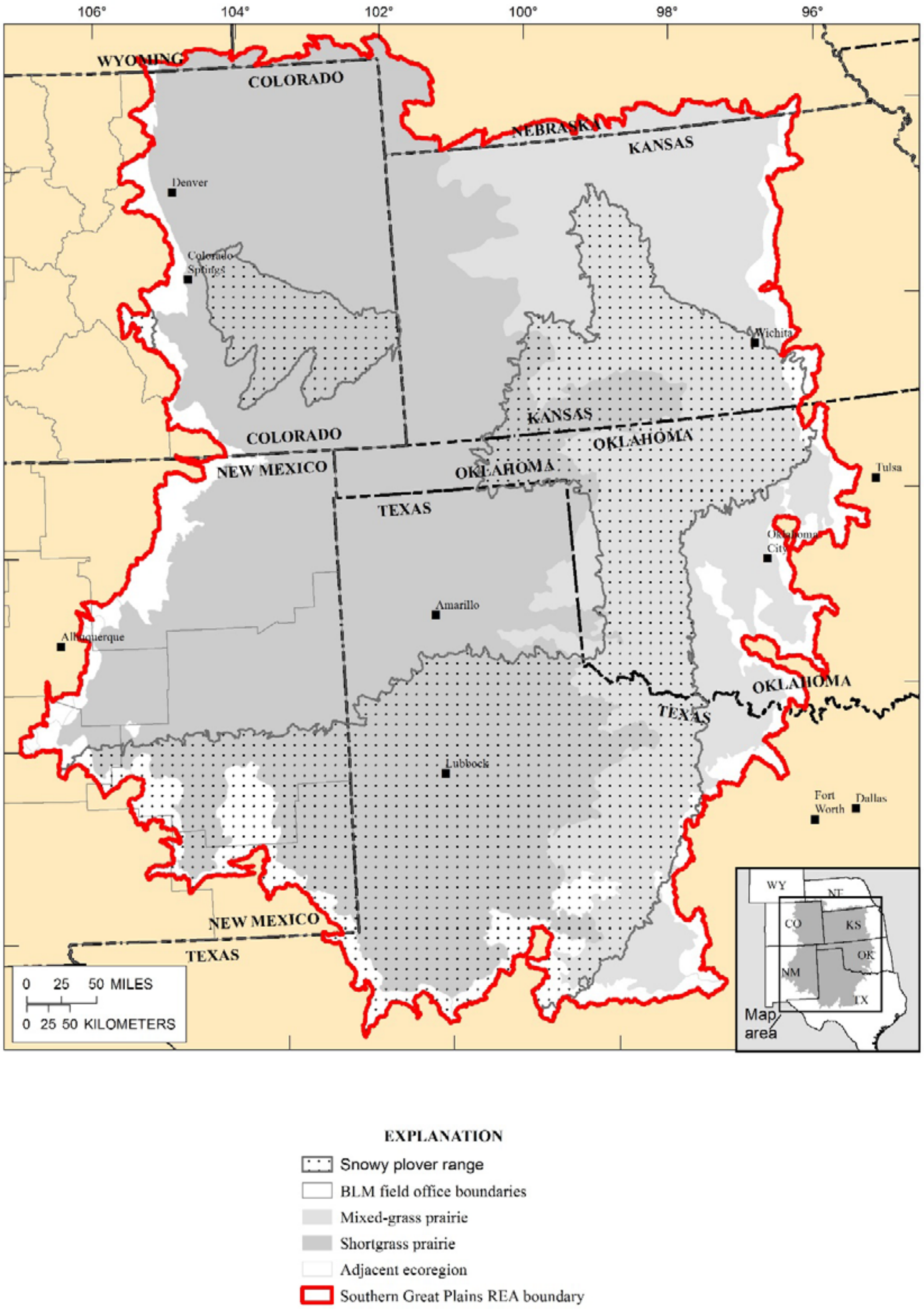

Figure 32. Current range (2002) of the snowy plover (*Charadrius nivosus*) in the Southern Great Plains ecoregion (data source: U.S. Geological Survey National GAP Analysis Program). (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (BLM, Bureau of Land Management; REA, Rapid Ecoregional Assessment)

reservoirs, but breeding populations in ephemeral wetlands and along rivers appear to be declining (Busby, 2002). No assessments of recent population trends have been compiled for other states in the SGP region. The Pacific coast population of the snowy plover was listed as threatened in 1993 because of continuing population declines and threats from human disturbance, habitat loss, and synanthropic predators (U.S. Fish and Wildlife Service, 1993). Within the plan area, the snowy plover is listed as a focal species by the U.S. Fish and Wildlife Service's Migratory Bird Program, 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.

Snowy plovers use a variety of breeding habitats, but the importance of different site types varies across the SGP ecoregion (Busby, 2002). In the Playa Lakes region of Texas, plovers bred almost exclusively around saline lakes and occasionally used riparian habitat along rivers, but they were not observed breeding at 106 ephemeral wetlands (playas) surveyed in 1998 and 1999 (Conway and others, 2005). Although plovers appear to strongly prefer saline lakes over ephemeral wetlands, they have been observed breeding at ephemeral wetlands in Kansas (Busby, 2002) and eastern Colorado (Mabee and Estelle, 2000). No systematic surveys of plover breeding populations at ephemeral wetlands and saline lakes have been attempted outside of the Playa Lakes region of Texas, however, so the degree to which plovers prefer saline lakes in other regions of the SGP is unclear.

Plovers arrive at their SGP breeding locations in late March or early April and generally begin laying eggs in late April or early May. They usually lay a clutch of three eggs, which is incubated by both the male and female for 25−28 days (Page and others, 2009). Young are attended by one or both adults until they are able to fly at 28−33 days. Although snowy plovers consistently fledge multiple broods in coastal locations, plovers generally fledge only one brood at breeding locations within the SGP (Page and others, 2009). In a study of plover nesting ecology in the playa lakes of region of Texas, however, plovers were observed producing multiple broods at saline lakes (Conway and others, 2005), which the authors attributed to the extended hydroperiod of those lakes relative to other nesting habitats.

Nesting success of snowy plovers within the SGP region is highly variable, ranging from 7.1 (Saalfeld and others, 2011) to 73.3 percent (Grover and Knopf, 1982). The principal factors influencing nesting success of snowy plovers in the SGP ecoregion are severe weather and flooding, predation, and disturbance by humans and domestic animals (Grover and Knopf, 1982; Conway and others, 2005; Saalfeld and others, 2011). In the southern High Plains of Texas, nesting success of snowy plovers declined 31 percent over a 10-year period, largely because of increases in nest predation (Saalfeld and others, 2011) that was attributed to increases in populations of common ravens (*Corvus corax*) and Chihuahuan ravens (*Corvus cryptoleucus*) in the region.

Landscape Structure and Dynamics

The distribution of snowy plovers in the SGP ecoregion is dictated by the availability of suitable nesting habitat. Historically, snowy plovers were probably restricted to breeding along saline lakes, sand bars on large rivers, and large ephemeral wetlands where evaporation created suitable nesting substrate (Busby, 2002). More recently, some large reservoirs within the region may provide suitable nesting habitat if water is drawn down in early spring, exposing large, unvegetated areas (Mabee and Estelle, 2000). Saline lakes, however, have a longer hydroperiod than other breeding habitats in the region (Reeves and Temple, 1986), providing opportunities for multiple brooding, which has not been observed for plovers in other habitats of the SGP (Conway and others, 2005).

Annual variation in precipitation has a large influence on the distribution of breeding habitat within the SGP. High amounts of rainfall may flood ephemeral wetlands and delay the drawdown of reservoirs, making them unsuitable sites for breeding in some years (Busby, 2002). On the other hand, low amounts of rainfall may lead to dewatering in some wetlands and rivers, causing plovers to abandon breeding sites. More recently, groundwater mining has resulted in the dewatering of some wetlands and rivers in the SGP, making them unsuitable for nesting plovers (Busby, 2002). Groundwater mining may be especially problematic at saline lakes, which in the SGP are discharge wetlands fed by springs that in many cases are no longer hydrologically connected to aquifers from which water is being withdrawn (Reeves and Temple, 1986).

Associated Species of Management Concern

Other species whose habitat requirements and response to management are similar to those of the snowy plover include the piping plover (*Charadrius melodus*), killdeer (*Charadrius vociferus*), spotted sandpiper (*Actitis macularius*), American avocet (*Recurvirostra americana*), black-necked stilt (*Himantopus mexicanus*), and least tern (*Sternula antillarum*). Plovers often nest in association with least tern colonies along rivers within the region (Busby, 2002).

Change Agents

Development

Energy and Infrastructure

No studies have examined the effects of energy development and related infrastructure on snowy plovers, largely because snowy plover breeding sites rarely coincide with energy development. In Wyoming, energy development was associated with increases in common ravens (*Corvus corax*) (Bui and others, 2010), which are known predators of plover eggs and young (Page and others, 2009). Therefore, energy development close to plover nesting areas could result in increased nest predation and reduced nesting productivity.

Agricultural Activities

Conversion of wetlands to agriculture removes potential breeding habitat for snowy plovers and has likely caused a significant reduction in nesting habitat within the SGP ecoregion. For example, wetlands north of Garden City, Kansas, that supported snowy plovers in the 1950s (Davis, 1964; Zuvanich and McHenry, 1964) have been converted to agriculture and are no longer used by breeding plovers (Busby, 2002). While the conversion of wetlands for agricultural use is the most common reason for physical alteration of wetlands in the SGP, altering wetland for enhancing waterfowl habitat generally increases vegetation cover adjacent to waterbodies, thereby degrading or eliminating snowy plover breeding habitat (Busby, 2002). In addition, sediment infilling from upland agriculture is projected to fill most playas of the SGP by the early 2100s (Burris and Skagen, 2013). Saline lakes also are likely to be affected by sedimentation, which means that much breeding habitat for snowy plovers may be eliminated in the SGP ecoregion unless mitigation measures are implemented.

Extensive groundwater pumping in the Great Plains has caused a drop in the level of the Ogallala aquifer, leading to declines in discharge to streams and springs in the region (McGuire and others, 2003). Decreased water availability has shortened hydroperiods and increased the salinity of the region's saline lakes (Brune, 2002), in some cases making them unsuitable for migrating (Andrei and others, 2008) and nesting (Conway and others, 2005) shorebirds. Reduced discharge from springs can also alter the temporal hydrodynamics of saline lakes, also making them less suitable for nesting plovers. As input from springs declines, precipitation has a greater influence on water input, resulting in greater variability in water levels. Greater fluctuations in water levels can, in turn, increase nest loss of ground-nesting birds (Busby, 2002).

Groundwater pumping also has led to reduced instream flows of rivers in the region (Cross and others, 1985), which can affect snowy plover nesting habitat in two ways. First, decreases in flow may lead to dewatering of some river sections that support plover nest sites. For example, portions of the Cimarron River that supported breeding colonies of least terns and snowy plovers in the 1980s have had little or no flow during the nesting season for much of the past 20 years (Schulenberg and Schulenberg, 1982; Cross and others, 1985; Dodds and others, 2004). In addition to dewatering, reduced instream flow minimizes the scouring that occurs during high flows, allowing riparian vegetation to encroach on plover nesting sites (Busby, 2002). Along a 125-km stretch of the Cimarron River between Meade County, Kansas, and Freedom, Oklahoma, the number of active least tern colonies, which often nest in association with snowy plovers, declined from 18 to 4 between 1982 and 1993. Boyd (1994) attributed this decline to vegetation encroachment and erosion of historical nesting sites. Interior least tern populations, however, have remained stable or increased along rivers in SGP region (Lott and others, 2013), suggesting that the loss of nesting habitat for terns and plovers may not have been as severe as originally believed.

When livestock graze along rivers, they may congregate in the riverbed and trample nests of ground-nesting birds. Livestock have been observed trampling snowy plover nests along the Cimarron River in Kansas (Schulenberg and Schulenberg, 1982; Boyd, 1992), but it is unlikely that livestock would cause problems at plover breeding sites away from rivers because they are less likely to move across unvegetated sites adjacent to lakes and playas.

Pesticides and other contaminants associated with agricultural activities are a management concern for this species. Pesticides and other contaminants may concentrate in wetlands, causing problems for species that forage or breed in wetland environments. Selenium may concentrate in wetlands that receive runoff from agricultural fields and has been shown to cause developmental abnormalities in shorebirds that use those wetlands (Hamilton, 2004). No comprehensive studies of contaminant levels in waterbirds have been conducted within SGP ecoregion, but in a small sample (n = 7) of interior least tern (*Sterna antillarum athalassos*) eggs obtained from Quivira NWR in central Kansas in 1992 and 1993 (Allen and others, 1998), the levels of arsenic, mercury, selenium, and chlorinated hydrocarbon compounds did not exceed the thresholds beyond which these contaminants are considered unsafe for embryo viability (U.S. Department of the Interior, 1998). The only study of contaminant levels in snowy plovers was conducted in coastal southern California, and there was no evidence of lethal or sublethal levels of heavy metals, organochlorines, or polychlorinated biphenyls (PBC) and their derivatives in snowy plover eggs (Hothem and Powell, 2000).

General Anthropogenic Effects

Human disturbance has been linked to decreases in nesting productivity (Ruhlen and others, 2003) and reduced breeding site occupancy (Webber and others, 2013) in coastal snowy plover populations. In addition, human disturbance was identified as a key threat to coastal populations of the western snowy plover and was one of the factors that led to its threatened status (U.S. Fish and Wildlife Service, 1993). In the SGP, however, human disturbance rarely causes nesting failure (Grover and Knopf, 1982; Conway and others, 2005; Saalfeld and others, 2011), probably because plovers generally nest in sites that receive little human use.

Altered Fire Regime

It is unlikely that changes in the fire regime will influence snowy plover distribution in the SGP. Fire suppression could result in the expansion of juniper (*Juniperus* spp.) (Engle and others, 1996) and mesquite (*Prosopis* spp.) woodlands (Hagen and others, 2004) within the plan area, but these species are not likely to become established in snowy plover breeding habitat.

Invasive Species

Invasive plants have been identified as a key factor leading to the reduction of snowy plover breeding habitat in coastal locations (Page and others, 2009). Plovers require largely unvegetated ground for breeding and therefore any plant species that invades plover nesting sites will eliminate potential breeding habitat (Page and others, 2009). Tamarisk (*Tamarix* spp.) is an aggressive invasive along rivers and wetlands in the region and may reduce potential plover breeding habitat both through expansion into potential breeding areas and through dewatering wetlands and rivers via evapotranspiration (Busby, 2002).

Disease

Snowy plovers are susceptible to botulism (Alcorn, 1942), which is a major source of nonhunting mortality of waterfowl in the SGP plan area (Haukos and Smith, 1992). West Nile virus (WNv) has been detected in Charadriiformes including the Kentish plover (*Charadrius alexandrinus*), which until recently was considered the same species as the snowy plover (Page and others, 2009). We are unaware of any documented cases of WNv in snowy plovers, but because of their small population size and patchy breeding distribution, it is likely that they have been sampled less intensively than other bird species.

Climate Change

Projections over the next century indicate that temperatures will increase and annual average precipitation may decrease and become more variable in the SGP (Karl and others, 2009). If these changes alter hydrology of lakes and rivers in the region, they may have a negative effect on snowy plover nesting habitat. Increased evapotranspiration from higher temperatures combined with reduced annual rainfall may reduce the hydroperiod of saline lakes, ephemeral wetlands, and riverine habitat, shortening the breeding season. In addition, increased variation in precipitation could lead to greater nest losses from flooding.

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting the snowy plover is illustrated in figure 33. Ecological attributes and CAs identified in the REA are enumerated in tables 35 and 37.

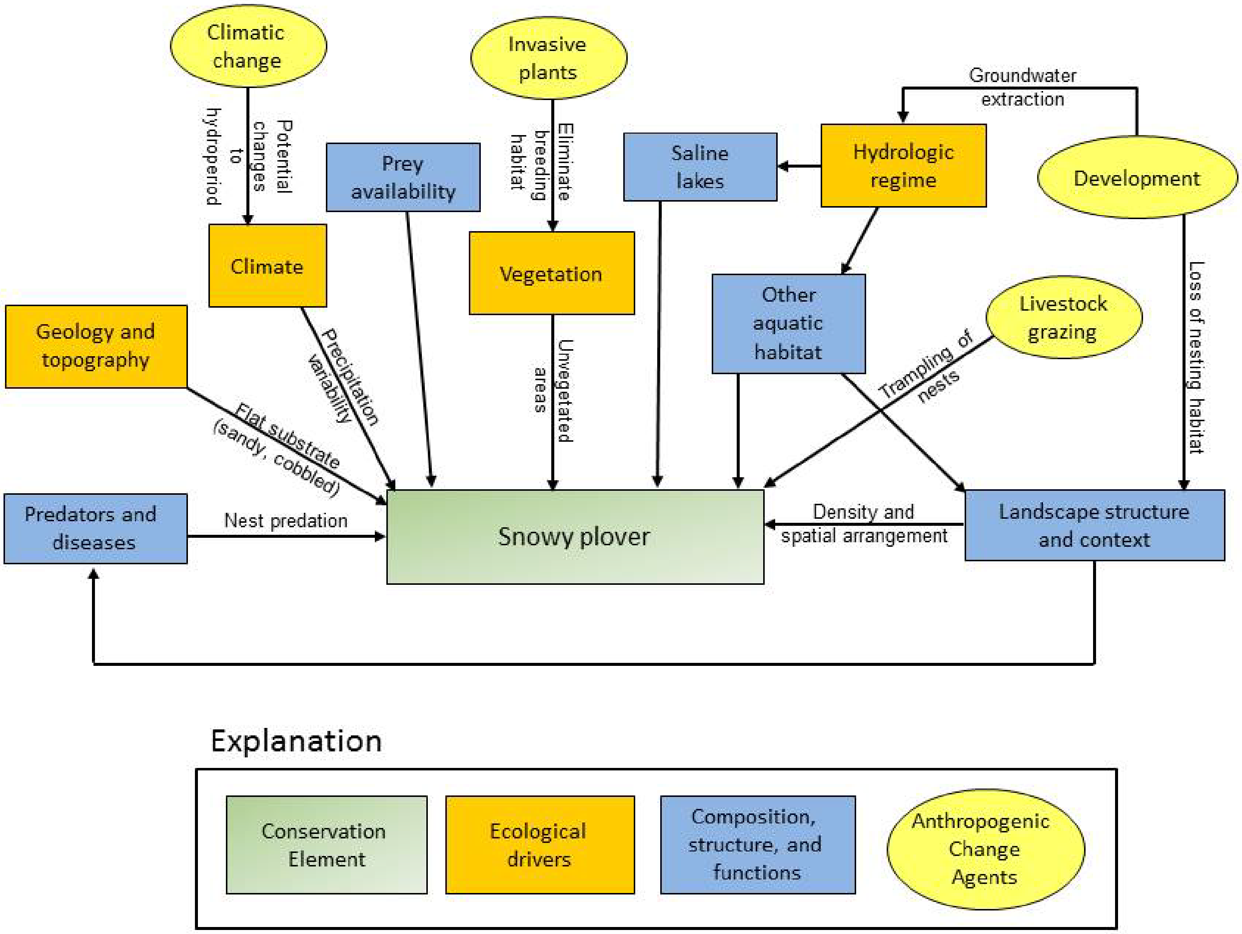

Figure 33. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for the snowy plover (*Charadrius nivosus*) in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of snowy plover populations and habtiats are shown in orange rectangles (see also table 36); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 37).

Table 36. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for the snowy plover (*Charadrius nivosus*).

Attribute	Variables
Amount and distribution	Snowy plover distribution.
Landscape structure	Size and spatial distribution of breeding (density, connectivity).
Landscape dynamics	Nesting habitat availability and productivity (food resources), invasive species.
Associated species of management concern	Piping plover, killdeer, spotted sandpiper, American avocet, black-necked stilt, and interior least tern.

Table 37. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for the snowy plover (*Charadrius nivosus*).

Attribute	Variables
Development (energy and infrastructure)	Habitat alteration leads to an increase in predators.
Development (agricultural activities)	Habitat alteration (direct loss of habitat due to cultivation of wetlands), groundwater extraction (reduced discharge to saline lakes), livestock grazing (trampling of nests), pesticides and other contaminants.
Development (human effects)	Habitat alteration (disturbance linked to decrease in nesting productivity).
Invasive species	Habitat alteration (reduction of breeding habitat), increased evapotranspiration may lead to dewatering of wetlands and streams (tamarisk).
Disease	Presence and effect of West Nile virus and botulism on the population.
Climate change	Potential changes to habitat as increased temperature and increased variability in precipitation could alter hydroperiod of habitat.

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Chapter 19. Bat Species Assemblage

By Mark A. Hayes

Key Ecological Attributes

Distribution and Ecology

Of the 45 bat species that occur in the contiguous United States, 37 percent (17 species) regularly occur in the SGP ecoregion (table 38, fig. 34). This document provides an introduction to the natural history of bats that occur in the SGP, including a brief discussion of bat diversity and distributions, roosting ecology, diet, and reproductive patterns. Recorded occurrences of four bat species that occur in the SGP are shown in figure 34, each species with different life history attributes, physical adaptations, geographic distributions, and evolutionary histories: (1) Townsend's big-eared bat (*Corynorhinus townsendii*), (2) pallid bat (*Antrozous pallidus*), (3) Mexican free-tailed bat (*Tadarida brasiliensis*), and (4) eastern red bat (*Lasiurus borealis*).

The SGP bat assemblage includes species with different continental distribution patterns, thus populations of these bats provide ecological connections on local, regional, and continental scales. Several species, such as the pallid bat and Yuma myotis (*Myotis yumanensis*), are generally associated with arid landscapes of the West, and the eastern edge of their distributions occur in the SGP (Hermanson and O'Shea, 1983; Dalquest and others, 1990). Other species, such as the eastern red bat and northern long-eared bat (*Myotis septentrionalis*), are generally associated with the temperate forests of eastern North America, and the western edges of their ranges occur in the SGP (Shump and Shump, 1982a; Caceres and Barclay, 2000). Several long-distance migratory species and subspecies, such as the hoary bat (*Lasiurus cinereus*) and Mexican free-tailed bat (*Tadarida brasiliensis*), may travel more than 1,000 km between their summer and winter grounds, thus providing important ecological connections between the SGP and other ecoregions of the United States, as well Mexico and Canada. For example, an individual Mexican free-tailed bat that migrates between the SGP and Mexico provides an ecological connection between summer and winter grounds by consuming insect prey in both areas during different times of year.

Some bat species that occur in the SGP ecoregion undertake long-distance migrations, but they tend to roost individually or in small groups in the foliage of trees and, as a result, they are rarely or never seen by most people (Carter and others, 2003). Other bat species can occur in large, conspicuous groups, with some summer roosts containing hundreds or thousands of bats. As a result, these bats may be well known to some people living near these roosts. For example, Mexican free-tailed bats that spend the winter in south Texas and Mexico migrate to the SGP in the spring, forming several large summer colonies in Oklahoma (fig. 34). Each of these colonies provides a rare natural spectacle on summer evenings as thousands of bats leave their day roosts and spread out over the landscape in search of insect prey. Several of these colonies may contain as many or more bats as usually occur at the well-known Carlsbad Caverns National Park bat colony in New Mexico (Glass, 1982; McCracken, 2003).

Table 38. Species of bats known to occur regularly in the Southern Great Plains (SGP) ecoregion, including common name, species binomial, general distribution within the SGP, and rationale for including each species in the SGP Rapid Ecoregional Assessment. A distribution descriptor of “Southern Great Plains” indicates the species is known to occur in all three ecoregions within the SGP (Central Great Plains, High Plains, and Southwestern Tablelands). Species are considered to occur regularly in the SGP if the species is known to do so based on (1) one or more authoritative citations, and (or) (2) the Global Biodiversity Information Facility (2014) database indicates that at least five specimens of the species came from the SGP ecoregion with distinct geographic coordinates. Occult little brown bat (*Myotis occultus*) and canyon bat (*Parastrellus hesperus*, also known as western pipistrelle), may occur regularly in the SGP (Global Biodiversity Information Facility, 2014) but did not meet the criteria for inclusion.

[GBIF, Global Biodiversity Information Facility]

Species	General distribution in the Southern Great Plains	Rationale for inclusion in Southern Great Plains Rapid Ecoregional Assessment
	Family Vespertilionidae	
Pallid bat (*Antrozous pallidus*)	Southwestern Tablelands	Hermanson and O’Shea (1983), GBIF (2014)
Big brown bat (*Eptesicus fuscus*)	Southern Great Plains	Kurta and Baker (1990), GBIF (2014)
Silver-haired bat (*Lasionycteris noctivagans*)	Southern Great Plains	Kunz (1982), GBIF (2014)
Eastern red bat (*Lasiurus borealis*)	Southern Great Plains	Shump and Shump (1982a), GBIF (2014)
Hoary bat (*Lasiurus cinereus*)	Southern Great Plains	Shump and Shump (1982b), GBIF (2014)
Western small-footed myotis (*Myotis ciliolabrum*)	Southern Great Plains	Holloway and Barclay (2001), GBIF (2014)
Little brown myotis (*Myotis lucifugus*)	Southern Great Plains	Fenton and Barclay (1980), GBIF (2014)
Northern long-eared myotis (*Myotis septentrionalis*)	Central Great Plains	Caceres and Barclay (2000), GBIF (2014)
Fringed myotis (*Myotis thysanodes*)	Southwestern Tablelands	O’Farrell and Studier (1980), GBIF (2014)
Cave myotis (*Myotis velifer*)	Southern Great Plains	Fitch and others (1981), GBIF (2014)
Long-legged bat (*Myotis volans*)	Southwestern Tablelands	Warner and Czaplewski (1984), GBIF (2014)
Yuma myotis (*Myotis yumanensis*)	Southern Great Plains	GBIF (2014)
Evening bat (*Nycticeius humeralis*)	Central Great Plains, Southwestern Tablelands	Watkins (1972), GBIF (2014)
Tricolored bat (*Perimyotis subflavus*)	Southern Great Plains	Fujita and Kunz (1984), GBIF (2014)
Townsend’s big-eared bat (*Corynorhinus townsendii*)	Southern Great Plains	Kunz and Martin (1982), GBIF (2014)
	Family Molossidae	
Big free-tailed bat (*Nyctinomops macrotis*)	Southern Great Plains	Milner and others (1990), GBIF (2014)
Brazilian free-tailed bat (*Tadarida brasiliensis*)	Southern Great Plains	Wilkins (1989), GBIF (2014)

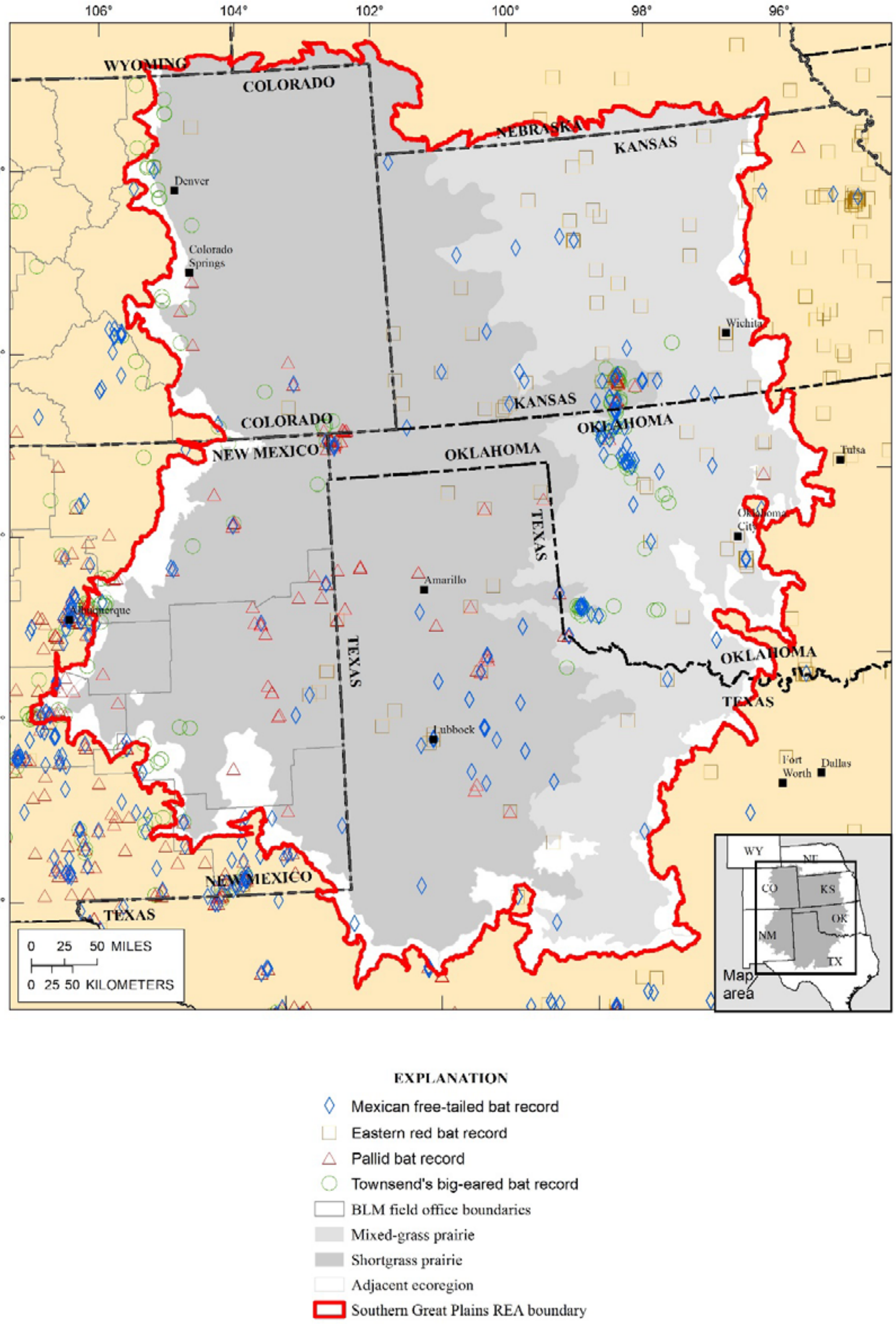

Figure 34. Recorded occurrences of the four species included in the bat species assemblage in the Southern Great Plains ecoregion (data source: Global Biodiversity Information Facility). (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (BLM, Bureau of Land Management; REA, Rapid Ecoregional Assessment)

All bat species that regularly occur in the SGP are primarily insectivorous, consuming various kinds of insect and other invertebrate prey. Four dietary classifications can be applied to bat species in the SGP (Findley, 1993): (1) aerial insectivores of forests and clearings, (2) gleaning insectivores, (3) water-surface foragers, and (4) open-air aerial insectivores. Forest and clearing aerial insectivores tend to forage on airborne insects in wooded landscapes. These bats, which include the silver-haired bat (*Lasionycteris noctivagans*) and big brown bat (*Eptesicus fuscus*), can usually maneuver well while flying in and among trees. Gleaning insectivores are bats capable of capturing prey from vegetation or the ground. These bats, which include the fringed myotis (*Myotis thysanodes*) and pallid bat (*Antrozous pallidus*), are capable of rapid maneuvers in flight and can often hover over prey before the final attack. Water-surface foragers are bats that tend to forage on insect prey over bodies of water. These bats, which include the Yuma myotis and little brown bat (*Myotis lucifugus*), often can be seen foraging on aquatic insects emerging from rivers, lakes, and other waterbodies. These bats are usually adept fliers capable of rapid maneuvers and provide trophic connections between terrestrial and aquatic systems. Open-air aerial insectivores are bats that usually forage on airborne insects in open areas or high above the ground and well above the vegetation. These bats, which include the Mexican free-tailed bat and hoary bat, usually use rapid, direct flight when pursuing prey and may not be capable of maneuvering in tight quarters, such as in densely forested areas. The amount and variety of insect prey consumed by bats make them important predators of nocturnal insects. For example, Mexican free-tailed bats (Cleveland and others, 2006) consume large amounts of crop pests and likely provide substantial benefits to agricultural economies.

Bats that occur in the SGP breed in either autumn, winter, or spring, but in all cases give birth to young in the late spring and summer (Harvey and others, 2011). Female bats that copulate in the autumn and winter will store sperm in utero until spring when fertilization occurs (Racey and Entwistle, 2000). Gestation typically lasts from less than two months to more than three months (table 38 provides references that detail reproductive biology for most species that occur in the SGP). Females usually give birth to one altricial young per year, although solitary foliage-roosting bats (eastern red bat, hoary bat, and silver-haired bat) may give birth to 2–5 young (Racey and Entwistle, 2000). Reproductive females nurse the young until they are capable of flying and foraging on their own. During pregnancy and lactation, reproductive females require continuous access to high-quality food and water resources (Kunz and Stern, 1995). Young bats of the temperate zone are often fully weaned and volant by two months of age (Kunz and others, 2009). Environmental conditions, including roost temperatures, prey availability, and access to water, may influence gestation rate, timing of parturition, lactation, and timing of volancy (Heideman, 2000). By late summer and early autumn, bats are preparing to migrate south to warmer winter locations or to hibernate in the SGP or nearby ecoregions. Juveniles have the substantial challenge of successfully learning to fly and forage on their own and preparing for a long migratory flight or a prolonged hibernation period. While survival rates for adult bats are generally high, first-year survival is usually substantially lower because of the added stresses on young bats (O'Shea and others, 2004).

Bats are prey to many animals, including other mammals, birds, reptiles, amphibians, and fish (Sparks and others, 2000). Raptors and snakes are probably the most important predators of bats in the SGP. Owls, falcons, and hawks are known to deliberately hunt bats, and in the SGP it is likely that raptors are aware of and exploit large bat colonies, such as those of Mexican free-tailed bats. Snakes hunt bats at roost sites in caves, mines, and rock features and often have been observed preying on young bats that fall to the ground underneath roosting areas.

There is little information on bat populations of the SGP or nearby ecoregions prior to European settlement (but see Czaplewski and Peachey, 2003, for information on bat fossils from the SGP), but bat populations can be vulnerable to rapid population declines (Hutson and others, 2001). Although individual bats often live more than 10 years (O'Shea and others, 2004), with the oldest documented bat living to at least 41 years of age (Podlutsky and others, 2005), bat populations tend to grow very slowly and may take years or decades to recover from population declines (O'Shea and others, 2003). In large part, this is a result of their reproductive rates. Unlike some other mammalian species, most bats only give birth once per year, often to a single young (Racey and Entwistle, 2000). Although none of the bat species that occur in the SGP are listed under the U.S. Endangered Species Act, several species are considered species of conservation concern (O'Shea and others, 2003).

Landscape Structure and Dynamics

Some bat species in the SGP congregate in select caves for reproduction and hibernation. Caves that serve as maternity roosts, where adult females give birth to and raise young, are usually relatively warm sites that facilitate fetal growth and rapid growth of newborns (Dalquest and others, 1990; Miller, 2011), and they are usually located near predictable foraging areas and high-quality water resources (Miller, 2011). Caves sought for winter hibernation are relatively cold, stable roosting environments where bats lower their core body temperatures and metabolic rates to help them conserve fat and survive the long cold period when little or no insect prey is available (Humphrey and Kunz, 1976; Prendergast and others, 2010).

Hibernation sites may have more stable temperatures than other potential sites (Sherwin and others, 2009; Hayes and others, 2011). Such specialized sites are not abundant throughout the SGP, and large segments of the regional bat populations of some species may be restricted to only a few roost sites during certain times of year (Prendergast and others, 2010). Some species may have very high fidelity to certain roosts, with some roost sites being used for many decades (Glass, 1982; Lewis, 1995; Prendergast and others, 2010). As such, bats can be extremely vulnerable to disturbance from human activities, as well as to physical destruction of roosts (Hutson and others, 2001). Many instances of wanton vandalism and intentional killing of bats have been reported from bat roosts in the United States, especially where human access to the sites is not restricted (Hutson and others, 2001).

The areas of the SGP with greatest bat species richness and abundance occur where topography and vegetation provide a variety of roosting options (Humphrey, 1975; Humphrey and Kunz, 1976). It is convenient to classify bats that occur in the SGP based on the type of roost sites they tend to occupy (Kunz, 1982). Some species usually roost in caves but also may roost in abandoned mines and other subterranean features that provide cavelike conditions. These species, such as the Townsend's big-eared bat and Mexican free-tailed bat, can be referred to as cave-roosting bats (Kunz, 1982; Tuttle, 2003). Other species, such as the pallid bat, tend to roost in rock features that provide crevices and sheltered areas high on cliff faces and in hogbacks and escarpments, or in between and under rocks in boulder fields and talus slopes. These species can be referred to as rock crevice–roosting bats (Kunz, 1982; Bogan and others, 2003). Other species usually roost among the foliage of trees or other vegetation, typically roosting individually or in small groups. These species can be referred to as solitary foliage-roosting bats (Kunz, 1982; Carter and others, 2003).

Bat Species Highlights

In this section we highlight four bat species that regularly occur in the SGP. These four species are described in three groups based on the roosting resources they typically use: cave-roosting species, rock crevice–roosting species, and foliage-roosting species. Townsend's big-eared bat and Mexican free-tailed bat are the representatives for cave-roosting species. We selected the Townsend's big-eared bat because it is known to occur in the SGP throughout the year and is also considered by some biologists to be a species of significant conservation concern that is known to be sensitive to human disturbance at roost sites. The Mexican free-tailed bat is highlighted because it forms large summer congregations in some parts of the SGP and likely consumes large quantities of agricultural insect pests during the summer. Unlike Townsend's big-eared bat, which occurs in the SGP throughout the year, the Mexican free-tailed bat migrates to wintering grounds south of the SGP. The pallid bat was selected to represent rock crevice–roosting species. Although these bats use caves, abandoned mines, and other roosting resources, generally they are considered rock crevice specialists. A number of other species that regularly occur in the SGP also roost in rock crevices, but few are considered rock crevice specialists. The eastern red bat was selected to represent individual foliage-roosting species. We highlight this species because more is known about its winter roosting ecology than that of other foliage-roosting species, such as the hoary bat. The roosting ecology of each bat species that occurs in the SGP is summarized in the relevant American Society of Mammalogy's mammalian species monograph (table 38) and (or) other publications about that species.

Cave-Roosting Species

Townsend's Big-Eared Bat

Townsend's big-eared bat has been documented throughout much of western North America from Oaxaca, Mexico, to British Columbia, Canada, and currently there are five recognized subspecies (Piaggio and others, 2009). With the exception of an isolated population in the eastern United States, the species' eastern distribution limits are in and near the Black Hills of South Dakota and south-central Kansas and western Oklahoma (Kunz and Martin, 1982). In the SGP, this species has been documented throughout the Southwestern Tablelands and in areas with cave and karst habitat (Humphrey and Kunz, 1976; Kunz and Martin, 1982; Prendergast and others, 2010; Miller, 2011). Townsend's big-eared bat is not known to be migratory and is known to hibernate in caves during winter (Humphrey and Kunz, 1976; Prendergast and others, 2010); individuals that spend summer months in the SGP are likely to reside in the area throughout the year.

Townsend's big-eared bats roost in caves, abandoned mines, rock crevices and shelters, and manmade structures, such as buildings, tunnels, and cliff dwellings (Kunz and Martin, 1982). These bats are sensitive to disturbance at roost sites, and loss of roosting resources may be a population-limiting factor (Humphrey and Kunz, 1976: Pierson and others, 1999). During spring and summer, females congregate in maternity colonies where birth takes place and pups are nursed until they are independent. During the summer maternity period, males are generally solitary or roost in small groups away from maternity roosts. Maternity roosts may consist of small groups or hundreds of reproductive females. For example, during the summer, maternity roosts in abandoned copper mines of southeastern Colorado and northeastern New Mexico may contain hundreds of Townsend's big-eared bats, among other species (Ellinwood, 1978; Hayes, 2011). Long-distance migration between summer and winter roosts has not been reported, and distance between summer and winter roosting sites may be less than 100 km (Kunz and Martin, 1982). During winter, bats tend to roost singly or in small clusters; however, large, mixed-sex congregations have been observed during winter (Humphrey and Kunz, 1976; Prendergast and others, 2010). Townsend's big-eared bats exhibit a high degree of fidelity to a given area and are known to return to the same maternity and hibernation sites year after year (Prendergast and others, 2010). During winter, however, an individual roosting within a cluster of sites may emerge for short periods and move to nearby roost sites (Sherwin and others, 2003; Hayes and others, 2011). Townsend's big-eared bats have used some caves and abandoned mines in the SGP as maternity and hibernation roosts since at least the 1960s (Humphrey and Kunz, 1976; Ellinwood, 1978). The species is known to occur in piñon-juniper woodlands and savannas, pine-oak woodlands, and areas with canyons and mesas, especially where predictable water resources are available (Armstrong and others, 2011; Hayes and others, 2011; Miller, 2011). It is generally considered to be a gleaning insectivore (Kunz and Martin, 1982; Findley, 1993) and consumes small moths, lacewings, beetle, flies, wasps, bees, and ants (Kunz and Martin, 1982). They are adept flyers and are capable of gleaning insect prey from vegetation surfaces, but they also may consume insect prey on the wing and while foraging over water.

Although the species has a widespread range, Townsend's big-eared bat is considered a globally vulnerable species (G3 in 2012) because small, local populations are declining and these bats are vulnerable to human disturbance (NatureServe, 2014). The ranks and status conferred to this species by state wildlife agencies of the SGP vary as follows: Colorado, imperiled (S2) and a Species of Special Concern; Kansas, imperiled (S1) and a Species in Need of Conservation; New Mexico, vulnerable (S3); Oklahoma, vulnerable (S3); and Texas, potentially vulnerable (S3). The International Union for Conservation of Nature (2013) lists Townsend's big-eared bat as a species of least concern with a stable population trend. Townsend's big-eared bat was listed as a Category 2 candidate for listing under the U.S. Endangered Species Act, but since elimination of this designation in 1996, it has been considered a species of concern (O'Shea and others, 2003). Hibernacula used by these bats in the SGP apparently have remained relatively stable since 1965 (Prendergast and others, 2010). Most caves and abandoned mines known to be used by Townsend's big-eared bats in the SGP are located on private lands (Prendergast and others, 2010; Miller, 2011).

Mexican Free-Tailed Bat

The Brazilian free-tailed bat (*Tadarida brasiliensis*) has been documented throughout Mexico and southern portions of the United States, including the SGP (Wilkins, 1989). There are two recognized subspecies in the United States (Wilkins, 1989; McCracken, 2003): the Mexican free-tailed bat (*T. b. mexicana*) occurs in western North America to eastern Oklahoma and Texas, and LeConte's free-tailed bat (*T. b. cynocephala*) occurs from Louisiana eastward throughout the southeastern United States. Mexican free-tailed bats that summer in the SGP migrate south in autumn to Texas (south of the Edwards Plateau) and Mexico where they are thought to remain active throughout the winter period (Glass, 1982).

Mexican free-tailed bats roost in caves, sinkholes, abandoned mines, rock crevices and shelters, tree crevices, and manmade structures such as bridges, buildings, tunnels, caves, and nest boxes (Wilkins, 1989; Ellison, O'Shea, and others, 2003). During spring and summer, female Mexican free-tailed bats congregate in large maternity colonies where birth takes place and pups are nursed to independence. During the summer maternity period, males are generally solitary or roost in small groups away from maternity roosts. Maternity roosts in the SGP may consist of small groups or hundreds of thousands, and perhaps more than one million, reproductive females (Glass, 1982; McCracken, 2003). For example, each of five caves known to be used as maternity roosts by this species in western Oklahoma may contain hundreds of thousands of individuals during the summer (Glass, 1982; McCracken, 2003). This species frequently migrates long distances, sometimes more than 1,800 km, between summer and winter roosts (Glass, 1982). Mexican free-tailed bats may move among available roost sites, and individual bats often move among roosts in different ecoregions (Glass, 1982); generally, however, populations exhibit a high degree of fidelity to a given area, and maternity roosts are usually established in the same sites year after year (McCracken, 2003). Near the end of the maternity period, bats may move among a cluster of nearby roost sites, perhaps to relieve pressure from overcrowding (Glass, 1982). Caves in the SGP of western Oklahoma have been used as maternity roosts since at least the 1950s (Glass, 1982).

Brazilian free-tailed bats are adapted for rapid, direct flight and are known to travel more than 50 km from roost sites to foraging areas (Wilkins, 1989). These bats tend to forage in areas uncluttered by vegetation, such as grasslands and agricultural fields, and are known to forage at high altitudes (McCracken, 2003). The species is generally considered to be an open-air insectivore (Findley, 1993) and consumes a wide variety of insect prey (Wilkins, 1989). Given that Mexican free-tailed bats occur in large numbers and are predators of agricultural insect pests, they contribute substantial natural value to agricultural economies (Cleveland and others, 2006).

NatureServe (2014) lists the Brazilian free-tailed bat as a globally secure species (G5). The ranks and status conferred to this species by State wildlife agencies of the SGP vary considerably as follows: Colorado, critically imperiled (S1); Kansas, status not available (SNA); New Mexico, vulnerable or apparently secure (S3/S4); Oklahoma, vulnerable (S3); and Texas, secure (S5). The International Union for Conservation of Nature (2013) lists the Brazilian free-tailed bat as a species of least concern with a stable population trend.

Rock Crevice–Roosting Species

Pallid Bat

The pallid bat has been documented in western North America from central Mexico to British Columbia, Canada, and is common in the southwestern United States (Hermanson and O’Shea, 1983). With the exception of a few scattered records, the eastern limits of the distribution appear to be in and near the Southwestern Tablelands of Kansas, Oklahoma, and Texas. Currently there is one recognized subspecies in the United States (Hermanson and O’Shea, 1983). The pallid bat is not known to be migratory and likely spends the winter hibernating in caves, abandoned mines, and rock crevices. Individuals that summer in the SGP are likely to reside in the area throughout the year (Hermanson and O’Shea, 1983).

Pallid bats are common in arid regions with canyons and rock outcroppings, and usually they are found near water (Hermanson and O’Shea, 1983). They roost in rock crevices and shelters, caves, abandoned mines, and manmade structures such as buildings, bridges, and tunnels (Hermanson and O’Shea, 1983). Pallid bats appear to be uncommon in caves in some parts of their range (Hermanson and O’Shea, 1983), but they have been found repeatedly in association with caves in the SGP. These bats often roost in groups, with maternity colonies sometimes comprising several hundred bats (Hermanson and O’Shea, 1983). During spring and summer, females congregate in maternity colonies where birth takes place and pups are nursed to independence. During the summer maternity period, males generally roost in groups away from maternity roosts, but the bachelor roosts can contain hundreds of male bats (Hermanson and O’Shea, 1983). For example, in the Southwestern Tablelands of southeastern Colorado, male pallid bats were found roosting high on cliff faces near the Purgatoire River (Schorr, 2010). Long distance migration between summer and winter roosts has not been reported (Hermanson and O’Shea, 1983). Pallid bats are known to occur in piñon-juniper woodlands and savannas, pine-oak woodlands, and areas with canyons and mesas, especially where predictable water resources are available (Hermanson and O’Shea, 1983; Miller and Jensen, 2013).

The pallid bat is generally considered to be a gleaning insectivore (Findley, 1993). It consumes insects and ground-dwelling arthropods, often taking prey directly from the ground or gleaning prey from vegetation (Hermanson and O’Shea, 1983). Pallid bats are also known to use aerial hawking for capturing prey (Hermanson and O’Shea, 1983).

NatureServe (2014) lists the pallid bat as a globally secure species (G5), as it occupies a large range in the western part of the North American continent. The ranks and status conferred to this species by State wildlife agencies of the SGP vary as follows: Colorado, apparently secure (S4); Kansas, critically imperiled (S1) and a Species in Need of Conservation; New Mexico, secure (S5); Oklahoma, vulnerable (S3); Texas, secure (S5). The International Union for Conservation of Nature (2013) lists the pallid bat as a species of least concern with a stable population trend.

Individual Foliage-Roosting Bat

Eastern Red Bat

The eastern red bat has been documented throughout eastern North America from Mexico through the northern Great Plains of Canada (Shump and Shump, 1982a). In the SGP, most occurrence records have come from the eastern half of the ecoregion (Shump and Shump, 1982a). There is currently one recognized subspecies in the United States (Shump and Shump, 1982a). Eastern red bat is a strongly migratory species that appears to overwinter in the southeastern United States and Mexico and migrates to the Great Plains during spring (Cryan, 2003). Eastern red bats are known to hibernate in leaf litter during winter in the southeastern United States, including the nearby Ozark Highlands ecoregion (Saugey and others, 1998; Mormann and Robbins, 2007), and it is possible that some individuals spend part or all of winter in the SGP.

Eastern red bats inhabit eastern temperate forests and riparian areas, where they roost in the foliage of trees in forests, woodlands, and riparian areas (Shump and Shump, 1982a). They often roost individually or in small groups in trees near streams, agricultural fields and pastures, and other edge habitats (Shump and Shump, 1982a). In the Great Plains, this species is common in some urban areas. For example, McClure (1942) found this species to be very common in the trees of a small Iowa town.

The eastern red bat is generally considered to be an open-air insectivore (Findley, 1993). It consumes a wide variety of insect prey, including flying true bugs, beetles, flies, moths, and ground-dwelling beetles (Shump and Shump, 1982a). Eastern red bats copulate in the late summer and autumn, and average litter size is two (Shump and Shump, 1982a). In late summer and early autumn, these bats begin migrating to their wintering grounds.

The eastern red bat is a globally secure species (G5 in 2012), given its large range and relatively stable population (NatureServe, 2014). The ranks and status conferred to this species by State wildlife agencies of the SGP vary as follows: Colorado, imperiled (S2); Kansas, secure (S5); New Mexico, vulnerable (S3); Oklahoma, apparently secure (S4); Texas, apparently secure (S4). The International Union for Conservation of Nature (2013) lists eastern red bat as a species of least concern with a stable population trend.

Change Agents

Development

Since the 1800s, settlement of the SGP has had a profound influence on the abundance and distribution of human structures, trees, and water resources and as a result may have influenced bat populations and distributions (Sparks and Choate, 2000). The development of human structures increased substantially and tended to occur from east to west and along well-defined routes of transportation and commerce; bats that use such structures had abundant potential roosting opportunities that were not available prior to settlement (Sparks and Choate, 2000). Trees are now distributed more broadly and are more abundant than they were prior to settlement, as they have been planted as wind breaks, wildlife habitat, shelterbelts, and in landscaping, and are associated with most farms, towns, and cities in the SGP. Prior to settlement, however, trees tended to have a patchy distribution along rivers and creeks, but these water courses now tend to have continuous riparian corridors. This increased abundance of trees now provides more roosting habitat for bats that use them, such as solitary, foliage-roosting tree bats. The distribution and availability of high-quality water resources also has changed dramatically. The development of reservoirs and other impoundments, irrigation canals, and groundwater withdrawals has changed when and where surface water is available to bats, the quality of available water, and the distribution and abundance of insect prey that require, or associate with, water. In many locations, lowered water tables have resulted in the loss of riparian woodlands, thus diminishing available roosting habitats. It has been hypothesized that the extirpation of American bison (*Bison bison*) in the Great Plains may have resulted in reduced water availability associated with bison wallows (Sparks and Choate, 2000).

As the number of people living and working in the SGP increases, the numbers of people exploring caves, abandoned mines, and other resources used by bats also increase. Even slight human disturbance can cause some species to vacate roosts, as shown by a case of Townsend’s big-eared bats vacating a roost when they were disturbed by people (Pierson and others, 1999; Hayes and others, 2011). Therefore, a key conservation strategy used at important bat roosts is to restrict human access by installing metal bat gates (Pierson, 1998), which allow bats continued use of the sites while preventing unauthorized people from entering them.

Energy and Infrastructure

Energy development and infrastructure can have a variety of direct and indirect negative effects on bats through collisions with infrastructure, disturbance, habitat loss, and contamination of food and water resources. A key threat currently having direct effects on bats in North America is the development and expansion of wind energy facilities (Cryan, 2011). Dead bats have been found underneath wind turbines across North America, and bat fatalities have been documented at almost all wind facilities where thorough bat surveys have been conducted

(Ellison, 2012). Recent results suggest that thousands of bats may be killed annually at some wind facilities, and hundreds of thousands of bats may be killed annually in the contiguous United States (Ellison, 2012). The Great Plains region surpasses all other areas of the United States in terms of available wind resources, and proposed wind energy development in the Great Plains states indicates that it will increase sixfold in the coming years (U.S. Department of Energy, 2008). The current production of and capacity for wind-generated electricity in the Great Plains states may represent more than half of the total production for the contiguous United States (Fargione and others, 2012).

Oil and gas extraction processes can result in lowered water tables, which in turn can lead to dewatering of small streams, ephemeral ponds, and springs important to bats (Finley and others, 1983). These activities also may contaminate surface and groundwater resources, potentially exposing bats to contaminants (Bat Conservation International, 2012). Disturbances associated with energy development, including noise, also may influence the occurrence and behavior of foraging and roosting bats. Indeed, some bats have been shown to avoid acoustically loud environments when foraging (Shaub and others, 2008).

Agricultural Activities

There is little information on the effects of agriculture and grazing on bats in the SGP ecoregion; therefore, inferences are drawn from studies in other ecoregions. Monoculture croplands, pesticides, and livestock grazing may reduce insect prey available to bats in the Great Plains. In the Canadian Great Plains, however, a recent study indicated that insect and bat species richness and diversity did not differ significantly between rural agricultural and urban areas (Coleman and Barclay, 2013). Some bats are well adapted to take advantage of agricultural infrastructure, such as barns, outbuildings, and cellars, and some bats commute along hedgerows and shelterbelts. Bats also have been observed feeding on insects emerging from corn bins and wood piles, which in some cases for short periods may provide abundant prey. In other ecoregions, bat richness and diversity appeared to be greater in low-intensity agricultural areas, such as pastures with native vegetation cover, and scattered trees, including the associated logs and snags (Lentini and others, 2012). Some bats may be drawn to agricultural areas by the insect prey associated with some crops, including agricultural pests. For example, Mexican free-tailed bats are known to fly long distances in pursuit of emergent insect larvae, such the cotton bollworm (*Helicoverpa zea*; commonly known as corn earworm), which is an important agricultural pest (Cleveland and others, 2006). In some areas, these bats may substantially reduce populations of agricultural crop pests, thus providing substantial economic value to agricultural systems (Ghanem and Voight, 2012). Overall, bats can exert top-down control of agricultural crop pests (Ghanem and Voigt, 2012).

Effects of range management practices on bat populations in grassland ecosystems may influence bat populations through loss of riparian habitat, changes in vegetation structure, pesticide use, and increased availability of drinking water (livestock watering tanks and ponds) for some species (Chung-MacCoubrey, 1996; Ellison, Wunder, and others, 2003). Livestock trampling and grazing can increase soil compaction, reducing the soil’s permeability to water, and alter plant community composition, structure, and cover, all of which can affect invertebrate prey availability and abundance; however, direct effects of grazing and range management practices on bats in the SGP remain understudied.

Altered Fire Regime

There is little information on the effects of altered fire regimes on bats, although it is likely that some bats have benefitted from fire suppression in the SGP. Generally, fire suppression has allowed a westward expansion of forested areas (see the "Invasive Species" section, below), especially those associated with riparian corridors. Prescription burns, however, may have negative effects on bats, such as the eastern red bat, that hibernate in leaf litter (Mormann and Robbins, 2007). During winter in Arkansas, for example, red bats were observed apparently emerging (perhaps in response to smoke and heat) from leaf litter hibernacula during prescription burns (Saugey and others, 1998). Altered fire regimes also can influence the availability of roosting structures in trees, and they can change the species composition and abundance of insects and other prey consumed by bats.

Invasive Species

Invasive plant species may have positive and negative influences on bats in the SGP. For example, the spread of eastern redcedar (*Juniperus virginiana*) may increase foraging and roosting sites for some species (Miller, 2011). On the other hand, eastern redcedar and other invasive vegetation may reduce unimpeded access (open air space) to water resources required by some bat species (Humphrey and Kunz, 1976), which would be especially problematic near maternity roosts where unimpeded access to water is crucial for reproductive females and young.

Introduced Insects and Disease

We found no published literature regarding the influence of introduced insects on bat populations in the SGP. The only introduced disease known to have large effects on bat populations, and which could affect bat populations in the SGP, is white-nose syndrome (WNS) (Wibbelt and others, 2009). This disease, which affects hibernating bats, was first documented in the eastern United States in February 2006 (see Castle and Cryan, 2010, for an overview of this disease). During the first two years after this disease emerged, some bat populations in eastern North America may have declined by more than 75 percent, and from 2006 to 2011 more than 5.5 million bats of several species may have died, leading to regional population collapses and a potential for extinction of some species (U.S. Fish and Wildlife Service, 2012). This disease has been particularly destructive to species that hibernate in large congregations in the eastern United States and Canada (Castle and Cryan, 2010).

The fungus *Pseudogymnoascus destructans* is now considered to be the causal agent of WNS (Minnis and Lindner, 2013). It is not clear whether this fungus is a nonnative, invasive species recently introduced into bat hibernation habitats in North America or a virulent strain of a fungus with global distribution. As of spring 2014, there had not been any definitive documentations of WNS in the SGP ecoregion. During winter 2009–2010, however, there were reports of one or more bats with WNS from a cave in northwestern Oklahoma, although the presence of WNS or the causative fungus have not been confirmed (see www.whitenosesyndrome.org). Nevertheless, there is an almost continuous distribution of cave and karst habitats where WNS has been confirmed in the Southwestern Tablelands of the SGP (Culver and others, 1999; Veni, 2002); thus, it is possible that the Southwestern Tablelands represent a bridge across which WNS could expand into western North America.

Climate Change

Bat ecologists are just beginning to understand how climate change may affect bat species in North America. In the SGP, projected changes in climate could lead to changes in roost microclimates, timing and success of reproduction and hibernation, and the distribution and abundance of vegetation, prey, and water resources (Sherwin and others, 2012). The potential effects of a changing climate on bat species in arid and semiarid parts of North America are of increasing concern. In the SGP, temperate zone insectivorous bats may be particularly susceptible to a warmer, drier climate because of their high rates of evaporative and respiratory water loss and need for constant access to high-quality water resources. Because of their slow reproductive rates, factors that suppress reproductive output of bats are of concern. Indeed, some researchers have hypothesized that climate warming, and the resulting loss of natural surface water, may result in reduced reproductive output and lactation by bats in arid landscapes (Adams and Hayes, 2008; Adams, 2010). For example, at the transition between the Southern Rockies and the SGP, reproductive rates of *Myotis* species appeared to decline by up to 50 percent in drier years when streamflow and water availability were significantly reduced (Adams, 2010). For species that occur in warmer, drier regions, the cumulative effects of slight annual declines in reproductive rates during dry years over 50–100 years could result in dramatically reduced bat populations, even under the more conservative climate change scenarios (Hayes, 2011).

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting the bat species assemblage is illustrated in figure 35. Ecological attributes and CAs identified in the Rapid Ecoregional Assessment are enumerated in tables 39 and 40.

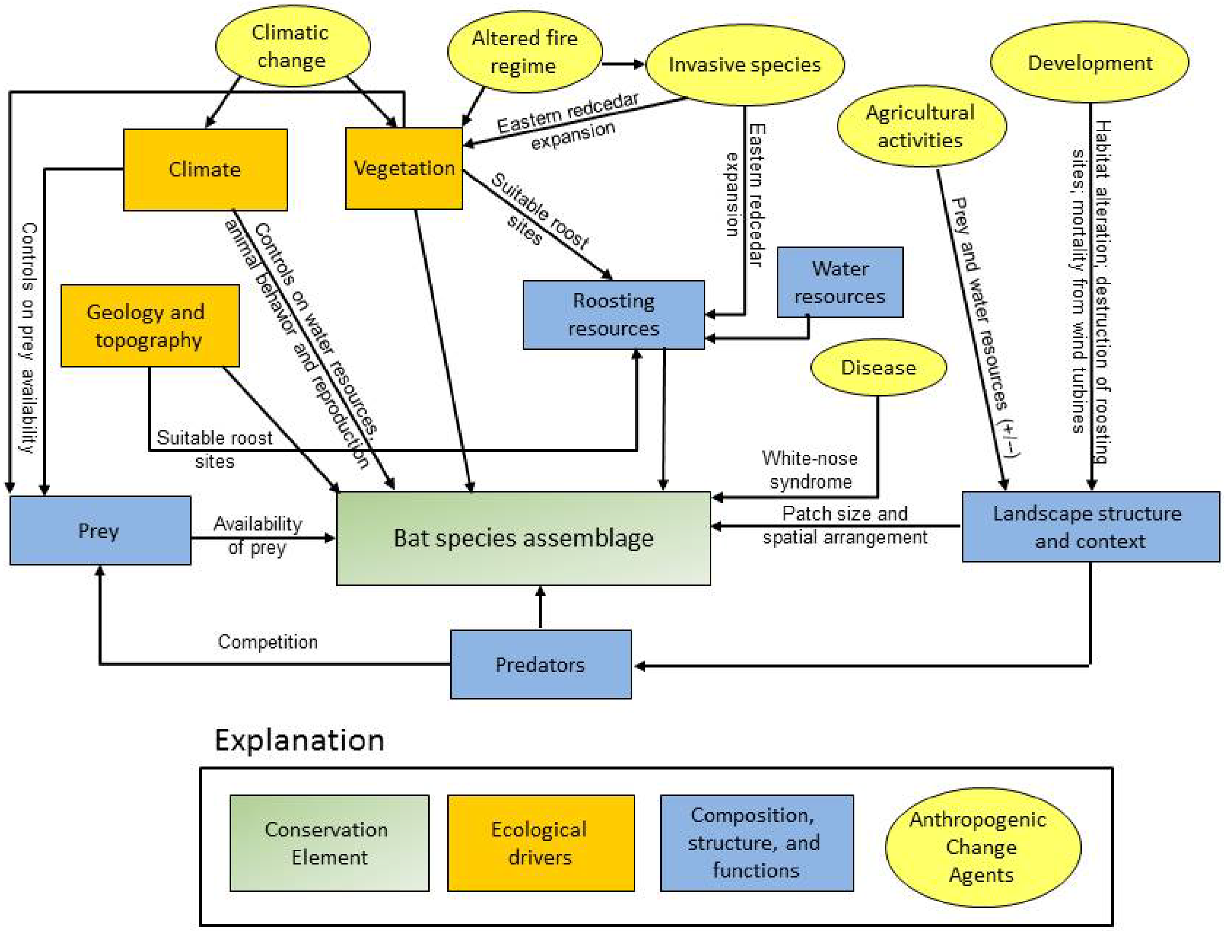

Figure 35. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for the bat species assemblage in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of bat species assemblage populations and habtiats are shown in orange rectangles (see also table 39); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 40).

Table 39. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for the bat species assemblage.

Attribute	Variables
Amount and distribution	Bat species distribution (occurrence records).
Landscape structure	Size and spatial distribution of habitat, availability and type of roost sites.
Landscape dynamics	Habitat productivity (availability of prey food resources), drought (effects to water resources, prey availability), roosting resources, predator dynamics.

Table 40. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for the bat species assemblage.

Attribute	Variables
Development (energy and infrastructure)	Habitat alteration (disturbance/destruction of roosting resources, noise pollution), contamination/loss of water resources, contamination of food resources, mortality (wind turbines).
Development (agricultural activities)	Habitat alteration (fluctuation in availability of prey [±], effects to water resources [±]).
Altered fire regime	Fire suppression results in habitat alteration (expansion of forested areas may increase roosting structures, effects to prey availability).
Invasive species	Habitat alteration (expansion of eastern redcedar—increase in forage and roosting [+], reduction in water resources [-]).
Insects and disease	Presence and effect of white-nose syndrome on population.
Climate change	Potential changes to habitat condition (distribution and abundance of vegetation, prey, and water resources; roost microclimate); potential changes to behavior (timing and success of reproduction and hibernation).

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Chapter 20. Black-Tailed Prairie Dog

By David A. Eads

Key Ecological Attributes

Distribution and Ecology

The black-tailed prairie dog (*Cynomys ludovicianus*) is a colonial, burrowing rodent that historically inhabited about 30 million ha of open grasslands in western North America. Since European settlement, however, the species has been extirpated from a large proportion of its historical range, including significant portions of the SGP ecoregion (Proctor and others, 2006). In the SGP, the black-tailed prairie dog (BTPD) is presently found in portions of southern and eastern New Mexico, western and central Texas, western Oklahoma, eastern Colorado, and western and central Kansas (fig. 36), and although they were once abundant in these five states, BTPDs now inhabit less than 1 percent of the area they occupied before 1900 (Proctor and others, 2006; U.S. Fish and Wildlife Service, 2009). Moreover, extant populations in the SGP and elsewhere tend to occur in isolated complexes comprising small, fragmented colonies that are uncharacteristic of the species' distribution before European settlement (U.S. Fish and Wildlife Service, 2009).

Because of its historical declines in abundance, fragmented distribution, and susceptibility to multiple CAs, the BTPD has been petitioned repeatedly for listing under the Endangered Species Act. In 2000, the U.S. Fish and Wildlife Service concluded that the BTPD was a candidate for listing, but that decision was reversed in 2004, partly because new estimates (many from aerial surveys) suggested that the cumulative area occupied by BTPDs had been underestimated (U.S. Fish and Wildlife Service, 2004). The same decision was made during subsequent appraisals (U.S. Fish and Wildlife Service, 2009), sparking additional proposals for Federal protection of the BTPD (Miller and Reading, 2012). It has been suggested, however, that recent aerial surveys overestimated the cumulative area occupied by BTPDs by as much as 94 percent in some cases (Sidle and others, 2012); thus, the debate about the species' status continues (Rauscher and others, 2013). Meanwhile, there are proposals for more effective conservation of the BTPD, with particular reference to its importance as a keystone species and ecosystem engineer (Miller and Reading, 2012). In addition, biologists have evaluated the suitability of habitats for BTPDs for guiding and prioritizing restoration actions (Proctor and others, 2006).

It is difficult to evaluate the suitability of habitats for the BTPD because scientific studies of the species began in the 1940s when its densities and distributions had already been reduced dramatically. Nonetheless, GIS-based analyses have provided at least some insights. For instance, in the southwestern portion of the SGP, correlations between habitat variables and the occurrence of active BTPD colonies suggested a positive association with surface soils of high organic content, pH, wetness, and depth to a restricted layer that is important for burrow construction (M.F. Antolin, ecologist, Colorado State University, oral. commun., January 2014). Habitat suitability was negatively associated with topographic slope, with slopes of less than 10 percent being preferred.

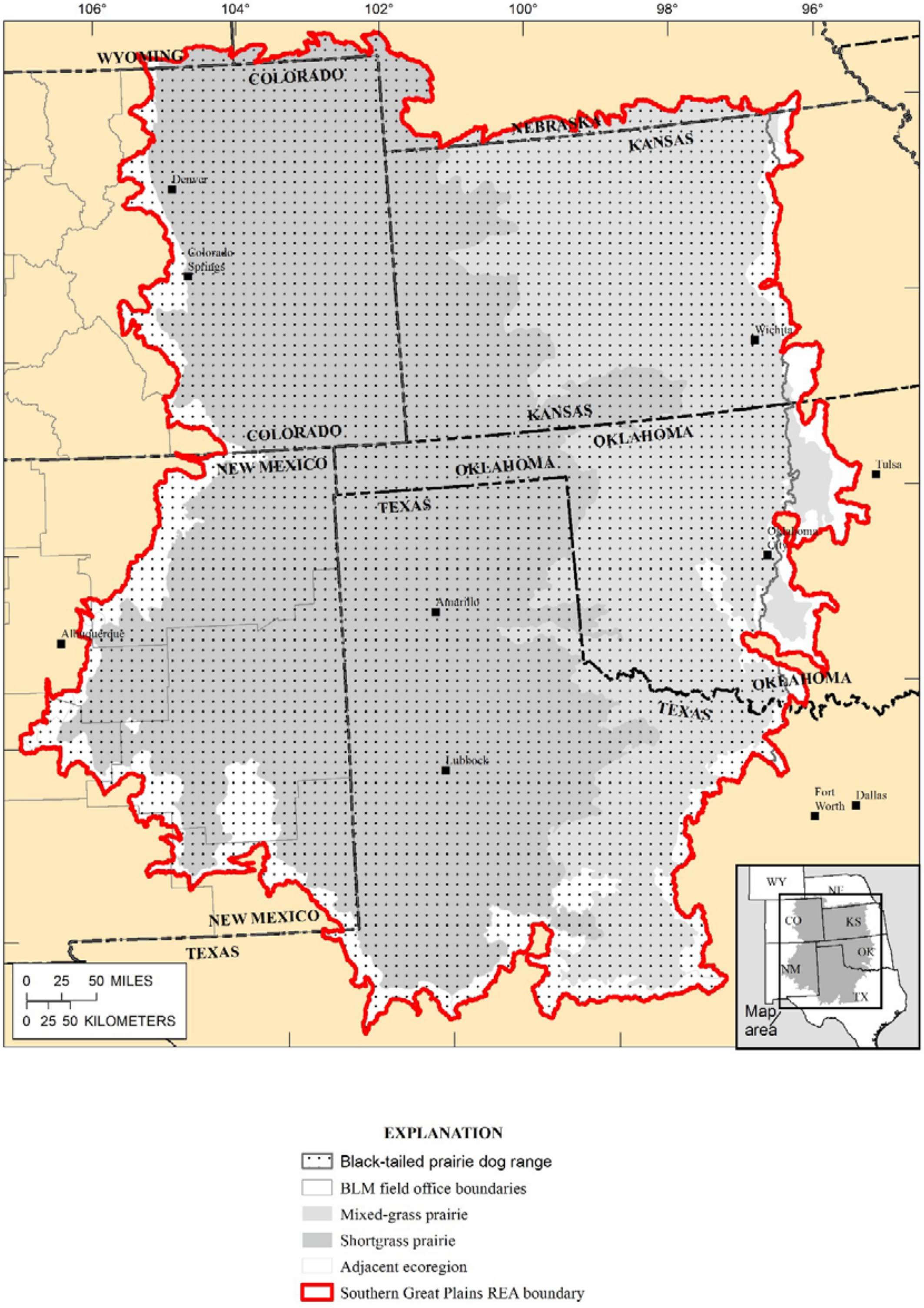

Figure 36. Current range of the black-tailed prairie dog (*Cynomys ludovicianus*) in the Southern Great Plains ecoregion (data source: U.S. Geological Survey National GAP Analysis Program). (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (BLM, Bureau of Land Management; REA, Rapid Ecoregional Assessment)

In addition to soil characteristics and slope, vegetation height is an important determinant of habitat suitability for BTPDs. These animals prefer an unimpeded view of their surroundings for early predator detection (Hoogland, 1995). Although BTPDs often clip the bases of tall plants (at least 15 cm high), trees and some shrubs are resistant to clipping and can grow to impede their views (Hoogland, 1995). If tall vegetation or other visual obstructions such as fences impede their views, BTPDs may experience increased rates of mortality or move elsewhere to avoid the obstructions (Franklin and Garrett, 1989). As in the case of other rodents, an impeded view also may lead to reduced foraging rates, body weight, and reproductive output (Bednekoff and Blumstein, 2009).

Vegetation is also the primary source of nutrients and water consumed by BTPDs. In the SGP, during spring and fall in particular, BTPDs commonly feed on various graminoids and forbs, including sand dropseed (*Sporobolus cryptandrus*), western wheatgrass (*Pascopyrum smithii*), blue grama (*Bouteloua gracilis*), and scarlet globemallow (*Sphaeralcea coccinea*) (Detling, 2006). During winter, BTPDs often feed on succulent cacti, such as pricklypear (*Opuntia* spp.). Although BTPDs are predominantly herbivorous throughout the year, they will consume insects, small mammals, and other prairie dogs; in fact, infanticide is common among BTPDs (Hoogland, 1995).

Colonies of BPTDs are essentially collections of harem-polygynous families, or coteries. During a single day in March–February, adult females are receptive to mating, which occurs in their burrows (Hoogland, 1995). Breeding females produce one litter per year, and young-of-year start to emerge aboveground during May–June, at which time they start foraging on vegetation. Males tend to disperse from their natal territories, whereas females usually exhibit a strong fidelity to their natal sites (Garrett and Franklin, 1988; Hoogland, 2013). Dispersal commonly occurs in June–August along the drainages of seasonal streams and roadways (Antolin and others, 2006).

There are three main sources of natural mortality among BTPDs: predation, infanticide, and winter mortality. A diversity of predators prey on BTPDs, including the black-footed ferret (*Mustela nigripes*), an endangered carnivore that consumes prairie dogs almost exclusively. Approximately 39 percent of BTPD litters are lost to infanticide, and overwinter survival tends to be low due to food shortages and associated declines in body condition (Hoogland, 1995). In general, BTPD populations grow slowly because survivorship among yearlings is less than 55 percent; it is only slightly higher in middle-aged classes. Furthermore, the percentage of individuals that mate as one-year-olds is only 6 percent for males and 35 percent for females, females can wean only one litter per year, the probability of weaning a litter is only 43 percent per year, and the average litter size at first emergence from the burrow is only 3.08 (Hoogland, 1995).

Landscape Structure and Dynamics

For conservation purposes, BTPD colonies are categorized into groupings across a landscape for maximizing the efficacy of management actions spatially and temporally. Groups of BTPD colonies separated by up to 7 kilometers are considered complexes (Biggins and others, 1993). Landscape dynamics within and around BTPD complexes are strongly influenced by precipitation, grazing, and fire, as well as interactive effects of these factors (Lauenroth and Burke, 2008). For example, droughts can increase the incidence of fires, thereby suppressing the expansion of shrubs in grasslands, including mesquite (*Prosopis* spp.) in Texas and New Mexico. Droughts and associated fire-induced suppression of mesquite can create open viewsheds, perhaps allowing BTPDs to expand and attain greater densities than they would if these disturbances did not occur (Augustine and others, 2007). Subsequent to disturbance, the function of BTPD foraging and clipping may be analogous to fire at certain spatial and temporal scales if they remove young shrubs (Kotliar, 2000). Furthermore, BTPDs create disturbances similar to those created by native ungulates (that is, grazing creates large patches of altered vegetation) (Whicker and Detling, 1988). These ecosystem processes are diminished if natural fires are suppressed by human actions (Lauenroth and Burke, 2008).

Although BTPDs can help structure grassland landscapes, various CAs such as plague have resulted in fragmented populations of BTPDs and fewer complexes to conserve (Lomolino and Smith, 2001). Indeed, metapopulations of BTPDs are composed of subpopulations with independent population dynamics that are subject to local extinctions and subsequent recolonization (Antolin and others, 2006). Recolonization is paramount to the persistence of metapopulations and depends on successful dispersal between colonies, and topographic variation, tall vegetation, areas of urban or agricultural development, and wetlands all function as impediments or barriers to BTPD dispersal (Johnson and Collinge, 2004; Antolin and others, 2006; Magle, Ruell, and others, 2010; Sackett and others, 2012).

At the scale of individual colonies, some studies suggest that more isolated colonies (greater distances between neighboring colonies) are more likely to persist than less isolated colonies (Lomolino and Smith, 2001; Johnson and others, 2011), possibly because isolation may protect BTPDs from plague, a highly lethal disease that was introduced to North America in 1900 (U.S. Fish and Wildlife Service, 2004, 2009). In another study, however, there was "no predictable relationship between extinction probabilities and intercolony distance, indicating that spatial isolation does not reduce the vulnerability of colonies to plague" (Stapp and others, 2004, p. 235).

Colony persistence is also sometimes greater among larger colonies than smaller ones, presumably because larger colonies are better protected from predators and stochastic events, and dispersing BTPDs are more likely to encounter big colonies because of their greater surface area (Hoogland, 1995; Lomolino and Smith, 2001; Snäll and others, 2008). If a colony size declines in size, however, it may become increasingly susceptible to extirpation due to small population effects, intensified rates of predation, and various CAs (Stapp and others, 2004). Both small and large colonies are sometimes decimated by plague epizootics (Johnson and others, 2011); thus, manipulation of colony size via poisoning or other management strategies is unlikely to affect the incidence of plague. At a finer scale, the dispersion of BTPDs in colonies changes over time, presumably because of spatial variation in precipitation, vegetative communities, predation, and disease (Jachowski and others, 2008). These spatial dynamics have important implications because plague is thought to percolate through patches of hosts and their fleas during epizootics (Davis and others, 2008). Moreover, spatial dynamics within colonies may have important implications for animals that associate with BTPDs, including species discussed below.

Associated Species of Management Concern

The BTPD is a keystone species that, with effective management, can maintain habitat conditions required by other species, such as clear viewsheds, nesting or denning sites and refugia (burrows), and high densities of certain vertebrate prey. In turn, BTPDs play a major role in the structure and function of grassland communities (Kotliar and others, 2006). Four species of management concern in the SGP associate closely with BTPDs—ferruginous hawk (*Buteo* *regalis*), western burrowing owl (*Athene cunicularia hypugaea*), mountain plover (*Charadrius montanus*), and black-footed ferret—and BTPD conservation efforts may benefit these species as well.

Ferruginous hawks (Chapter 13) often prey selectively on BTPDs (Plumpton and Anderson, 1997) and may undergo local population declines when BTPDs become scarce. In Colorado, a population of ferruginous hawks declined by about 89 percent subsequent to a plague epizootic among BTPDs during 1988–1989, then increased as the BTPD population recovered, and declined again by about 63 percent subsequent to another epizootic during 1994–1995 (Seery and Matiatos, 2000). Western burrowing owls (Chapter 12) commonly inhabit BTPD colonies, where they nest in prairie dog burrows. These owls do not excavate their own burrows; rather, they rely on prairie dogs and other ground squirrels to construct the burrows (Kotliar and others, 2006). Indeed, a study of 17 BTPD colonies in the Nebraska panhandle revealed a 63 percent decline in burrowing owl populations when poison was used to eradicate the prairie dogs in these colonies (Desmond and others, 2000). Predator detection among burrowing owls also might be enhanced in prairie dog colonies because BTPDs emit alarm calls when a predator is present (Bryan and Wunder, 2013). Furthermore, the vegetation within BTPD colonies is typically of short stature, which facilitates predator detection (Thiele and others, 2013). Mountain plovers (Chapter 17) prefer to nest in areas with sparse vegetation and open viewsheds, and BTPDs create ideal nesting sites for these ground-nesting birds by clipping tall vegetation (Knowles and others, 1982). Numbers of plovers can decline dramatically (sometimes to extinction at the colony scale) when BTPD populations decline, as found during epizootics of plague in Montana and Colorado (Augustine and others, 2008; Dinsmore and Smith, 2010).

The black-footed ferret is an endangered carnivore that preys almost exclusively on prairie dogs and dens in prairie dog burrows. A breeding family of ferrets may kill more than 750 prairie dogs in their home range annually (Biggins and others, 1993). The distribution of BTPDs affects space use and territoriality by ferrets, thereby affecting habitat carrying capacity for the species (Eads and others, 2014). Ferrets have been reintroduced to two BTPD sites in the SGP: one in the Vermejo Park Ranch, New Mexico, and another in Logan County, Kansas. The Vermejo population, however, is believed to be extirpated on BTPD habitat (D.H. Long, wildlife biologist, Turner Endangered Species Fund, oral commun., May 2014), whereas the Logan County population is prospering (T.M. Livieri, wildlife biologist, Prairie Wildlife Research, oral commun., May 2014). Proctor and others (2006) identified 22 “focal areas” (more than 4,000 ha each) in the SGP that could be managed for ferret reintroduction.

Change Agents

Development

Energy and Infrastructure

Energy development could reduce the amount of habitat available to BTPDs or fragment their colonies (U.S. Fish and Wildlife Service, 2009). Energy development also may affect BTPD densities (indexed as densities of active burrow openings) indirectly. In Wyoming, for example, the average number of active BTPD burrow openings in colonies with continued occupancy during 1995–1999 was relatively static in colonies near wind turbines but increased more than fourfold in colonies far from wind turbines (Johnson and others, 2000). Wind turbines are noisy when rotating, which may increase antipredator behaviors of BTPDs in nearby colonies and suppress movements away from their burrow openings when foraging (Rabin and others, 2006). Because vegetation is generally scarce around burrow openings, BTPDs near turbines may experience nutritional limitations (Biggins and others, 2012).

Urbanization

Urbanization can reduce the amount of habitat occupied by or available to BTPDs (U.S. Fish and Wildlife Service, 2009). For instance, in areas immediately east of the Front Range in Colorado, approximately 200,000 ha of potential habitat has been lost to urbanization (U.S. Fish and Wildlife Service, 2009), and the availability of habitat in the area continues to decline because of urbanization (Magle, Reyes, and others, 2010). An area of 200,000 ha represents the cumulative area of 50 focal areas that could be used for conserving BTPDs and associated species, like black-footed ferrets (Proctor and others, 2006).

In addition to reducing habitat availability for BTPDs and associated species, urban development is an important barrier to dispersal, which in turn reduces gene flow within BTPD metapopulations and rates of recolonization and increases extinction risk (Johnson and Collinge, 2004; Magle, Ruell, and others, 2010). Moreover, fragmentation can negatively affect species that associate with BTPDs. In Colorado, for example, the diversity and richness of bird species, and counts of many avian species, decreased in BTPD habitat fragmented by urbanization, especially fragments that had been isolated for longer periods (Magle and others, 2012). Although urbanization may not threaten the BTPD with extinction throughout its range (U.S. Fish and Wildlife Service, 2009), the amount of habitat lost to urbanization and its effects on gene flow and other species are substantial from an ecological perspective.

Agricultural Activities

Millions of acres of the western Great Plains have been converted to cropland, resulting in the destruction of BTPD colonies and loss of habitat (Forrest and Luchsinger, 2006). Croplands can be converted back to native grasslands and BTPDs can be translocated there (Long and others, 2006), but potential competition with livestock is often considered when making such management decisions. Although there is some evidence that livestock and BTPDs may compete for forage in some cases (Detling, 2006), plant species consumed by livestock are sometimes more abundant on BTPD colonies than at off-colony sites (Detling, 2006). Furthermore, although the potential effects of BTPDs on cattle have been emphasized historically, herbivory by domestic livestock can negatively affect prairie dogs and American bison (*Bison bison*) (Miller and others, 2007).

Shooting and Poisoning

Since the early 1900s, negative human perceptions toward BTPDs have stimulated poisoning campaigns, some of which been extensive. For example, during 1903–1912 in Colorado, poisons were used to kill 91 percent of BTPDs in the state, and at least 31 million more were poisoned during 1912–1923 (Forrest and Luchsinger, 2006). The U.S. Fish and Wildlife Service (2000) estimates that humans treat 10–20 percent of active BTPD habitats with poison each year. Poisoning of BTPDs is discouraged for multiple reasons. First, poisonous baits, such as zinc phosphide or chlorophacinone, can kill other animals that consume the baits, and nontarget scavengers can die upon consuming poisoned animals (Forrest and Luchsinger, 2006; Vyas and others, 2012). Second, fumigants, such as aluminum phosphide tablets and gas cartridges, can kill other animals that inhabit BTPD burrows (Forrest and Luchsinger, 2006). Third, poisoning of BTPDs can lead indirectly to declines in other species that use BTPD burrows, such as burrowing owls, because once the BTPDs are poisoned, the unmaintained burrows can collapse (Desmond and others, 2000). Fourth, the costs of buying and applying poisons are often greater than the perceived monetary benefit that might be gained by reducing BTPD densities (Miller and others, 2007). Last, wildlife managers encounter less opposition when they translocate rather than poison BTPDs (Lamb and others, 2006).

Many BTPD populations are also subjected to shooting. The number of BTPDs lost to shooting is substantial: throughout their range, about 2,000,000 BTPDs are shot annually (Reeve and Vosburgh, 2006). Shooting also negatively affects BTPDs in indirect ways. For example, when BTPDs were subjected to a pulse of shooting, surviving prairie dogs reduced their foraging time by 66 percent, with surviving adults exhibiting a 35 percent reduction in body condition and an 82 percent reduction in reproductive output (Pauli and Buskirk, 2007b). Moreover, shooting may stimulate BTPDs to disperse (Reeve and Vosburgh, 2006), and dispersing prairie dogs are especially susceptible to predators (Garrett and Franklin, 1988). These trends are important because BTPDs have relatively low rates of survival and reproduction, and when shooting is combined with other sources of mortality, populations can be extirpated or decline to such low levels that their ecological functionality is negligible (Reeve and Vosburgh, 2006; Miller and Reading, 2012).

Shooting also can affect species that associate with BTPDs. For instance, predators that prey on BTPDs may experience nutritional deficits if local BTPD populations are diminished by shooting. In addition, nontarget species are sometimes shot, including burrowing owls (Reeve and Vosburgh, 2006). Furthermore, burrows commonly collapse or are plugged by BTPDs when colonies are subjected to shooting, thus reducing the availability of refugia for species that use BTPD burrows (Biggins and others, 2012). If disposed of improperly, the carcasses of BTPDs that contain lead shot can be a source of lead poisoning throughout wildlife food chains (Reeve and Vosburgh, 2006). In a study during which expanding lead shot was used on BTPDs, the carcasses contained large numbers of bullet fragments that weighed less than 25 milligrams each, which is small enough to be ingested by and large enough to poison secondary consumers or BTPDs that scavenge on carcasses (Pauli and Buskirk, 2007a). Due to the direct and indirect effects of shooting on BTPDs and associated wildlife, proposals have been submitted for restricting the shooting of BTPDs (Miller and Reading, 2012), and until more is known about these dynamics, managers are advised to restrict shooting activities and enforce site closures, particularly where ferrets have been reintroduced (Reeve and Vosburgh, 2006).

Altered Fire Regime

Altered fire regimes in the SGP may negatively affect BTPDs in some cases. For instance, fire suppression can enhance shrub expansion on grasslands (Van Auken, 2000), thus reducing the availability of open viewsheds that are preferred by BTPDs. If woody shrubs become established in a given area and BTPDs attempt to colonize the area or are translocated there, the areas may attract predators, thereby inhibiting BTPDs from establishing residency (Long and others, 2006).

Diseases

Of the introduced species that may affect BTPDs, the plague bacterium *Yersinia pestis* is especially important. The BTPD is highly susceptible to this disease, which is now widespread throughout the western United States, including the SGP (Cully and others, 2006). Epizootic outbreaks of plague occur about every 5–14 years and kill 95–100 percent of prairie dogs in affected areas (Cully and others, 2006). Plague also reduces prairie dog densities during interceding, enzootic periods by causing chronic mortality in their populations (Biggins and others, 2010). Although the U.S. Fish and Wildlife Service (2009) has suggested that plague does not pose an imminent threat to the existence of BTPDs throughout their range, in some cases it can decimate entire complexes of prairie dog towns, thus causing extirpations across large landscapes (Cully and others, 2006, 2010). For instance, from 1989 to 1998 in Cimarron County, Oklahoma, plague contributed to a 75 percent decline in the total area occupied by BTPDs, from about 4,300 to 900 ha, and the average size of colonies declined by about 70 percent (Lomolino and Smith, 2001). From 2001 to 2005, during which 815 BTPD colonies were monitored in areas of Colorado, Kansas, New Mexico, Oklahoma, South Dakota, Texas, and Wyoming (Cully and others, 2010), colonies at the sites affected by plague were smaller, distances between neighboring colonies were greater, and the proportion of potential habitat actively occupied by prairie dogs was smaller. In a portion of northwestern Texas, 43 percent of the cumulative area inhabited by BTPDs was reduced (Cully and others, 2010).

A recent plague epizootic in the Conata Basin, South Dakota, demonstrates how the disease affects BTPDs when it first arrives in a new area. The first known plague outbreak at this site occurred in 2008, and the total area occupied was reduced by about 50 percent from 12,695 ha to 6,445 ha (R.L. Griebel, wildlife biologist, U.S. Forest Service, oral commun., November 2009). The remaining, contiguous habitat has been treated annually with insecticides to kill fleas. Otherwise, plague likely would have eliminated BTPDs from a much larger proportion of the site.

In addition to reducing the area occupied by BTPDs, plague causes dramatic oscillations in their abundance and inhibits them from serving their functions as keystone species and ecosystem engineers (McDonald and others, 2011; Miller and Reading, 2012). For example, the population peaks of BTPDs that once might have created “boom years” in predator populations may now be subdued by plague, thus resulting in declines of predators that hunt prairie dogs (McDonald and others, 2011). This includes black-footed ferrets, which also are highly susceptible to plague infection and mortality (Matchett and others, 2010). Furthermore, plague alters trophic relationships that can lead to reduced abundances of ferruginous hawks, burrowing owls, mountain plovers, and other species (Seery and Matiatos, 2000; Biggins and Kosoy, 2001; Dinsmore and Smith, 2010).

Climate Change

Some climate models project longer periods of more frequent drought in the SGP (Knapp and others, 2008), which could have important implications for BTPDs. During a drought year in northeastern New Mexico, BTPDs suffered a 98 percent reduction in reproductive output (D.A. Eads and D.E. Biggins, wildlife biologists, U.S. Geological Survey, unpub. data, 2010–2012). Similarly, during a relatively dry year in Chihuahua, Mexico, BTPDs lost weight, experienced increased rates of mortality, and produced fewer offspring (Facka and others, 2010). Overall, there is strong evidence that droughts limit BTPD populations, perhaps especially in the arid

SGP (Garrett and others, 1982; Davidson and others, 2010; Avila-Flores and others, 2011; Lloyd and others, 2013). Drought also may inhibit these rodents from serving their ecological functions as keystone species, thus resulting in the degradation of grasslands (Martínez-Estévez and others, 2013).

Climate change also may interact with other CAs to affect BTPDs. For instance, one ecological model suggests that contemporary climate change and the resulting increase in temperatures will reduce the abundance of fleas, suggesting the occurrence of plague epizootics in BTPDs will decline in the future (Snäll and others, 2009); however, recent data from New Mexico (D.A. Eads and D.E. Biggins, wildlife biologists, U.S. Geological Survey, unpub. data, 2010–2012) demonstrate that fleas can attain high densities during drought years, presumably because the climate in BTPD burrows mediates the effects of drought on fleas (Ben Ari and others, 2011). Also, when subjected to both drought and plague, BTPD densities may remain chronically suppressed, which would complicate conservation management of BTPDs and associated species (Miller and Reading, 2012).

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting the black-tailed prairie dog is illustrated in figure 37. Ecological attributes and CAs identified in the Rapid Ecoregional Assessment are enumerated in tables 41 and 42.

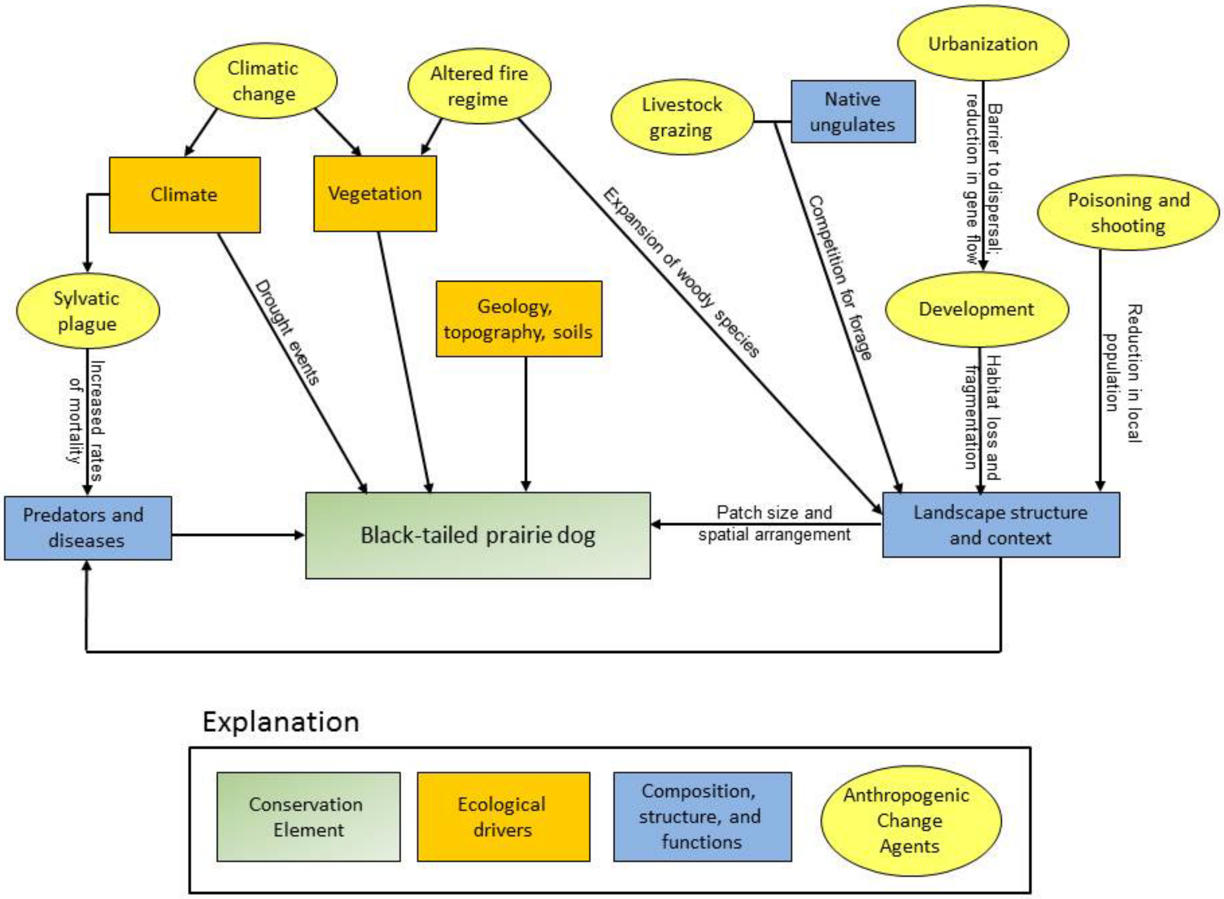

Figure 37. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for the black-tailed prairie dog (*Cynomys ludovicianus*) in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of black-tailed prairie dog populations and habtiats are shown in orange rectangles (see also table 41); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 42).

Table 41. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for the black-tailed prairie dog (*Cynomys ludovicianus*).

Attribute	Variables
Amount and distribution	Black-tailed prairie dog distribution (colony or complex location, size, and configuration).
Landscape structure	Size and spatial distribution of habitat (patch size, connectivity (black-tailed prairie dog distribution).
Landscape dynamics	Habitat productivity (food resources), drought, fire, shrubland-grassland ecotone dynamics, predator dynamics.
Associated species management of concern	Ferruginous hawk, burrowing owl, mountain plover, black-footed ferret, swift fox.

Table 42. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for the black-tailed prairie dog (*Cynomys ludovicianus*).

Attribute	Variables
Development (energy and infrastructure)	Habitat alteration (direct loss of habitat), habitat fragmentation and reduction in habitat connectivity, effects on antipredator and foraging behaviors.
Development (urbanization)	Habitat alteration (direct loss of habitat), habitat fragmentation and reduction in habitat connectivity (barrier to dispersal, reduction of gene flow).
Development (agricultural activities)	Habitat alteration (direct loss of habitat due to cultivation), habitat fragmentation and isolation (reduction in habitat connectivity), competition for forage with livestock and native ungulates.
Poisoning and shooting	Reduction in local population and ecosystem services, negative effects on body condition and reproductive output of surviving animals, stimulated dispersal (increase in predation), portal for poisons and lead to enter wildlife food chain.
Altered fire regime	Fire suppression results in habitat alteration (expansion of woody species, decrease in landscape heterogeneity).
Invasive species	Habitat alteration (shrub encroachment and increase in woody plants).
Introduced diseases	Presence and effect of plague on populations and metapopulations, reductions in ecosystem services and functioning.
Climate change	Drought effect on animal condition (reduction in body weight, increase in mortality, decrease in reproductive output), habitat condition (reduction in amount and quality of food resources [vegetation] due to drought), interactions with plague and other Change Agents.

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Chapter 21. Mule Deer

By Cynthia P. Melcher

Key Ecological Attributes

Distribution and Ecology

The mule deer (*Odocoileus hemionus*) is widely distributed throughout western North America from the south-central Alaskan coast and south-central Yukon Territory south through Baja California and the Central Plateau of Mexico, and east from the Pacific coast to about the 100th meridian. Their range is limited primarily by environmental factors that exert strong effects on population dynamics, such as prolonged cold winters, deep snow, and drought (Wallmo, 1981). In the SGP ecoregion, mule deer are distributed across the northwestern half of the ecoregion (fig. 38), although these animals range widely and small numbers of them can occur elsewhere in suitable habitat. There are up to 10 subspecies of mule deer. The predominant subspecies in the SGP is the Rocky Mountain mule deer (*O. h. hemionus*), although the desert mule deer (*O. h. eremicus* [formerly *O. h. crooki*]; Heffelfinger, 2000) range overlaps the SGP in the Texas panhandle and extreme southwestern corner of the SGP (Anderson and Wallmo, 1984; Heffelfinger, 2000). Because little information specific to mule deer in the SGP ecoregion has been published, our discussions herein pertain primarily to the species overall; however, we have highlighted information specific to Rocky Mountain and desert mule deer subspecies in and adjacent to the SGP states when it was available.

Most populations of Rocky Mountain mule deer are migratory because heavy snows restrict their movements, including those along the Front Range of Colorado (Kufeld and others, 1989) and in north-central New Mexico (Bender and others, 2007). Snow depths of 0.25−0.3 m reduce mule deer movements, and if depths increase to 0.5−0.6 m, mule deer are likely to seek other areas of their range with less snow (Anderson and Wallmo, 1984). Most desert mule deer, however, are nonmigratory, although some populations make seasonal movements in response to drought and rainfall (that is, water availability) (Wallmo, 1981; Rautenstrauch and Krausman, 1989). Both resident and migratory populations show strong fidelity to discrete home ranges and migratory routes (Julander and others, 1961; Wallmo, 1981; Kufeld and others, 1989; Relyea and others, 2000; Sawyer and Kauffman, 2011; Brunjes and others, 2013).

Mule deer are remarkably adaptable and occur in most major biomes of western North America, including alpine, forest, woodland, shrubland, desert scrub, grassland, and riparian habitats (Anderson and Wallmo, 1984; Mule Deer Working Group, 2004). Of at least 60 natural vegetation types found west of the 100th meridian in the United States, they are found in all but a few, as long as there is ample forage, cover (thermal and escape), and water resources (Wallmo, 1981). There is a lack of published studies, however, on the specific habitat relationships of mule deer in the SGP. Habitat studies in adjacent ecoregions, or studies on other aspects of mule deer ecology in and near the SGP, indicate that Rocky Mountain and desert mule deer are generally found in broken terrain with ample cover and woody browse, succulent forbs, and nutritious grasses (Wallmo, 1981; Fitzgerald and others, 1994; Relyea and others, 2000).

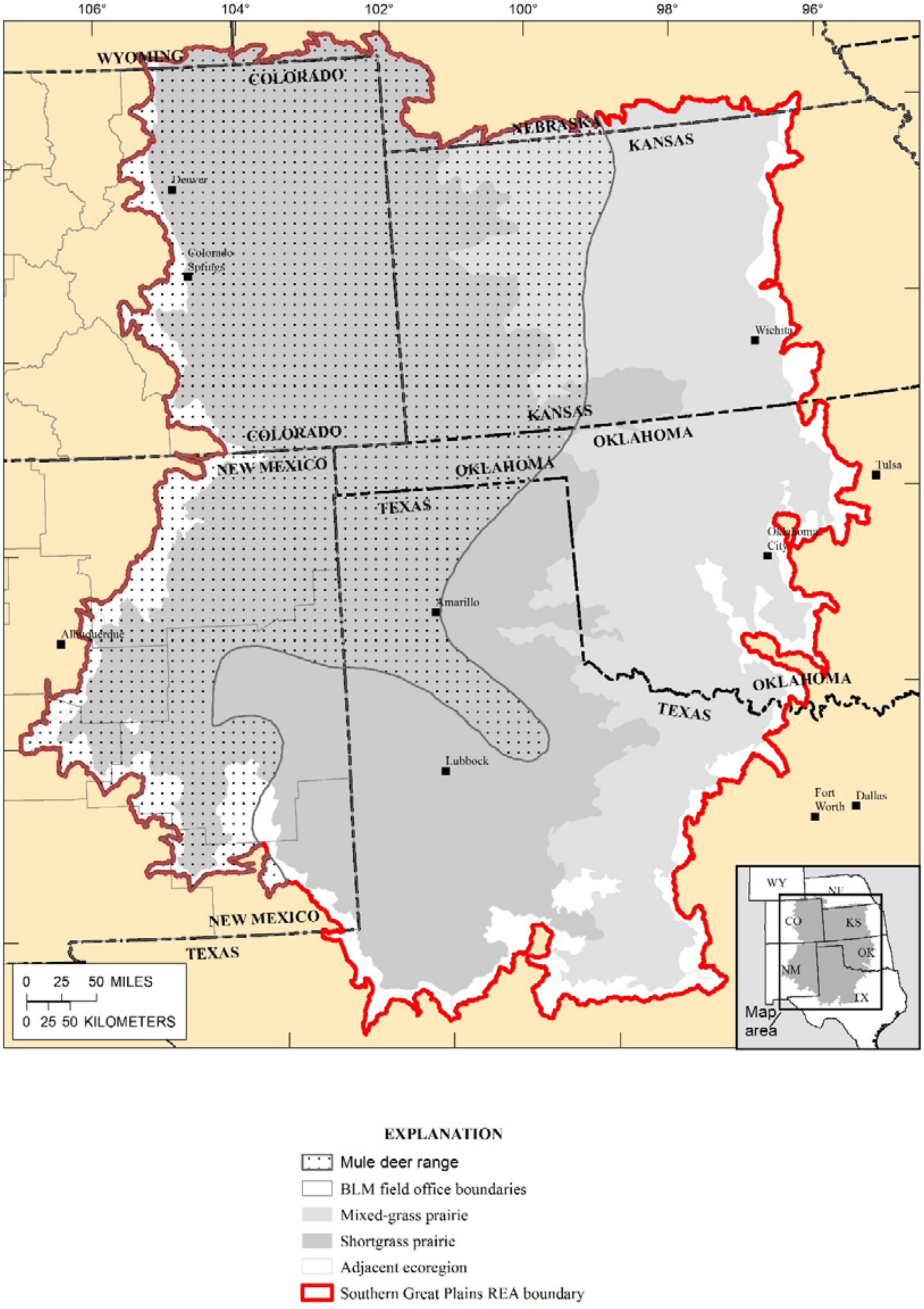

Figure 38. Distribution of mule deer (*Odocoileus hemionus*) within the boundary of the Southern Great Plains Rapid Ecoregional Assessment (data source: Feldhamer and others, 2003). (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (BLM, Bureau of Land Management; REA, Rapid Ecoregional Assessment)

In or near the SGP, habitats known to be used by or likely to meet the needs of mule deer include shrublands and woodlands of foothills, rolling hills, buttes, mesas, and plateaus; savannas and woody draws or canyons embedded in grassland and sandsage contexts; riparian corridors and other woodlands or shrublands associated with wetlands; desert scrub of broad valleys and escarpments; and hayfields, pasturelands, and small-grain fields (Wiggers and Beasom, 1986; Fitzgerald and others, 1994; Relyea and others, 2000). In eastern Colorado, mule deer densities are greatest in Front Range foothills shrublands (Fitzgerald and others, 1994). Examples of where mule deer are found elsewhere in the SGP are the Canadian River corridor (Sowell and others, 1985); Picket Wire Canyon in the Comanche National Grassland (U.S. Forest Service, 2013); Palo Duro Canyon on the Llano Escatado (Simpson and Gray, 1983; Texas Parks and Wildlife, 2013) and Sugarite Canyon at Raton Mesa (New Mexico Department of Game and Fish, 2013); the Edwards Plateau (Brunjes and others, 2013); and in wildlife management areas characterized by rolling hills with islands of woody vegetation and (or) subshrubs in a grassland context (Kansas Department of Wildlife, Parks, and Tourism, 2013; Oklahoma Department of Wildlife Conservation, 2013).

A crucial factor controlling mule deer survivorship and fecundity is nutritional status. In turn, nutritional status depends on forage quantity and quality, including crude protein content and digestibility (Anderson and Wallmo, 1984). A study of mule deer in north-central New Mexico, where densities have declined over the past several decades, revealed that starvation was the greatest cause of winter mortality in two of three years (Bender and others, 2007). Winter mortality due to starvation among fawns and older adults can be especially high. Nutritional status also can affect fetal and neonatal survival (Julander and others, 1961; Bishop and others, 2009). Because mule deer breed in the late fall and early winter, poor summer range conditions also may result in low ovulation rates (Julander and others, 1961). A secondary effect of poor body condition due to malnutrition is greater susceptibility to predation (Bender and others, 2007).

Mule deer are generally characterized as browsers of woody plants (shrubs, small trees, and subshrubs), but to meet all their nutritional requirements they require a significant component of forbs and grasses (including rushes and sedges) in their diet (Anderson and Wallmo, 1984). A review of 99 food habit studies for Rocky Mountain mule deer (all but 8 outside of the SGP) indicated that diet varied not only by season and region, but by plant species availability, deer demographics, and data-collection methods (such as behavioral observations versus stomach analyses) (Kufeld and others, 1973).

There is some debate over the extent to which mule deer need access to freestanding water (Simpson and others, 2011), likely because of differences in gender/age, season, forage quality, and trends in climate conditions. Water needs increase not only when temperatures are high, but also when energetic demands are high, such as during the rut or when does are lactating (Boroski and Mossman, 1998) or when forage quality is poor or does not meet water requirements (Simpson and others, 2011). A study of desert mule deer in southwestern Arizona indicated that availability of freestanding water can influence seasonal mule deer movements and home range size (Rautenstrauch and Krausman, 1989). In desert regions, mule deer will consume succulent plants to help meet their water needs (Wallmo, 1981), although this may be at the expense of crucial nutrients (Simpson and others, 2011), and during winter they may meet their water needs by eating snow if available (Wallmo, 1981). In regions where water may be limiting, management efforts for mule deer often focus on making enhancements to streams, springs, and natural tanks and installing artificial sources of water (Boroski and Mossman, 1998; Simpson and others, 2011).

Landscape Structure and Dynamics

Reports of mule deer home range size have varied from 49 to 3,379 ha, depending on subspecies, season, age/gender, body size/condition, reproductive status, habitat conditions, and methods used to estimate range size (Anderson and Wallmo, 1984; Relyea and others, 2000; Sánchez-Rojas and Gallina, 2000; Kie and others, 2002; Brunjes and others, 2013). Some of the larger home ranges reported for Rocky Mountain mule deer have come from the northern Great Plains (Severson and Carter, 1978; Wallmo, 1981) and more arid regions (Rodgers and others, 1978; Brunjes and others, 2013) of their range, where resources are more limiting. In the foothills west of Fort Collins, Colorado, average mule deer home range size was 211 ha (range 172–292 ha) (Kufeld and others, 1989). In west-central Texas, where density was 0.24 deer per ha, mean home range size was 390 ha in spring and 282 ha in summer (Brunjes and others, 2013). Landscape structure also strongly influences mule deer habitat use and home range size. Metrics of habitat heterogeneity at a coarse scale are important predictors of home range size, underlining the importance of habitat heterogeneity in the distribution of mule deer (Kie and others, 2002). In a northern California study, home ranges tended to increase as the heterogeneity of cover types increased, while home range sizes decreased in larger patches with high-contrast edges (such as woodland versus bare ground) (Kie and others, 2002). In Kansas, however, deer densities are greatest in the extreme northeastern edges of the SGP, where it is speculated that rough terrain excludes most hunters.

Variation in snow cover, rainfall, drought, and habitat productivity can influence mule deer movements (Wallmo, 1981; Sawyer, Kauffman, Nielson, and Horne, 2009). Male mule deer often move beyond their home ranges during the rut (generally late fall through late winter), particularly if home ranges of females are large. Dispersal movements (usually young males after their first winter) have been reported to range from at least 5 km to more than 200 km, with young males moving farther (Anderson and Wallmo, 1984); distances moved are likely influenced by habitat productivity. Snow reduces mule deer movements, and deep snow usually forces mule deer to seek other areas of their range with less snow (Anderson and Wallmo, 1984). Similarly, even where winters are mild, seasonal drought and rainfall patterns may drive migratory movements (Anderson and Wallmo, 1984).

Associated Species of Management Concern

Mule deer are important prey for several predator species, mountains lions in particular (Fitzgerald and others, 1994). Coyote predation on mule deer varies by region, depending on local predator communities. Generally, smaller predators, such as young or female mountain lions (*Puma concolor*), coyotes (*Canis latrans*), and bobcats (*Lynx rufus*), can take mule deer fawns and adults in poor condition. Larger carnivores, such as adult male mountain lions, are capable of taking adult mule deer in good condition.

Because mule deer consume vast quantities of vegetation, including browse species, their foraging may check the advance of seral stages and expansion of woody vegetation, thus maintaining or promoting spatial habitat heterogeneity. By the same token, if local mule deer populations approach or exceed carrying capacity, they may set back the vegetation community to early seral stages. Their fecal and urine output also help to recycle nutrients within their habitats.

Change Agents

Primary native factors that directly influence mule deer population size and distribution, survivorship, and rate of growth include habitat quality (availability, productivity, and quality of forage) and predation pressure. In turn, habitat quality is strongly influenced by short- and long-term climate trends, patch heterogeneity, and terrain. Anthropogenic CAs include habitat loss and fragmentation due to development and agriculture, including livestock grazing; altered fire regimes; changes in climate; introduced diseases; altered predator communities; and hunting pressure (Mule Deer Working Group, 2004). Indeed, 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).

Development

Energy and Infrastructure

The SGP encompasses some of the most active areas for oil and gas drilling (U.S. Energy Information Administration, 2011), as well as some of the best areas for wind energy potential, in the United States. Nonetheless, studies of energy development effects on mule deer in SGP are lacking, and there are few data on how wind energy affects big game in general (Anderson, 2010). (There was, however, a recent study of female mule deer spatial ecology along the Wyoming-Colorado border in an area of proposed wind energy development [Webb and others, 2013], which will provide a baseline for comparison once the wind farms are developed.) In the Wyoming Basin, it was recently found that wintering Rocky Mountain mule deer avoided all types of gas well pads and moved farther away from well pads associated with greater traffic volume than those with less traffic (Sawyer, Kauffman, and Nielson, 2009). In a related study, migratory behavior of mule deer varied with the intensity of both energy and exurban development (Sawyer, Kauffman, Nielson, and Horne, 2009). Although moderate levels of development do not appreciably affect mule deer migration patterns, higher levels of development may lead mule deer to detour from traditional routes, thus increasing their rates of movement and reducing their time spent at crucial stopover points where they can forage (Sawyer, Kauffman, Nielson, and Horne, 2009). Energy development, including oil/gas and wind, in portions of the SGP are contributing to habitat fragmentation and potential loss of traditional movement or migration routes.

Roads can result in significant disturbance and mortality to mule deer. In a review of state records across the United States, a conservative estimate of number of deer (*Odocoileus* spp.) killed on highways was 500,000 in 1991, with an increase during the 10 years prior to that (Romin and Bissonette, 2013). Although it is not clear whether highway mortality has negative effects on mule deer populations and demographics, it may be a greater concern where populations are already undergoing precipitous declines because of other causes. Perhaps a more important effect of roads is mule deer avoidance of them, which effectively results in habitat loss. During winter in Colorado, mule deer avoided using habitat up to 400 m from roads (on the East Slope, avoidance was particularly notable 0–200 m from the road), especially in shrubland habitats as opposed to pine forests and juniper woodlands, which may have provided more visual screening (Rost and Bailey, 2013). Because shrubland habitat is crucial for wintering mule deer, avoidance of roads could reduce the effective availability of habitat. Another concern for mule deer is off-highway vehicle traffic, which is increasing rapidly throughout the West (Ouren and others, 2007).

Agricultural Activities

Agriculture is a large industry in the SGP, but the extent to which agricultural croplands overlap historical or current mule deer range is not known. Undoubtedly croplands have resulted in some mule deer habitat loss in native grasslands that surround woody draws and other areas with vertical cover, as well as in riparian floodplains. Although mule deer forage more on browse and forbs (Kufeld and others, 1973), at times they forage heavily in agricultural lands, including cereal crops, alfalfa, and pasture grasses, which may partially offset any losses of food resources (but not escape or thermal cover) to agriculture. The presence of high-quality crops and pasture can influence the timing of seasonal deer movements and can be utilized in early spring when nutritional status is poor and energy demands (pregnancy) are high (Garrott and others, 2013).

Chronic, heavy grazing by livestock and high densities of deer and other wild ungulates have been reported to reduce forage quantity/quality and alter vegetation communities for mule deer in many parts of the species' range (Julander and others, 1961; Vavra and others, 2007), leading to shifts in habitat use by mule deer (Loft and others, 2013). In the Pacific Northwest, Vavra and others (2007) found that chronic, heavy grazing may favor community dominance by unpalatable plants. Additionally, browse species often produce more secondary plant compounds and material of low digestibility (lignin, cellulose) when chronically browsed, which in turn reduces overall palatability, digestibility, and nutritional value (Vavra and others, 2007). In the Sierra Nevada, female mule deer enlarged their home ranges in response to moderate-to-heavy cattle grazing, and grazing reduced mule deer cover (Loft and others, 2013). Grazing can also promote invasions of nonnative plants (see below).

Altered Fire Regime

Natural fire regimes in mule deer habitat of the SGP are poorly understood, in part because many areas are devoid of trees (lack of fire scars). However, both fire suppression and the resulting increasing frequency and intensity of fire are believed to be affecting habitats throughout much of the mule deer's range (Baker, 2011; Clements and Young, 2013). Overall, the complexity of fuel types and structure and the high variability in annual precipitation in primary mule deer habitats of the SGP likely supported a wide range of fire frequencies and intensities.

Fire in mule deer habitat sets back seral stages, opens the denser habitats, recycles nutrients, increases the nutritional value of new vegetative growth, and promotes species diversity. In the grassland uplands and grassland mosaics, fire exclusion can permit mulches to accumulate, shade out (exclude) warm-season species, and promote cool-season species (Samson and Knopf, 1996). The extent to which such dynamics affect important forbs consumed by mule deer requires further study. Overall, small fires of low-to-medium intensity likely promoted development of the landscape metrics that have been found to correlate with smaller mule deer home range size (Kie and others, 2002).

Effects of fire on mule deer nutrition have received some attention. Fire, however, can improve the nutritional quality of mule deer winter diets under certain circumstances. In a study of prescribed fire (late fall) effects on mule deer nutrition in foothills grassland and mountain shrubland communities west of Fort Collins, Colorado, fires were more intense and homogeneous in the mountain shrub communities (Hobbs and Spowart, 1984). In the same study, the overall change in quality (crude protein content and digestibility) of forages in burned plots was relatively small, but the effect of burning on overall nutritional quality of mule deer winter diets in the first year was substantially increased. Hobbs and Spowart (1984) attribute these effects to the postburn sprouting of cool-season grasses, which were obscured by ground litter in unburned plots, and the enhanced warmth of blackened, bare soils likely stimulated winter growth.

Invasive Species

In the SGP and elsewhere, the extent to which invasive species (plants, in particular) affect mule deer habitat, home range use, or diet is poorly understood. Above, we discussed the interactive effects of grazing with invasive species, highlighting the advantages conferred to many invasives by heavy grazing (Vavra and others, 2007). Once established, nonnative and invasive plants can supplant native communities and alter the functions and (or) values of and disturbance process in a given habitat (Baker, 2011; Clements and Young, 2013). If invasives promote larger, more intense fires, it is likely that habitat heterogeneity would result, which could force mule deer to enlarge their home ranges.

Some nonnative plants, however, can be more palatable to ungulates than their native counterparts. During a study of captive mule deer in Utah to evaluate their preference for 16 largely nonnative grasses, more cheatgrass (*Bromus tectorum*) was consumed than other grasses during spring and fall (Austin and others, 1994). Overall, nonnative wheatgrasses (*Agropyron* spp.), orchardgrass (*Dactylis glomerata*), and cheatgrass composed 80 percent of their spring diet and 48 percent of their fall diet. In the same study, deer avoided native basin wildrye (*Leymus cinereus*) and two varieties of nonnative Russian wildrye (*Psathyrostachys juncea*). The extent to which wild mule deer might favor native versus nonnative grasses or other plants, much less the way in which they may affect home range size, is not well known.

Insects and Disease

Mule deer are susceptible to a wide range of diseases and parasites that can be a primary cause of mortality or predispose them to mortality from other causes (Wallmo, 1981). Mule deer can contract some diseases through contact with livestock (particularly cattle, sheep, and swine), such as foot-and-mouth disease (Wallmo, 1981) and malignant catarrhal fever (Schultheiss and others, 2007). Heavy parasite loads are indicators of crowding and result in poor body condition (Wallmo, 1981). A current disease of significant concern in north-central Colorado is chronic wasting disease, where 15 percent of the mule deer population may be affected (Gross and Miller, 2001). In addition to Colorado, chronic wasting disease is currently found in the SGP ecoregion in western Kansas (Centers for Disease Control and Prevention, 2013). The disease also is found in southern New Mexico and western Texas but in counties outside of the SGP ecoregion.

Several diseases that occur seasonally have the potential to increase with climate change. Bluetongue and epizootic hemorrhagic disease are both viral diseases of mule deer transmitted primarily by biting midges (*Culicoides sonorensis*); thus, longer summers could potentially increase the incidence of disease by enhancing rates of reproduction and survivorship of midges (Wallmo, 1981; Schmidtmann and others, 2011).

Climate Change

By 2100, average temperatures in the SGP are projected to increase (Karl and others, 2009). Projections for 2009–2099 also indicate that annual average precipitation may decrease in the SGP, especially in the southwestern portion of the region (Karl and others, 2009). The greatest differences are projected to occur in spring and summer during the growing season, the time of year when mule deer need to forage heavily and lay on fat to help them survive the following winter. If prolonged drought is accompanied by higher temperatures, habitat productivity is likely to decline, forcing mule deer to enlarge their home ranges and (or) make seasonal movements to ensure access to freestanding water or more productive habitat. A study of desert mule deer in west Texas indicated that drought was a major factor driving variation in survivorship, particularly among females and fawns/yearlings (Lawrence and others, 2004). As drought reduces the productivity of forage and cover, deer are more likely to be in poor condition and more exposed. In turn, this leaves mule deer considerably more susceptible to predation and disease.

Rapid Ecoregional Assessment Components

A general conceptual model for the key ecological attributes and CAs affecting the mule deer is illustrated in figure 39. Key ecological attributes and CAs identified in the REA are described in tables 43 and 44.

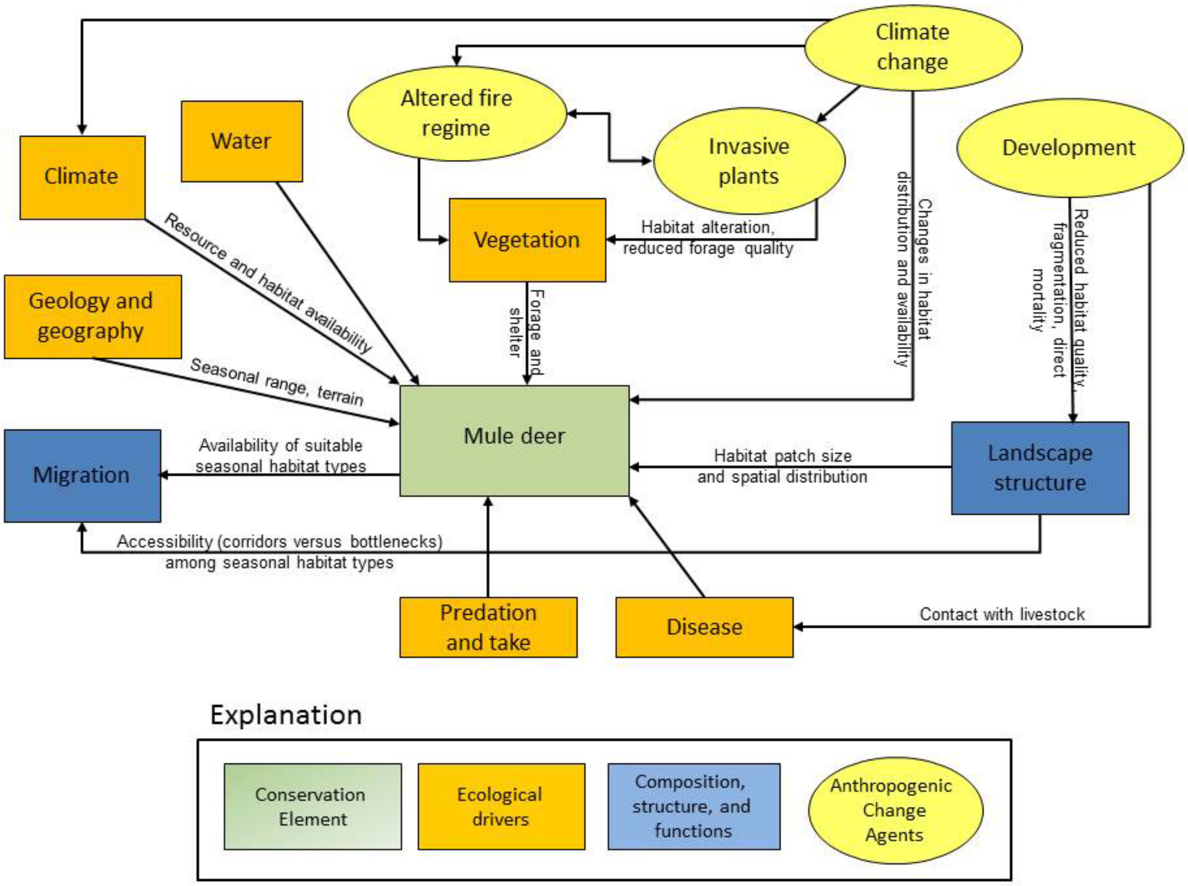

Figure 39. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for the mule deer (*Odocoileus hemionus*) in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of mule deer populations and habtiats are shown in orange rectangles (see also table 43); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 44).

Table 43. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for the mule deer (*Odocoileus hemionus*).

Attribute	Variables
Amount and distribution	Mapped mule deer distribution, habitat type (winter, summer, parturition), forage quality, water availability.
Landscape structure	Size and spatial distribution (patch size, amount of edge, shape index, connectivity).
Landscape dynamics	Seasonal snow cover and snow depth, time since fire, habitat productivity.

Table 44. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for the mule deer (*Odocoileus hemionus*).

Attribute	Variables
Development (energy and infrastructure)	Habitat alteration (direct loss of habitat), habitat fragmentation and reduction in habitat connectivity, direct displacement of animals, avoidance, increase in human interactions (mortality).
Development (agricultural activities)	Habitat alteration (direct loss of habitat, changes to vegetation communities, reduction in seed production), habitat fragmentation (reduction in habitat connectivity), forage quality (decrease in plant palatability and nutritional value).
Altered fire regime	Habitat alteration through changes to fire frequency, increase in fire frequency (decrease in shrubland, loss of winter cover, increase in exotic species, increase in forage quality) or decrease in fire frequency (expansion of woody plants, reduction in forage quality).
Invasive species	Habitat alteration (changes in plant community, reduction in forage quality, altered disturbance regime).
Insects and disease	Presence and effect of chronic wasting disease on population, contact with livestock.
Climate change	Habitat alteration (changes in plant community, reduction in forage quality), animal behavior (enlarged home range, changes in season movement), animal condition (increased stress).

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Chapter 22. Swift Fox

By Natasha B. Carr and Cynthia P. Melcher

Key Ecological Attributes

Distribution and Ecology

The swift fox (*Vulpes velox*) is native to the shortgrass and mixed-grass prairies of the North American Great Plains and the grassland-steppe ecotones along the western and southern peripheries of its range (fig. 40) (Egoscue, 1979; Olson and Lindzey, 2002a; Harrison and Schmitt, 2003; Gese and Thompson, 2014). The estimated historic range of the swift fox, which is based on both historic records and the distribution of short-structured grasslands prior to Euro-American settlement, extended from Alberta, Saskatchewan, and Manitoba southward to New Mexico and Texas (Sovada and others, 2009). The Rocky Mountains form a geographic barrier that separates swift fox populations in the east from the morphologically similar kit fox (*Vulpes macrotis*) to the west (Dragoo and others, 1990; Mercure and others, 1993). A hybrid zone occurs where the distributions of the species meet in New Mexico and Texas along the southwestern border of the SGP ecoregion (Mercure and others, 1993). Although genetic analyses initially provided justification for combining swift and kit foxes into one species (Dragoo and others, 1990), more recent genetic studies provide support for treating them as two distinct species (Mercure and others, 1993).

Historically, the swift fox likely was common throughout the Great Plains of the United States and Canada but suffered severe declines resulting from the conversion of native grasslands to cropland, trapping for the fur trade, and indiscriminant poisoning of gray wolves (*Canis lupus*), coyotes (*Canis latrans*), and prairie dogs (*Cynomys* spp.) (Kilgore, 1969; Egoscue, 1979). In addition, altered grazing regimes resulting from the widespread replacement of bison and prairie dogs (Cid and others, 1991) with cattle may have contributed to the declines (Egoscue, 1979; Sovada and others, 2009). By the 1950s, the swift fox was largely or entirely extirpated from Montana, North Dakota, Nebraska, Oklahoma, and Canada, while populations were greatly reduced in South Dakota, Wyoming, Colorado, New Mexico, and Texas (Allardyce and Sovada, 2003; Sovada and others, 2009). Following the cessation of poisoning campaigns, swift fox populations began dramatic natural recoveries in the 1950s, especially in Colorado, Kansas, Oklahoma, New Mexico, and Wyoming (Sovada and others, 2009). Reintroductions were used to help restore the swift fox to its northern range (including Canada, Montana, and South Dakota), where recovery of populations lagged, with some released animals subsequently dispersing to other states (Soper, 1964; Allardyce and Sovada, 2003; Sovada and Carbyn, 2003; Russell, 2006; Cullingham and Moehrenschlager, 2013). Currently, the swift fox occupies approximately 40 percent of its estimated historic range (Sovada and others, 2009). The current distribution of swift foxes within the SGP is delineated in 40.

In 1995, the U.S. Fish and Wildlife Service concluded that listing the swift fox under the U.S. Endangered Species Act was warranted but precluded (Stephens and Anderson, 2005; (Sovada and others, 2009). Subsequently, the U.S. Fish and Wildlife Service removed the swift fox from the candidate list based on the conclusion that the species' habitat requirements were more flexible and the species was more abundant and widespread than previously thought

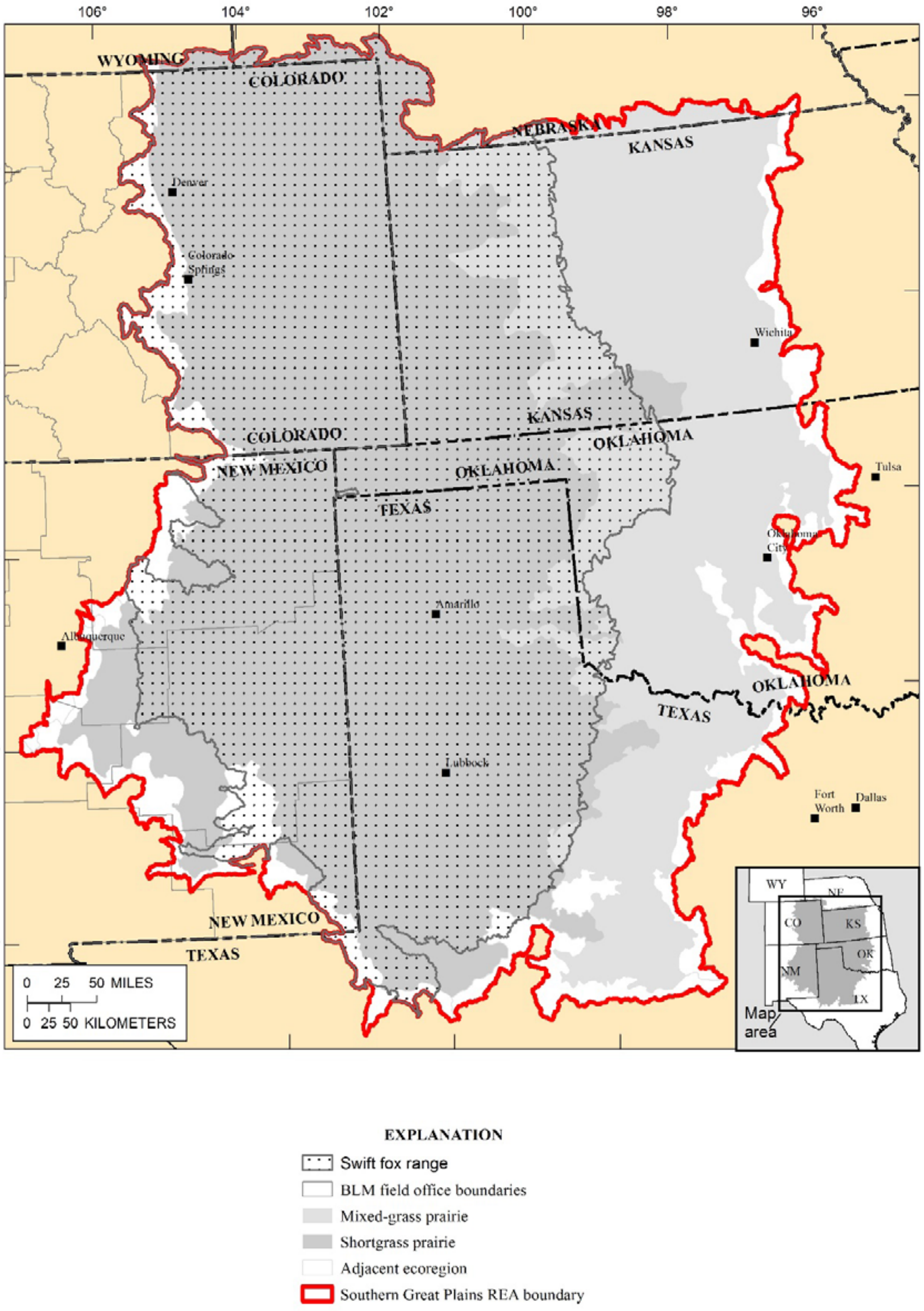

Figure 40. Current range of the swift fox (*Vulpes velox*) in the Southern Great Plains ecoregion (data source: U.S. Geological Survey National GAP Analysis Program). (Map developed by Jason Schmidt, Photo Science Inc., a Quantum Spatial Co.) (BLM, Bureau of Land Management; REA, Rapid Ecoregional Assessment)

(Stephens and Anderson, 2005). Currently, populations are apparently viable and stable or increasing in the core of the species' distribution, including Kansas, Colorado, Oklahoma, New Mexico, and Wyoming (Olson and Lindzey, 2002 a, b; Stephens and Anderson, 2005; Moehrenschlager and others, 2013). Elsewhere within their current range, however, populations are small and fragmented and may be vulnerable to decline (Moehrenschlager and others, 2013).

To avoid having to invoke the Endangered Species Act to protect the swift fox from further declines, wildlife management agencies and other organizations from the 10 states where the swift fox historically occurred formed a Swift Fox Conservation Team (Sovada and others, 2009). The conservation team developed a conservation assessment and a conservation strategy to coordinate monitoring and management of the swift fox and to address the following primary objectives: to maintain and protect extant populations; to identify, manage, and protect suitable swift fox habitat where feasible; and to work cooperatively with private land owners and land management agencies to maintain swift fox habitat and genetically connected populations (Dark-Smiley and Keinath, 2003; Stephens and Anderson, 2005). A number of recent, comprehensive and synthetic reviews provide additional details on the ecology and conservation of the swift fox at state and regional levels (including Kahn and others, 1997; Dark-Smiley and Keinath, 2003; Sorvada and Carbyn, 2003; Stephens and Anderson, 2005; Meyer, 2009).

Swift foxes typically inhabit areas with limited topographic relief dominated by grasses less than 30 cm high, including blue grama (*Bouteloua gracilis*), buffalograss (*Buchloe dactyloides* [formerly *Buchloe dactyloides*]), and needle-and-thread grass (*Hesperostipa comata*) (Meyer, 2009). Swift foxes also may use areas with scattered or low-density shrubs, including piñon-juniper, shinnery oak (*Quercus havardii*), mesquite (*Prosopis* spp.), tree cholla (*Cylindropuntia imbricata*), fourwing saltbush (*Atriplex canescens*), greasewood (*Sarcobatus vermiculatus*), and sagebrush (*Artemisia* spp.), especially along the periphery of their range (Olson and Lindzey, 2002 a, b; Dark-Smiley and Keinath, 2003; Thompson and Gese, 2007; Thompson and others, 2008; Meyer, 2009). Although swift foxes tend to use vegetation that is shorter and less dense than what is typically available, this is not always the case (Russell, 2006). It is generally assumed that short, open cover typical of swift fox habitat enhances visibility and detection of predators, coyotes in particular (Kitchen and others, 1999; Harrison, 2003; Sovada and others, 2009). Indeed, studies have shown that coyote predation on swift foxes is lower in areas with high visibility than in areas with low visibility (Olson, 2000; Russell, 2006).

Swift foxes use their dens year round and have a greater reliance on dens compared to other canids in North America (Kilgore, 1969; Egoscue, 1979; Jackson and Choate, 2000). Swift foxes are primarily nocturnal and crespuscular, thus they spend much of the day in their dens, which are used for pup-rearing, shelter, and escape from predators (Egoscue, 1979; Stephens and Anderson, 2005; Meyer, 2009). Characteristics of denning sites are generally similar to that available in their home range, but there is evidence that swift foxes may sometimes locate den sites in elevated areas with gradual slopes (Pruss, 1999; Jackson and Choate, 2000; Olson, 2000; Harrison, 2003; Kintigh and Anderson, 2005; Kitchen and others, 2006). The dens are excavated (or burrows of other species are modified) in a variety of friable soil types (Kilgore, 1969; Stephens and Anderson, 2005; Meyer, 2009).

Area requirements for swift foxes vary from about 760 to 3,200 ha; however, this wide variation likely has been influenced by methodological differences among studies (Stephens and Anderson, 2005; Meyer, 2009). Even within a given area, home ranges can vary seasonally and annually in response to prey availability or the presence of predators and competitors (Stephens and Anderson, 2005). Home-range size differs among the sexes (Olson and Lindzey, 2002a) and typically is smallest during the pup-rearing period (Schauster and others, 2002). There is also evidence that coyote predation (as opposed to avoidance of coyotes) can decrease overlap among the home ranges of swift foxes and coyotes. In some regions, however, their home ranges broadly overlap (Kitchen and others, 1999; Kamler, Ballard, Gilliland, and others, 2003).

Swift foxes are omnivorous; their diet includes a wide variety of seeds, berries, grasses, insects, reptiles, ground-nesting birds and eggs, mammals, and carrion (Olson, 2000; Sovada and others, 2001; Harrison, 2003). They are also considered opportunistic, as their diet varies geographically and seasonally with prey availability, but typically the dominant foods are small mammals and insects (Sovada and others, 2001; Stephens and Anderson, 2005; see also Meyer, 2009, for a list of common prey species). Swift foxes also cache food (Sovada and others, 2001; Dark-Smiley and Keinath, 2003). Prey availability often increases with shrub cover, although in these habitats foxes may have less success at detecting potential predators such as coyotes than they would in more open habitats (Meyer, 2009). Consequently, there may be trade-offs between foraging efficiency and predation risk (Thompson and Gese, 2007; Gese and Thompson, 2014). There is mixed evidence as to the requirement for access to permanent water near denning sites (Stephens and Anderson, 2005).

Swift foxes are relatively short-lived but have high reproductive potential (Stephens and Anderson, 2005). Estimated annual survival rates of swift foxes range from 40 to 75 percent for adults and 5 to 33 percent for juveniles (Stephens and Anderson, 2005; Meyer, 2009). Predation is typically the most common cause of swift fox mortality, and coyotes are the primary predators (Olson and Lindzey, 2002b; Schauster and others, 2002; Kamler, Ballard, Gilliland, and others, 2003; Thompson and Gese, 2007). Other predators of the swift fox include large raptors, American badgers (*Taxidea taxus*), and bobcats (*Lynx rufus*) (Schauster and others, 2002; Olson and Lindzey, 2002b; Ausband and Foresman, 2007). Disease, including canine distemper, appears to be a minor source of mortality (Olson and Lindzey, 2002b). Swift foxes are commonly infected by parasites, including ticks (Arachnida), fleas (Siphonaptera), and roundworms and hookworms (Nematoda) (Miller and others, 1998; Meyer, 2009).

Landscape Structure and Dynamics

Prior to Euro-American settlement, North American prairie grasslands were frequently exposed to disturbance from drought, fire, and grazing by native ungulates and prairie dogs, resulting in a shifting mosaic of habitat patches characterized by different species composition and vegetation structure that varied with intensity, frequency, and time since disturbance (Krueger, 1986; Cid and others, 1991; Brockway and others, 2002; Thompson and others, 2008; Sovada and others, 2009; Limb and others, 2011; Winter and others, 2011; Gese and Thompson, 2014). Such disturbances would tend to provide conditions suitable for the swift fox by maintaining shortgrass systems or leading to short-term reduction of cover in mixed-grass and tallgrass prairies and in shrub-steppe systems (Sovada and others, 2009; Gese and Thompson, 2014). Consequently, the extent of conditions suitable for the swift fox was likely spatially and temporally dynamic (Sovada and others, 2009). The boundaries between shortgrass and mixed-grass prairies are particularly dynamic, with drought favoring shortgrass prairie and abundant precipitation favoring mixed-grass and tallgrass prairie (Sovada and others, 2009). Indeed, historic records of swift foxes in tallgrass prairies may have been the result of drought or fire that reduced vegetation stature (Sovada and others, 2009). Such dynamics confound attempts to define the historic range of the swift fox (Sovada and others, 2009). Historically, American bison (*Bison bison*), pronghorn (*Antilocapra americana*), and black-tailed prairie dogs (Chapter 20), were dominant herbivores in prairie grasslands (Krueger, 1986) and likely would have contributed to the dynamics of swift fox habitat (Sovada and others, 2009; Gese and Thompson, 2014).

Although there are limited studies on the effects of wildfire on swift foxes, they likely respond positively to fire because fire can maintain or create the open, short vegetation structure preferred by swift foxes, and denning animals typically can avoid direct mortality from fire (Thompson and others, 2008; Meyer, 2009). Indeed, following a prescribed fire in the shortgrass steppe of southeastern Colorado, swift foxes either had a positive or neutral response to a burn; swift foxes whose territories overlapped the burn increased their use of burned areas after the fire for foraging and denning (Thompson and others, 2008). The response of swift foxes to fire may depend, in part, on how fire affects prey populations (Meyer, 2009). Prey species vary in their response to fire, and swift foxes are expected to exploit prey species that increase following fires (Meyer, 2009).

Associated Species of Management Concern

Conservation of shortgrass or mixed-grass prairie for swift foxes also may provide an umbrella of protection for other at-risk species that also inhabit these systems (Sovada and others, 2009). Some examples would include the black-tailed prairie dog (Chapter 20), western burrowing owl (*Athene cunicularia hypugaea*) (Chapter 12), mountain plover (*Charadrius montanus*) (Chapter 17), ferruginous hawk (*Buteo regalis*) (Chapter 13), Cassin's sparrow (*Aimophila cassinii*), lark bunting (*Calamospiza melanocorys*), loggerhead shrike (*Lanius ludovicianus*), McCown's longspur (*Calcarius mccownii*), and massasauga (*Sistrurus catenatus*).

Change Agents

Development

Swift foxes appear to be fairly tolerant of road presence in their habitats, as indicated by the greater proximity of dens to roads and the greater occurrence of radio-collared swift foxes near roads than would be expected based on habitat availability (Harrison, 2003; Kintigh and Andersen, 2005; Russell, 2006). It has been suggested that because coyotes tend to avoid roads, greater swift fox occurrence near roads may be a response to reduced predation risk along roads (Kamler, Ballard, Fish, and others, 2003). Mortality from vehicles, however, can be significant where roads have fragmented landscapes inhabited by swift foxes (Moehrenschlager and others, 2004). Consequently, roads may serve as habitat sinks by attracting foxes to areas where risk from vehicle mortality is high (Meyer, 2009). Indeed, mortality from being hit by moving vehicles can exceed mortality from coyote predation, particularly where fox dens are located near roads or in areas where coyote populations are actively controlled (Meyer, 2009). Swift foxes also may be attracted to carrion along roads, and prey availability may be greater along roadsides were vegetative cover is greater than it is in nearby grazed rangelands (Stephens and Anderson, 2005; Sovada and others, 2009).

Although swift foxes appear to tolerate roads, they are sensitive to disturbance from humans near their dens and avoid denning near human residences; they also may abandon dens that have been disturbed (Kintigh and Anderson, 2005). In addition, the proximity of urban or rural developments could lead to predation or the spread of diseases by domestic dogs (Aguirre, 2009). It has also been suggested that because red foxes (*Vulpes vulpes*) are more tolerant of human activity and developed areas, increasing red fox populations could lead to greater competition with swift foxes (Kamler, Ballard, Fish and others, 2003; Sovada and others, 2009).

Energy and Infrastructure

There has been little research to address effects of energy development on swift foxes (Moehrenschlager and others, 2004). Energy development, however, including oil and gas, wind, and biofuels, in the remaining native grasslands or rangelands used by the swift fox 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). An increase in roads, traffic, and human activity associated with energy development could lead to fox avoidance of disturbed sites (Moehrenschlager and others, 2004) or direct mortality (Moehrenschlager and Sovada, 2004). Moreover, golden eagles (*Aquila chrysaetos*) are important predators of the swift fox, and the elevated structures associated with energy development often are used by these large raptors to scan for prey; thus, powerlines and other tall infrastructure around energy developments could provide opportunities for increased eagle predation on swift foxes (Committee on the Status of Endangered Wildlife in Canada, 2009).

Agricultural Activities

The conversion of large expanses of native prairie to cropland has greatly fragmented and reduced swift fox habitat (Sovada and others, 2009; Schwalm and others, 2014). Historically, agricultural conversion was most prevalent in mixed-grass prairie, whereas the dryer shortgrass prairie has been used more for grazing and dry-land farming (Sovada and others, 2009). Both rangelands and dry-land farmlands may be inhabited by the swift fox; consequently, most of the remaining swift fox habitat occurs within the historic distribution of shortgrass prairie (Matlack and others, 2000; Kamler, Ballard, Fish, and others, 2003; Sovada and others, 2009). The remaining native grasslands are embedded in a landscape mosaic dominated by rangelands and cultivated areas (Jackson and Choate, 2000); however, the minimum patch size of grassland habitat suitable for supporting viable swift fox populations is unknown (Stephens and Anderson, 2005). Swift fox populations have not recovered from historical lows where fragmentation resulting from agricultural activities is prevalent, nor have they recovered along the peripheries of their range (Stephens and Anderson, 2005; Sovada and others, 2009).

Swift foxes appear to prefer native grasslands and rangelands over cultivated areas (Kamler, Ballard, Fish, and others, 2003). Although they often inhabit non-irrigated farmlands, the viability of swift fox populations in these cultivated lands remains equivocal (Stephens and Anderson, 2005). In some studies, swift fox density, body condition, and survival were greater in rangeland than they were in dry-land agricultural pasture (Matlack and others, 2000; Kamler, Ballard, Fish, and others, 2003); in other studies, however, prey availability and den site density did not differ between rangeland and dry-land agricultural pasture (Jackson and Choate, 2000; Sovada and others, 2001). Mortality by coyotes and vehicles can be greater in rangeland, whereas nontraumatic mortalities can be greater in cropland, which could result in reduced population viability in these areas (Matlack and others, 2000; Stephens and Anderson, 2005). Swift foxes may construct dens in fallow fields, but generally they avoid irrigated cropland and CRP land, which is often planted with mid-height or tallgrass species (Kahn and others, 1997; Jackson and Choate, 2000; Kamler, Ballard, Fish, and others, 2003; Sovada and others, 2009). In contrast, coyotes and red foxes readily use CRP lands, which could negatively affect swift foxes

(Kahn and others, 1997). Consequently, it has been suggested that planting native grass species on CRP lands would benefit swift foxes (Kahn and others, 1997; Stephens and Anderson, 2005).

Despite fragmentation of native grasslands, swift fox populations in the core of their range remain viable in the largely human-modified landscape, but there is continued concern in areas where fox populations remain low. A study of genetics among swift fox populations throughout much of their current distribution in the United States indicates that, despite severe population reduction and fragmentation of remaining habitat, the genetic viability of swift fox populations is not currently a conservation concern (Schwalm and others, 2014). Natural barriers appear to have a greater restrictive effect on gene flow than potential barriers posed by broad-scale conversion to agricultural lands (Schwalm and others, 2014). Although populations are starting to recover in the periphery of their range, the residual effects of introduced populations of swift foxes on genetic diversity in Montana and Canada are still evident, and connectivity with other swift fox populations may be necessary to avoid the negative effects of small populations (Cullingham and Moehrenschlager, 2013).

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). Grazing practices, however, differ from the intense but spatially and temporally heterogeneous grazing by migratory herds of bison and colonies of prairie dogs (Cid and others, 1991), and the degree to which livestock grazing helps to create or maintain swift fox habitat is unclear (Sovada and others, 2009). 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 replicated by current grazing practices.

Altered Fire Regime

Fire suppression in native grasslands likely has had largely negative effects on the swift fox because fire helps to maintain the short, open vegetation structure, including minimal shrub cover, preferred by the species (Meyer, 2009). The use of 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; Gese and Thompson, 2014).

Invasive Species

It appears that presently no nonnative invasive species pose threats to the swift fox. Increasing populations of coyotes and red fox, however, may pose significant threats to swift fox populations. Historically, it is generally assumed that gray wolves largely coexisted with swift foxes in the Great Plains, and the wolves helped to reduce coyote populations. Following the eradication of gray wolves, however, coyote populations increased. Both coyotes and red foxes are habitat generalists and tend to tolerate human activities. Coyotes are the primary cause of swift fox mortality, and they may depress swift fox populations, as swift fox numbers can increase following coyote removal or control (Kilgore, 1969; Stephens and Anderson, 2005). Red foxes also may compete with swift foxes for habitat and prey (Dark-Smiley and Keinath, 2003; Stephens and Anderson, 2005; Sovada and others, 2009).

Introduced Insects and Disease

Disease does not appear to be a major threat to the swift fox (Moehrenschlager and Sovada, 2004). The overlap of the swift fox's range with that of domestic and wild canids, however, increases the potential for disease transfer (Pybus and Williams, 2003). Domestic dogs in particular are known reservoirs of infectious diseases that can affect wild canids (Aguirre, 2009). When seroprevalence (the number of individuals testing positive for a given disease) of certain canid diseases is high, the small size and fragmented condition of swift fox populations puts them at greater risk of disease outbreaks, as indicated by previous disease outbreaks that have decimated other endangered populations of canids (Committee on the Status of Endangered Wildlife in Canada, 2009).

Climate Change

Climate projections indicate the potential for shifts in precipitation patterns across the Great Plains, including an overall increase in the frequency and severity of drought (Karl and others, 2009). Although studies to assess potential effects of climate change on swift foxes are needed, changes in precipitation and temperature patterns are likely to affect primary productivity and plant species distribution and community composition, which in turn could affect the availability of swift fox prey species.

Rapid Ecoregional Assessment Components

A conceptual model for the key ecological attributes and CAs affecting the swift fox is illustrated in figure 41. Ecological attributes and CAs identified in the REA are enumerated in tables 45 and 46.

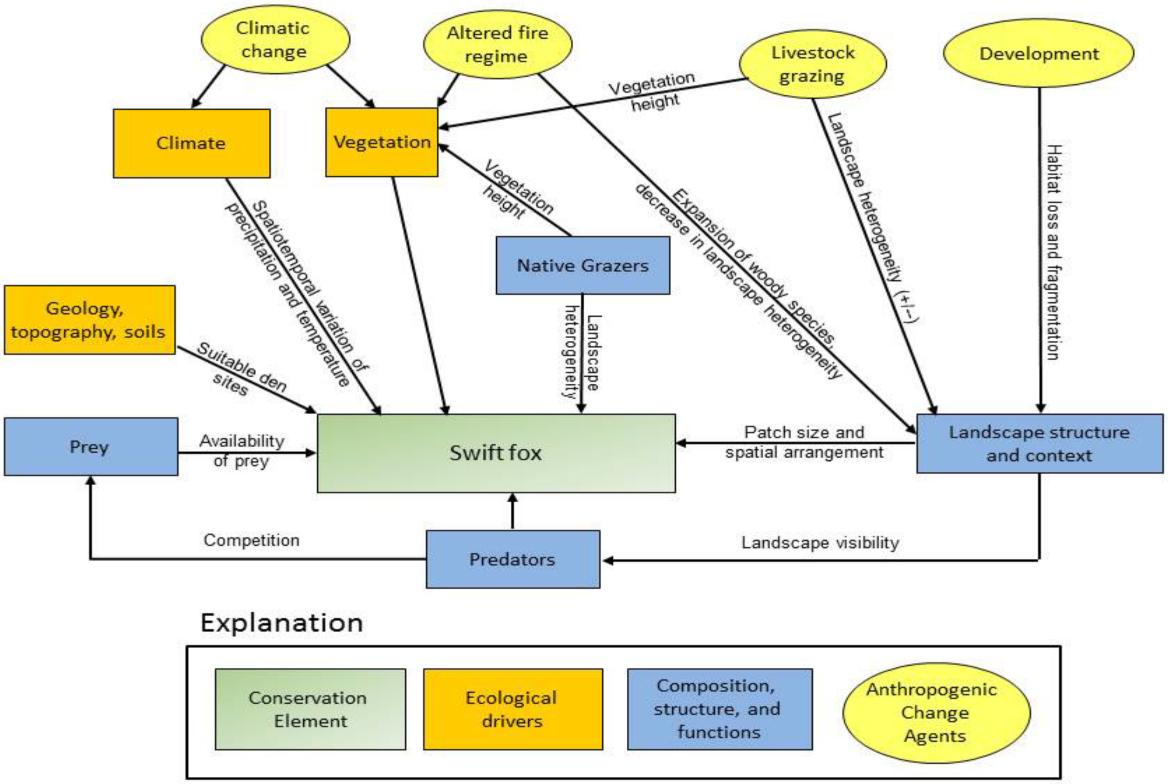

Figure 41. Generalized conceptual model highlighting the major key ecological attributes and Change Agents for the swift fox (*Vulpes velox*) in the Southern Great Plains ecoregion. Key ecological attributes and ecological processes regulating the occurrence, structure, and dynamics of swift fox populations and habtiats are shown in orange rectangles (see also table 45); additional ecological attributes are shown in blue rectangles; and anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals (see also table 46).

Table 45. Key ecological attributes identified by the Southern Great Plains Rapid Ecoregional Assessment for the swift fox (*Vulpes velox*).

Attribute	Variables
Amount and distribution	Swift fox distribution, den sites.
Landscape structure	Size and spatial distribution of habitat (patch size, connectivity).
Landscape dynamics	Habitat productivity (food resources), drought, shrubland-grassland ecotone dynamics, predator dynamics.
Associated species management of concern	Black-tailed prairie dog, ferruginous hawk, burrowing owl, mountain plover, Cassin's sparrow, lark bunting, loggerhead shrike, McCown's longspur, and massasauga.

Table 46. Anthropogenic Change Agents identified by the Southern Great Plains Rapid Ecoregional Assessment for the swift fox (*Vulpes velox*).

Attribute	Variables
Development (energy and infrastructure)	Habitat alteration (direct loss of habitat), habitat fragmentation, reduction in habitat connectivity, reduction in prey availability, increased risk of predation, increased in road-induced mortality.
Development (agricultural activities)	Habitat alteration (direct loss of habitat due to cultivation), habitat fragmentation and isolation (reduction in habitat connectivity), livestock grazing (beneficial in maintaining habitat quality, detrimental when grazing pressure is evenly distributed).
Altered fire regime	Fire suppression results in habitat alteration (expansion of woody species, decrease in landscape heterogeneity).
Invasive species	None, but range expansion of red fox could increase competition for resources.
Insects and disease	None, but potential of disease transfer between swift fox and other sympatric species; presence and effect of plague on black-tailed prairie dog population.
Climate change	Shift in suitable habitat, changes in prey availability, changes in interspecific competitors, habitat reduction.

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Appendix A. Documentation of Selection Process: Candidate Conservation Elements

Table A–1. Candidate Conservation Element species and species assemblages that did not meet the criteria for Phase II (sorted by taxa, then common name).

[Only selection criteria I and II are included in this list. NM, New Mexico; TX, Texas; CO, Colorado; OK, Oklahoma; KS, Kansas. Management priority: H, high; M, medium; L, low]

		Selection criteria					
		I					II
Taxa	**Species and species assemblages**	**NM**	**TX**	**CO**	**OK**	**KS**	**Priority score**
Amphibian/reptiles	Alligator snapping turtle	-	-	-	-	-	0
Amphibian/reptiles	Arid land (western) ribbon snake	L	-	-	-	-	1
Amphibian/reptiles	Checkered garter snake	-	-	-	-	L	1
Amphibian/reptiles	Chorus frog	L	-	-	-	-	1
Amphibian/reptiles	Common kingsnake	M	-	L	-	-	3
Amphibian/reptiles	Dunes sagebrush lizard	H	M	-	L	-	6
Amphibian/reptiles	Green toad	-	-	-	-	L	1
Amphibian/reptiles	Long-nosed snake	-	-	-	-	L	1
Amphibian/reptiles	Long-nosed leopard lizard	L	L	-	-	-	2
Amphibian/reptiles	Massasauga	M	M	-	-	-	4
Amphibian/reptiles	Milksnake	M	-	L	-	-	3
Amphibian/reptiles	Northern leopard frog	H	-	M	-	-	5
Amphibian/reptiles	Ornate box turtle	L	L	-	-	-	2
Amphibian/reptiles	Plain-bellied water snake	H	-	-	-	-	3
Amphibian/reptiles	Plains leopard frog	M	-	M	-	-	4
Amphibian/reptiles	Strecker’s chorus frog	-	-	-	-	L	1
Amphibian/reptiles	Texas garter snake	-	L	-	-	-	1
Amphibian/reptiles	Texas horned lizard	M	M	-	-	-	4
Amphibian/reptiles	Western hog-nosed snake	L	L	-	-	-	2
Amphibian/reptiles	Rio Grande (or western) river cooter	H	-	-	-	-	3
Birds	American white pelican	L	-	M	-	-	3
Birds	Bald eagle	L	-	M	-	-	3
Birds	Bell’s vireo	M	-	-	L	-	3
Birds	Black-capped vireo	-	-	-	L	-	1
Birds	Cassin’s sparrow	M	L	-	L	-	4
Birds	Common black-hawk	H	-	-	-	-	3
Birds	Eastern meadowlark	L	L	-	-	-	2
Birds	Grasshopper sparrow	L	-	-	L	-	2
Birds	Greater prairie-chicken	-	-	-	-	H	3
Birds	Harris’ sparrow	L	-	-	L	-	2
Birds	Lark bunting	L	-	-	L	-	2
Birds	Least sandpiper	L	-	-	L	-	2
Birds	Long-billed dowitcher	L	-	-	L	-	2
Birds	Northern bobwhite	-	H	-	-	H	6

Table A–1. Candidate Conservation Element species and species assemblages that did not meet the criteria for Phase II (sorted by taxa, then common name).—Continued

[Only selection criteria I and II are included in this list. NM, New Mexico; TX, Texas; CO, Colorado; OK, Oklahoma; KS, Kansas. Management priority: H, high; M, medium; L, low]

		Selection criteria					
		I					II
Taxa	Species and species assemblages	NM	TX	CO	OK	KS	Priority score
Birds	Northern pintail	M	M	-	L	L	0
Birds	Orchard oriole	-	L	-	-	-	1
Birds	Piping plover	L	-	L	L	L	4
Birds	Prairie falcon	M	-	-	L	-	3
Birds	Red-headed woodpecker	L	L	-	-	-	2
Birds	Sandhill crane	H	L	-	L	L	6
Birds	Scaled quail	M	M	L	-	L	0
Birds	Western meadowlark	L	L	-	-	-	2
Birds	Western sandpiper	-	-	-	L	-	1
Birds	White-faced ibis	L	-	L	-	-	2
Birds	Whooping crane	-	L	-	M	M	5
Birds	Wilson’s phalarope	L	-	-	L	-	2
Birds	Yellow-billed cuckoo	H	-	-	-	-	3
Fish	Alligator gar	-	-	-	-	-	0
Fish	Arkansas darter	-	-	M	L	M	5
Fish	Arkansas River speckled chub	-	-	-	-	-	0
Fish	Chub shiner	-	-	-	-	-	0
Fish	Flathead chub	-	-	-	-	L	1
Fish	Paddlefish	-	-	-	L	L	2
Fish	Pallid sturgeon	-	-	-	L	L	2
Fish	Pecos pupfish	H	-	-	-	-	3
Fish	Peppered chub	-	-	-	-	H	3
Fish	Plains minnow	-	-	-	-	M	2
Fish	Prairie chub	-	-	-	-	-	0
Fish	Red River pupfish	-	-	-	-	-	0
Fish	Red River shiner	-	-	-	-	-	0
Fish	Sharpnose shiner	-	-	-	-	-	0
Fish	Shovelnose sturgeon	-	-	-	L	L	2
Fish	Silver chub	-	-	-	-	L	1
Fish	Smalleye shiner	-	-	-	-	-	0
Fish	Topeka shiner	-	-	-	L	L	2
Invertebrates	American burying beetle	-	-	-	-	M	2
Invertebrates	Aquatic insect assemblage	-	M	L	-	M	5
Invertebrates	Bleufer (mussel)	-	-	-	-	-	0
Invertebrates	Creeper	-	-	-	-	M	2
Invertebrates	Cylindrical papershell	-	-	-	-	H	3
Invertebrates	Darkling beetle	-	L	-	-	-	2
Invertebrates	Fatmucket	-	-	-	-	L	1
Invertebrates	Freshwater mussel assemblage	H	-	-	-	H	6
Invertebrates	Hickorynut	-	-	-	-	L	1
Invertebrates	Large branchiopods (fairy and clam shrimp)	H	-	-	-	-	3

Table A–1. Candidate Conservation Element species and species assemblages that did not meet the criteria for Phase II (sorted by taxa, then common name).—Continued

[Only selection criteria I and II are included in this list. NM, New Mexico; TX, Texas; CO, Colorado; OK, Oklahoma; KS, Kansas. Management priority: H, high; M, medium; L, low]

		Selection criteria					
		I					II
Taxa	**Species and species assemblages**	**NM**	**TX**	**CO**	**OK**	**KS**	**Priority score**
Invertebrates	Pimpleback	-	-	-	-	L	1
Invertebrates	Pink heelsplitter	-	-	-	-	L	1
Invertebrates	Pistolgrip	-	-	-	-	L	1
Invertebrates	Plain pocketbook	-	-	-	-	L	1
Invertebrates	Salt playa fairy shrimp	H	L	-	-	-	4
Invertebrates	Scarab beetle	-	L	-	-	-	1
Invertebrates	Smooth pimpleback	-	H	-	-	-	3
Invertebrates	Stag beetle	-	L	-	-	-	1
Invertebrates	Texas fatmucket	-	H	-	-	-	3
Invertebrates	Texas fawnsfoot	-	H	-	-	-	3
Invertebrates	Texas pimpleback	-	H	-	-	-	3
Invertebrates	Threeridge	-	-	-	-	M	2
Invertebrates	Wabash pigtoe	-	-	-	-	M	2
Invertebrates	Weevil	-	L	-	-	-	1
Mammals	American beaver	M	-	-	-	-	2
Mammals	American bison	L	-	-	L	-	2
Mammals	Bat assemblage	M	L	-	-	H	6
Mammals	Big free-tailed bat	M	L	L	-	-	4
Mammals	Black-footed ferret	H	M	-	L	-	6
Mammals	Cave myotis	-	-	-	-	-	0
Mammals	Desert shrew	L	L	-	-	-	2
Mammals	Eastern spotted skunk	-	-	-	-	L	1
Mammals	Hoary bat	M	-	-	-	-	2
Mammals	Hog-nosed skunk	L	L	-	-	-	2
Mammals	Least shrew	M	-	-	-	-	2
Mammals	Mountain lion	M	L	-	-	-	3
Mammals	New Mexican meadow jumping mouse	H	-	L	-	-	0
Mammals	Palo duro mouse	-	M	-	-	-	2
Mammals	Preble's meadow jumping mouse	-	-	M	-	-	2
Mammals	Pronghorn	H	M	-	-	L	6
Mammals	Silver-haired bat	M	-	-	-	-	2
Mammals	Texas kangaroo rat	-	M	-	-	-	2
Mammals	Tri-colored bat (formerly known as eastern pipistrelle)	M	-	-	-	-	2
Mammals	Western big-eared bat	-	-	-	-	M	2
Mammals	Wild turkey	-	M	-	-	-	2
Mammals	Yellow-faced pocket gopher	L	L	-	-	-	2
Plants	Blowout penstemon	-	-	-	-	-	0
Plants	Colorado butterfly plant	-	-	L	-	-	1
Plants	Ute lady’s tresses	-	-	L	-	-	1

Appendix B. Preliminary Management Question Matrix

Table B–1. Candidate management questions and issues, organized by Conservation Element (CE) and Change Agent (CA).

[Questions/issues appear as written by team members who submitted them. General questions related to key ecological attributes (KEAs), CAs, and integrated Management Questions (MQs) are at the bottom of the table. Column S refers to status (1 = addressed in Phase I CE narrative; 2 = addressed in Phase II (dependent on data availability); 3 = unknown whether the question can be addressed OR the question/issue can be partially addressed; and 4 = out of scope/cannot be addressed). Questions/issues in boldface font have been identified as priority issues by an Assessment Management Team member. Note: This information will be updated and the status of each issue is subject to change. BTPD, black-tailed prairie dog.]

MQ no.	Conservation Element	Group	Question/issue	S
1	Arkansas River shiner	Development	**Reduced water flow and altered water quality; impacted by dewatering and impoundments**	3
2	Arkansas River shiner	Development	Hydrologic alteration (alteration of seasonal flood pulse; for example, spawning cues and groundwater withdrawals)	3
3	Arkansas River shiner	Development	Poor land management practices	4
4	Bat species assemblage	General	Lack of conservation information based on population trends, roosting and habitat requirements and locations	3
5	Bat species assemblage	Development	**How has energy development and associated infrastructure (powerline and wind generation) impacted bat habitat and mortality?**	3
6	Bat species assemblage	Development	Where is cave recreation (spelunking) likely to impact bat habitat?	2
7	Bat species assemblage	Development	Cave closure and inappropriate gating	3
8	Bat species assemblage	Development	Eradication from dwellings or other large roost sites with human proximity	3
9	Bat species assemblage	Development	Disturbance of roosts and hibernacula, damaged or destroyed by vandalism, mine closures and reclamation, recreational activities such as rock climbing, and, where man-made structures are occupied, demolition, modification, chemical treatments, or intentional eradication and exclusion	3
10	Bat species assemblage	Development	**Impacts of changing agricultural development**	3
11	Bat species assemblage	Altered fire regime	Loss or modification of foraging habitat due to wildfire, agricultural expansion, and (or) pesticide use	3
12	Bat species assemblage	Insects and disease	**What is the extent of white-nose syndrome occurrence in bat populations?**	1
13	Black-tailed prairie dog	General	What areas are protected by management decisions?	3
14	Black-tailed prairie dog	General	Where are the locations of known prairie dog colonies?	2
15	Black-tailed prairie dog	Development	Where does urban development impact BTPD habitat?	2
16	Black-tailed prairie dog	Development	**Where does energy development (renewable and non-renewable) contribute to fragmentation and loss of BTPD habitat?**	2
17	Black-tailed prairie dog	Development	**Loss of open prairie landscapes; genetic isolation**	3
18	Black-tailed prairie dog	Development	Direct killing of species (poisoning and control of species)	4
19	Black-tailed prairie dog	Invasive species	Shrub encroachment	3

Table B–1. Candidate management questions and issues, organized by Conservation Element (CE) and Change Agent (CA).—Continued

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MQ no.	Conservation Element	Group	Question/issue	S
20	Black-tailed prairie dog	Insects and disease	**Where does sylvatic plague impact BTPD species?**	3
21	Black-tailed prairie dog	Climate change	Where is BTPD habitat most susceptible to long-term drought and climate change?	2
22	Burrowing owl	General	Edge mortality	4
23	Burrowing owl	General	Predation (foraging and nesting areas are less widespread)	3
24	Burrowing owl	General	Where are known BTPD colonies?	2
25	Burrowing owl	Development	BTPD burrowing infill for agriculture	3
26	Burrowing owl	Development	Oil and gas development activity disturbs nesting habitat	2
27	Burrowing owl	Development	BTPD shooting and poisoning (laws still allow it)	3
28	Burrowing owl	Development	**Development (energy, urban, agricultural) results in loss or fragmentation of habitat**	2
29	Burrowing owl	Development	Direct killing; illegal shooting	4
30	Burrowing owl	Development	Improper grazing practices	4
31	Burrowing owl	Development	Pesticides and other contaminants which may affect arthropod diversity, availability and which may accumulate in owls	4
32	Ferruginous hawk	General	Where are known black-tailed prairie dog colonies?	2
33	Ferruginous hawk	Development	**Oil, gas, and wind development disturb ground nest sites**	3
34	Ferruginous hawk	Development	Decreased prey populations due to due to human activities (BTPD shooting and poisoning (laws still allow it)	4
35	Ferruginous hawk	Development	**Loss or fragmentation of habitat due to development (energy, urban, agricultural conversion, roads)**	2
36	Ferruginous hawk	Development	Human disturbance at nest sites	3
37	Ferruginous hawk	Development	Illegal shooting	4
38	Ferruginous hawk	Development	**Does extensive use of chemical control for prairie dogs impact ferruginous hawks and other raptor species?**	4
39	Ferruginous hawk	Altered fire regime	Changes in fire frequency and intensity which could shift vegetation communities and later food sources	3
40	Ferruginous hawk	Altered fire regime	Compromised degraded grasslands invaded by woody shrublands - shrubland cover higher than natural matrix	3
41	Ferruginous hawk	Invasive species	Ability of native grasslands and shrublands to support viable populations may be compromised by the invasion of exotic annuals	4
42	Ferruginous hawk	Invasive species	Shrub encroachment	3
43	Ferruginous hawk	Climate change	Climate change-driven shifts attributed to less precipitation, warmer temperatures, plant community composition changes (increasing tree cover in shrublands, shrub invasion in grasslands, loss of succulents, nonnative grass and woody species invasion exacerbated)	3

Table B–1. Candidate management questions and issues, organized by Conservation Element (CE) and Change Agent (CA).—Continued

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MQ no.	Conservation Element	Group	Question/issue	S
44	Freshwater mussel species assemblage	Development	**How does water withdrawal impact freshwater mussel species?**	1
45	Freshwater mussel species assemblage	Development	Hydrologic alterations	1
46	Freshwater mussel species assemblage	Development	**Habitat alterations (sedentary organisms do not respond fast enough to habitat changes; that is, land use changes, poor riparian management)**	3
47	Freshwater mussel species assemblage	Development	Water quality impairments	1
48	Freshwater mussel species assemblage	Development	**Fish community changes (fish host required to complete life-cycle)**	1
49	Freshwater mussel species assemblage	Climate change	How does drought impact freshwater mussel species?	1
50	Interior least tern	General	Inundation and fluctuating water levels of nesting habitat	1
51	Interior least tern	General	Jurisdictional issues due to water level changes around reservoirs	3
52	Interior least tern	General	Where are known occupied and potential habitats located?	2
53	Interior least tern	Development	Chemical contamination of prey base	4
54	Interior least tern	Development	Habitat loss due to altered flow regime and channelization	3
55	Interior least tern	Development	Nest disturbance - recreational activities on rivers and sandbars disturb the nesting terns, causing them to abandon their nests;	3
56	Interior least tern	Development	Hydroelectric power and water storage development: dams, reservoirs, and the resulting release management that changes natural braided channel river systems and sand/gravel bar networks have been eliminated in most areas, replaced by channelized, narrow or incised waterways;	4
57	Interior least tern	Development	Degraded riverine habitats that no longer support forage fishes	3
58	Interior least tern	Climate change	Water releases due to drought and human water needs continues to be a problem for this species as well as other river dependent species (Brazos water snake).	3
59	Interior least tern	Climate change	Where is Interior least tern habitat most susceptible to long-term drought and climate change?	2
60	Lakes and reservoirs	Development	Water withdrawal	3
61	Lakes and reservoirs	Development	Aging lakes and reservoirs (siltation, loss of habitat heterogeneity)	1
62	Lakes and reservoirs	Development	Land use changes	3
63	Lakes and reservoirs	Development	Poor land management practices	4
64	Lakes and reservoirs	Invasive species	Quagga mussels	1

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MQ no.	Conservation Element	Group	Question/issue	S
65	Lakes and reservoirs	Insects and disease	Toxic algae blooms	1
66	Lakes and reservoirs	Climate change	Long-term drought	1
67	Lesser prairie-chicken	General	Are the conservation measures identified for the lesser prairie-chicken unique to that species, or should they have some broader application?	3
68	Lesser prairie-chicken	General	**Does the Crucial Habitat Assessment Tool identify habitat values/indices for species other than the lesser prairie-chicken, and if so, how are those habitat indices being used?**	3
69	Lesser prairie-chicken	General	**What areas exhibit ideal lesser prairie-chicken habitat in terms of vegetation composition (nesting and brooding habitats)?**	3
70	Lesser prairie-chicken	General	What areas are under management decisions to improve conditions for the continued existence of habitat and viable lesser prairie-chicken populations? What degraded areas can be incorporated into management schemes to improve conditions?	3
71	Lesser prairie-chicken	General	Where are large tracts of relatively undisturbed occupied and potential habitat?	2
72	Lesser prairie-chicken	General	**Where does shinnery-oak control efforts overlap lesser prairie-chicken habitat?**	3
73	Lesser prairie-chicken	Development	**Excessive grazing pressure suppresses native grasses, diminishing habitat quality, lessening fire frequency, enhancing competitive advantage to invasive trees. Diminished habitat quality exposes lesser prairie-chickens to increased predation, including a new suite of predators that take advantage of invasive trees (for example, raptors).**	3
74	Lesser prairie-chicken	Development	**Where does energy development (renewable and non-renewable) contribute to fragmentation across lesser prairie-chicken's range?**	2
75	Lesser prairie-chicken	Development	Range contraction and fragmentation contributes to genetic isolation and lowered reproductive capacity`	3
76	Lesser prairie-chicken	Development	Where is lesser prairie-chicken habitat fragmented due to roads and powerlines?	2
77	Lesser prairie-chicken	Development	Where do inappropriate grazing practices impact lesser prairie-chicken habitat?	4
78	Lesser prairie-chicken	Development	**Where do intensive agricultural practices contribute to fragmentation across the lesser prairie-chicken's range?**	2
79	Lesser prairie-chicken	Altered fire regime	**Changes in fire frequency and intensity which could shift vegetation communities and later food sources**	1
80	Lesser prairie-chicken	Altered fire regime	**Degradation of remaining prairie habitats by improper grassland management: long-term fire suppression leads to invasive trees - eastern red cedar (*Juniperus virginiana*), mesquite (*Prosopis glandulosa*), excessive shinnery oak (*Quercus havardii*)—making habitat unsuitable**	3
81	Lesser prairie-chicken	Invasive species	How do we manage eastern red cedar in an area that is 95 percent privately owned?	3

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MQ no.	Conservation Element	Group	Question/issue	S
82	Lesser prairie-chicken	Climate change	**Climate change impacts: plant community composition changes (increasing tree cover in shrublands, shrub invasion in grasslands, loss of succulents, nonnative grass and woody species invasion exacerbated**	1
83	Lesser prairie-chicken	Climate change	**Where is lesser prairie-chicken habitat most susceptible to long-term drought and climate change?**	2
84	Lesser prairie-chicken	Climate change	**Climate change may increase frequency, intensity, and duration of droughts on the High Plains**	1
85	Lesser prairie-chicken	Climate change	**Enhanced vulnerability to catastrophic events, particularly drought (water availability appears to be a key factor in nest and brood survival)**	1
86	Long-billed curlew	Development	Loss or fragmentation of habitat due to development (energy, urban, agricultural conversion)	2
87	Long-billed curlew	Development	Pesticide/herbicide use	4
88	Long-billed curlew	Development	Improper grazing practices which degrade grasslands and (or) convert to nonnative vegetation	4
89	Long-billed curlew	Development	Pesticide use in agricultural fields diminishing forage base	4
90	Long-billed curlew	Altered fire regime	Changes in the natural fire regime	4
91	Long-billed curlew	Invasive species	Shrub encroachment	3
92	Long-billed curlew	Insects and disease	May be impacted by disease (West Nile virus)	4
93	Long-billed curlew	Climate change	Where is long-billed curlew habitat most susceptible to long-term drought and climate change?	2
94	Long-billed curlew	Climate change	Climate change: relative sea level rise expected to inundate or fragment low-lying habitats such as salt marshes, sandy beaches, barrier islands, and mudflats, and increasing frequency and severity of storms and changes in water temperatures will impact quality and quantity of coastal habitats and alter food resources.	3
95	Mixed-grass prairie	General	Where are areas that can serve as ideal ecological condition sites, or have potential to improve to that level or respond to treatments?	3
96	Mixed-grass prairie	General	**Where are areas of occupied and suitable dunes sagebrush lizard habitat being protected by current management decisions?**	3
97	Mixed-grass prairie	General	**Where are currently impacted dunes sagebrush lizard habitat areas with potential for recovery located?**	3
98	Mixed-grass prairie	General	**Where are large tracts of shinnery oak habitat not in vegetation composition balance located?**	4
99	Mixed-grass prairie	General	**Where are monotypic stands of shinnery oak (where composition of grasses and forbs are out of balance)?**	4
100	Mixed-grass prairie	Development	**Where are shinnery oak stands being impacted by herbicides?**	3
101	Mixed-grass prairie	Development	**Where is dunes sagebrush lizard habitat being lost and fragmented from oil and gas development?**	2
102	Mixed-grass prairie	Development	Where is dune sagebrush lizard habitat being impacted by herbicides?	4

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MQ no.	Conservation Element	Group	Question/issue	S
103	Mixed-grass prairie	Development	**Where are shinnery oak stands being lost or fragmented due to oil and gas development?**	2
104	Mixed-grass prairie	Development	**Where are shinnery oak stands being impacted by inappropriate grazing practices?**	4
105	Mixed-grass prairie	Development	**Where are shinnery oak stands being lost or fragmented due agricultural development (breaking out rangeland with sufficient underground water capable of supporting peanut production)?**	2
106	Mixed-grass prairie	Development	**Where is sand sagebrush being lost or fragmented due to oil, gas and wind development?**	2
107	Mixed-grass prairie	Development	**Where is sand sagebrush being lost to agricultural development?**	2
108	Mixed-grass prairie	Development	**Where are sand sagebrush stands being impacted by inappropriate grazing practices?**	4
109	Mixed-grass prairie	Development	Residual toxic contamination of shinnery/grassland dunes matrix habitat	4
110	Mixed-grass prairie	Development	**Energy development and production**	2
111	Mixed-grass prairie	Development	Habitat degradation/alteration largely due to poor management (mismanagement of grazing)	4
112	Mixed-grass prairie	Altered fire regime	**Habitat change resulting from changes fire frequency**	3
113	Mixed-grass prairie	Altered fire regime	**Habitat degradation/alteration largely due to poor management (lack of fire)**	3
114	Mixed-grass prairie	Altered fire regime	**Most mixed-grass prairies have experienced shrub/woody species encroachment and are now dominated by shrubs (shinnery oak/sand sagebrush) instead of grasses and forbs**	3
115	Mixed-grass prairie	Invasive species	**Where are invasive spp. Impacting shinnery oak?**	3
116	Mixed-grass prairie	Invasive species	Habitat degradation/alteration due to invasion of exotic species	3
117	Mixed-grass prairie	Climate change	Where is sand sagebrush most susceptible to long-term drought and climate change?	2
118	Mountain plover	General	Where are known black-tailed prairie dog colonies?	2
119	Mountain plover	Development	**Development (energy, urban, agricultural) results in loss or fragmentation of habitat**	2
120	Mountain plover	Development	Loss of native grazers including BTPDs	3
121	Mountain plover	Development	Loss and fragmentation of migratory and winter habitat	2
122	Mountain plover	Development	Nests and nestlings vulnerable to crop management and harvest conditions	2
123	Mountain plover	Climate change	Where is Mountain Plover habitat most susceptible to long-term drought and climate change?	2
124	Mountain plover	Climate change	Climate change may shift change grassland suitability in breeding areas.	3
125	Mule deer	Development	**Habitat loss and fragmentation due to energy development (won't utilize habitat that is frequently disturbed by energy producing activities)**	2

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MQ no.	Conservation Element	Group	Question/issue	S
126	Mule deer	Development	**How has energy development in the Permian Basin impacted mule deer habitat?**	2
127	Mule deer	Altered fire regime	Ecological succession and altered fire regime	3
128	Mule deer	Insects and disease	Where are mule deer susceptible to chronic wasting disease?	3
129	Mule deer	Climate change	Climate change and long-term drought effects to habitat and food base	3
130	Playas and saline lakes	Development	**How are the playas and saline lakes being affected by demand for water from agriculture/ranching, energy development, residential consumers and other uses?**	3
131	Playas and saline lakes	Development	**Where have playas been lost due to development?**	2
132	Playas and saline lakes	Development	**Where have playas been fragmented due to development?**	2
133	Playas and saline lakes	Development	**Where have playas been converted to agricultural lands?**	3
134	Playas and saline lakes	Invasive species	Where are invasive species impacting playas?	3
135	Playas and saline lakes	Climate change	Where are playas most susceptible to climate change and (or) long-term drought?	2
136	Prairie rivers and streams	Development	**Hydrologic alterations—anthropogenic alterations**	3
137	Prairie rivers and streams	Development	Poor land management practices (destruction of riparian zones)	4
138	Prairie rivers and streams	Development	Land use changes and habitat fragmentation	3
139	Prairie rivers and streams	Climate change	Hydrologic alterations—climate change	2
140	Riparian and wetlands	General	What degraded areas can be incorporated into management schemes to improve conditions?	3
141	Riparian and wetlands	General	Where are intact proper functioning riparian wetland areas found?	3
142	Riparian and wetlands	Development	**Habitat loss and fragmentation**	2
143	Riparian and wetlands	Development	Water withdrawal/diversions	3
144	Riparian and wetlands	Development	Modified disturbance regime (flooding) in riparian habitats; hydrologic alterations (disconnection of river and floodplain)	3
145	Riparian and wetlands	Development	Major flood events magnified by upland watershed conditions or wildfire	4
146	Riparian and wetlands	Development	Poor land management practices; incompatible grazing practices	4
147	Riparian and wetlands	Altered fire regime	Unnatural fire in riparian woodlands and forests	3
148	Riparian and wetlands	Invasive species	What areas are being invaded by non-native phreatophytes and noxious weeds?	3

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MQ no.	Conservation Element	Group	Question/issue	S
149	Riparian and wetlands	Invasive species	What type of exotic/invasive spp. Are impacting wetlands?	3
150	Riparian and wetlands	Climate change	Long-term drought	2
151	Shortgrass prairie	General	What is the rangeland condition across the shortgrass prairie?	4
152	Shortgrass prairie	General	Where are locations of conservation concern in the shortgrass prairie ecosystem?	2
153	Shortgrass prairie	Development	**Where has Shortgrass prairie been lost due to energy development?**	2
154	Shortgrass prairie	Development	How have inappropriate grazing practices contributed to degradation and (or) alteration of the shortgrass prairie?	4
155	Shortgrass prairie	Development	**Where has shortgrass prairie been lost due to conversion to crop land?**	2
156	Shortgrass prairie	Development	Where have past grazing management practices shifted ecological condition and vegetation composition from historical range of variation in the shortgrass prairie?	4
157	Shortgrass prairie	Development	Where has fencing impacted shortgrass prairie (that is, where are areas of high/low density of fencelines in the shortgrass prairie)?	3
158	Shortgrass prairie	Development	**What grazing allotments can be targeted for large scale fence modifications in shortgrass pronghorn antelope habitat?**	3
159	Shortgrass prairie	Development	**Where are high value conservation reserve program lands?**	3
160	Shortgrass prairie	Development	**Where are areas of shortgrass prairie most likely to be impacted by transmission corridor expansion?**	3
161	Shortgrass prairie	Altered fire regime	**Where can prescribed fire be re-introduced and shortgrass prairie maintained by prescribed fire?**	4
162	Shortgrass prairie	Altered fire regime	How/where has the lack of fire contributed to expansion of woody species in shortgrass prairie?	3
163	Shortgrass prairie	Altered fire regime	Where have invasive species, fire and range condition acted as a cumulative Change Agent and led to type conversions?	3
164	Shortgrass prairie	Invasive species	Where have shrub/woody species such as mesquite and cholla expanded and displaced grasses and forbs?	3
165	Shortgrass prairie	Invasive species	**Where are intact patches of shortgrass prairie going to be most threatened by invasive species in the future?**	2
166	Shortgrass prairie	Invasive species	Where have exotic and (or) invasive species contributed to degradation of shortgrass prairie?	3
167	Shortgrass prairie	Invasive species	Where are areas of shortgrass prairie heavily impacted by invasive species and are these areas performing the same ecological function as areas with largely native vegetation composition?	3
168	Shortgrass prairie	Invasive species	Is eastern redcedar expansion a concern in the shortgrass prairie?	3
169	Shortgrass prairie	Invasive species	Is one-seeded juniper expansion occurring in the shortgrass prairie?	3
170	Shortgrass prairie	Climate change	What is the current distribution and projected change in shortgrass prairie distribution due to climate change and other land use practices, including restoration practices, and changes in livestock distribution?	3

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MQ no.	Conservation Element	Group	Question/issue	S
171	Shortgrass prairie	Climate change	What are the potential changes in the community composition of shortgrass prairie as a result of climate change?	3
172	Shortgrass prairie	Climate change	Considering potential changes to vegetation communities as a result of climatic change, where are the areas with the highest restoration potential (that is, intact shortgrass prairie that will transition to shrub invaded versus shrub-invaded shortgrass prairie that will transition to shrub dominance)?	3
173	Shortgrass prairie	Climate change	Where is shortgrass prairie expected to expand or contract as a result of climate change?	2
174	Shortgrass prairie	Climate change	Where is shortgrass prairie at risk from long-term drought impacts?	2
175	Shortgrass prairie	Climate change	How will climate change impact phenology of the shortgrass prairie?	3
176	Snowy plover	Development	Human disturbance on shoreline (vehicles, recreation, unleashed dogs), conflict with human recreational uses, disturbance to nesting birds	3
177	Snowy plover	Development	Habitat loss due to dams and channelization	3
178	Snowy plover	Development	Shoreline loss and erosion from increased shipping channel wave action	4
179	Snowy plover	Development	Changes in salinity and water quality caused by lack of instream flows as well as ocean acidification may affect forage (small crustaceans, mollusks, marine worms, aquatic insects, and seeds)	4
180	Snowy plover	Development	**Playa lake loss and degradation from siltation and (or) conversion to agriculture**	3
181	Snowy plover	Development	**Wind generation**	2
182	Snowy plover	Altered fire regime	Loss, fragmentation, or alteration of breeding alkali flats and playas from vegetation encroachment	3
183	Snowy plover	Climate change	Loss, fragmentation, or alteration of breeding alkali flats and playas from flooding or drying	3
184	Snowy plover	Climate change	Relative sea level rise expected to inundate or fragment low-lying habitats such as sandy beaches, barrier islands, and mudflats	3
185	Snowy plover	Climate change	Increasing frequency and severity of storms and changes in water temperatures will impact quality and quantity of coastal habitats and alter marine food webs	3
186	Swift fox	Development	**Grassland habitat loss, fragmentation and conversion from renewable and non-renewable energy development**	2
187	Swift fox	Development	Road mortality	3
188	Swift fox	Development	Trapping and incidental poisoning intended for coyotes	4
189	Swift fox	Development	Poisoning, reduction, eradication practices for food sources -prairie dogs and ground squirrels	4
190	Swift fox	Climate change	Climate change impacts to suitable grasslands	3
191	General CAs	-	What are the dominant invasive species in the ecoregion?	3
192	General CAs	-	What are the known and likely introduction vectors of invasive species?	3
193	General CAs	-	**What areas are available for energy development with limits on surface use?**	3
194	General CAs	-	**What areas are available for energy development?**	3
195	General CAs	-	What can be predicted about the severity of fires within the ecoregion?	3
196	General CAs	-	What is the current distribution of invasive species and what are the ecological affects in these areas?	3

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MQ no.	Conservation Element	Group	Question/issue	S
197	General CAs	-	**What is the known fire history of the ecoregion and what is the potential future fire regime?**	3
198	General CAs	-	What is the known lightning strike frequency in the ecoregion? Does this data show a significant trend over time?	3
199	General CAs	-	What is the probability of fire, based on model scenarios, near existing communities?	3
200	General CAs	-	Where are aquatic invasive species existing or a potential problem?	3
201	General CAs	-	Where are the areas with high fire frequency?	3
202	General CAs	-	What is the road density across the Southern Great Plains ecoregion rapid ecoregional assessment area?	2
203	General CAs	-	Where are the major energy corridors in the Southern Great Plains ecoregion rapid ecoregional assessment area?	2
204	General CAs	-	**Need to focus on use of native seed mix on Conservation Reserve Program lands**	4
205	General CAs	-	Air quality related to oil and gas development in eastern Colorado	4
206	General CAs	-	Groundwater quality related to oil and gas development in eastern Colorado	3
207	General CAs	-	**What is the fate of irrigated crop circles once groundwater cannot be used for irrigation purposes?**	4
208	General CAs	-	**What happens if Conservation Reserve Program lands are no longer active (if Farm Bill programs are dissolved)?**	4
209	General CAs	-	To what degree is that demand for water contributing to the ecological changes the area is experiencing due to prolonged drought?	3
210	General KEAs	-	**What are the regionally significant vegetation types?**	2
211	General KEAs	-	What is the current distribution and habitat of each species of concern?	2
212	General KEAs	-	**Where are aquifers and their recharge basins? What is the current and projected land use in these areas?**	3
213	General KEAs	-	**Where are species of concern at risk and from what?**	3
214	General KEAs	-	Where are the aquatic systems that support these keystone species and what are their condition?	3
215	General KEAs	-	**Where are the current locations of aquatic/riparian habitats, including rivers, streams, lakes, ponds, wetlands, springs and reservoirs?**	3
216	Integrated MQs	-	For conservation element species that overlap federal and Indian mineral development, have mitigation measures been followed, and did they have the desired effect?	3
217	Integrated MQs	-	**Given the current patterns of occurrence, what is the potential future distribution of invasive species?**	3
218	Integrated MQs	-	How is climate change going to affect invasive species?	3
219	Integrated MQs	-	**How will climate change affect water resources?**	3
220	Integrated MQs	-	How will extreme climate/weather events (such as drought, storms) affect species?	3
221	Integrated MQs	-	In areas that have experienced wildfires, what is the resulting vegetative structure compared to the desired structure?	4
222	Integrated MQs	-	In what areas will changes in fire regime and intensity result in landform change (such as mass wasting, erosion)?	4

Table B–1. Candidate management questions and issues, organized by Conservation Element (CE) and Change Agent (CA).—Continued

[Questions/issues appear as written by team members who submitted them. General questions related to key ecological attributes (KEAs), CAs, and integrated Management Questions (MQs) are at the bottom of the table. Column S refers to status (1 = addressed in Phase I CE narrative; 2 = addressed in Phase II (dependent on data availability); 3 = unknown whether the question can be addressed OR the question/issue can be partially addressed; and 4 = out of scope/cannot be addressed). Questions/issues in boldface font have been identified as priority issues by an Assessment Management Team member. Note: This information will be updated and the status of each issue is subject to change. BTPD, black-tailed prairie dog.]

MQ no.	Conservation Element	Group	Question/issue	S
223	Integrated MQs	-	What affect will climate change have on wildfires?	3
224	Integrated MQs	-	What and where are the vegetation types and seral stages that are carbon sinks and carbon sources? What actions in those vegetation types alter the sink/source balance?	3
225	Integrated MQs	-	What are potential increases in economic activities due to Change Agents (drought)?	3
226	Integrated MQs	-	**What are predicted changes in the distribution of vegetation types given climate change?**	3
227	Integrated MQs	-	**What areas are experiencing the highest rate of invasive species?**	3
228	Integrated MQs	-	**What areas have the greatest species richness, including seasonal use areas?**	3
229	Integrated MQs	-	**What areas have the potential for aquatic habitat restoration?**	3
230	Integrated MQs	-	**What areas represent opportunities to acquire high quality habitat, through fee acquisition or conservation easement?**	3
231	Integrated MQs	-	**What areas represent opportunities to conduct affective habitat restoration for conservation element species?**	3
232	Integrated MQs	-	**What habitats are critical for species sustainability?**	3
233	Integrated MQs	-	**What habitats have been or have the potential to be severely affected by invasive species and where are they?**	3
234	Integrated MQs	-	What invasive species have the potential for control and which ones do not?	3
235	Integrated MQs	-	What native species are likely to be most affected by invasive species?	3
236	Integrated MQs	-	Where and what regionally significant vegetation types are suitable for potential corridor connectors?	3
237	Integrated MQs	-	**Where are aquatic habitat strongholds for sensitive species that are intact and provide the best opportunity for protection, restoration and enhancement?**	3
238	Integrated MQs	-	**Where are areas in which groundwater extraction has the potential to change surface flow?**	3
239	Integrated MQs	-	Where are areas of high soil erodibility due to wind or water erosion if existing vegetation cover is removed?	3
240	Integrated MQs	-	Where are areas of state and federal high conservation value and restoration potential most vulnerable to a changing climate?	3
241	Integrated MQs	-	Where are corridors that have the greatest potential for loss due to Change Agents?	3
242	Integrated MQs	-	Where are current and planned oil and gas activities located and where do they overlap with essential wildlife habitat?	3
243	Integrated MQs	-	**Where are current riparian or aquatic areas currently at risk of fragmentation impoundment, diversion and lowered water tables due to development, mineral extraction, and agricultural and residential development?**	3
244	Integrated MQs	-	Where are high priority habitats that have the greatest potential of loss from Change Agents?	3

Table B–1. Candidate management questions and issues, organized by Conservation Element (CE) and Change Agent (CA).—Continued

[Questions/issues appear as written by team members who submitted them. General questions related to key ecological attributes (KEAs), CAs, and integrated Management Questions (MQs) are at the bottom of the table. Column S refers to status (1 = addressed in Phase I CE narrative; 2 = addressed in Phase II (dependent on data availability); 3 = unknown whether the question can be addressed OR the question/issue can be partially addressed; and 4 = out of scope/cannot be addressed). Questions/issues in boldface font have been identified as priority issues by an Assessment Management Team member. Note: This information will be updated and the status of each issue is subject to change. BTPD, black-tailed prairie dog.]

MQ no.	Conservation Element	Group	Question/issue	S
245	Integrated MQs	-	Where are invasive species affecting wildlife corridors?	3
246	Integrated MQs	-	Where are likely sources and sinks of discharge from such developments that may diminish quality of receiving waters and habitats (such as saline discharges)?	3
247	Integrated MQs	-	Where are potential areas to restore connectivity?	3
248	Integrated MQs	-	Where are potential carbon sequestration areas?	3
249	Integrated MQs	-	Where are potential habitat restoration areas?	3
250	Integrated MQs	-	Where are sensitive aquatic species at risk from stream connectivity or at risk from stream connectivity or risk from interbreeding with closely related non-native or invasive species?	3
251	Integrated MQs	-	**Where are the areas identified or designated for conservation?**	3
252	Integrated MQs	-	**Where are the areas of highest potential to change?**	3
253	Integrated MQs	-	Where are the important regionally significant terrestrial features, functions, and services across the ecoregional landscape?	3
254	Integrated MQs	-	Where are the most species increases or losses likely to occur due to changes in temperature increases or water availability?	3
255	Integrated MQs	-	**Where will aquatic habitats potentially be affected by Change Agents?**	3
256	Integrated MQs	-	Where will current conservation element vegetation types at greatest risk from Change Agents?	3
257	Integrated MQs	-	**Which habitats and species are most likely to be negatively impacted by climate change?**	3

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