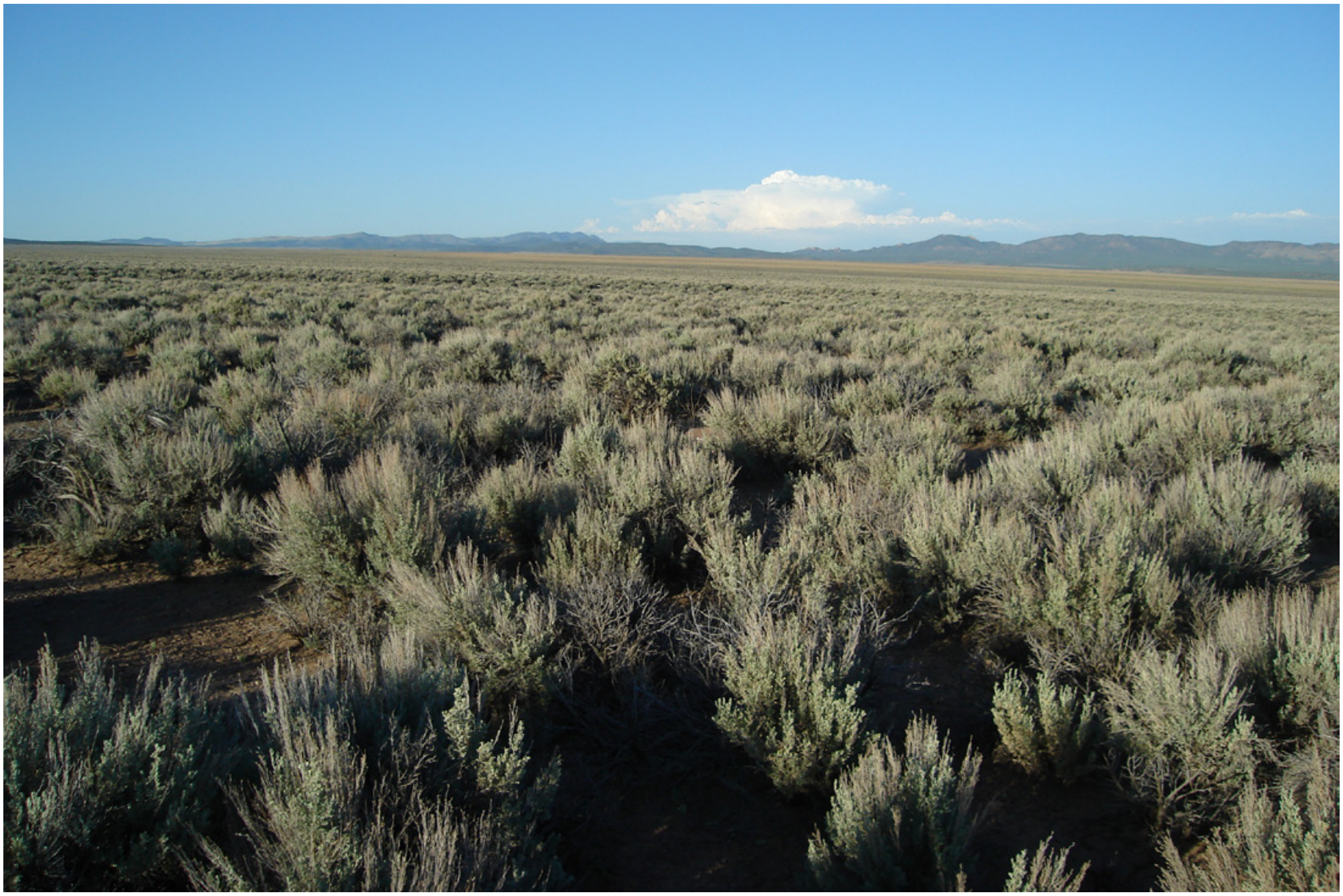
Prepared in cooperation with the Western Association of Fish and Wildlife Agencies, the Bureau of Land Management, and the U.S. Fish and Wildlife Service

Sagebrush Conservation Strategy—Challenges to Sagebrush Conservation



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Cover caption. Photograph of big sagebrush (*Artemisia tridentata*) in the Parowan Valley, Utah, 2006. Photograph by Steven Hanser, U.S. Geological Survey.

Sagebrush Conservation Strategy—Challenges to Sagebrush Conservation

By Thomas E. Remington, Patricia A. Deibert, Steven E. Hanser, Dawn M. Davis, Leslie A. Robb, and Justin L. Welty

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U.S. Department of the Interior
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U.S. Geological Survey, Reston, Virginia: 2021

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Suggested citation:
Remington, T.E., Deibert, P.A., Hanser, S.E., Davis, D.M., Robb, L.A., and Welty, J.L., 2021, Sagebrush conservation strategy—Challenges to sagebrush conservation: U.S. Geological Survey Open-File Report 2020–1125, 327 p., https://doi.org/10.3133/ofr20201125.

ISSN 0196-1497 (print)
ISSN 2331-1258 (online)

Preface

The sagebrush (*Artemisia* spp.) biome, its wildlife, and the services and benefits it provides people and local communities are at risk. Development in the sagebrush biome, for many purposes, has resulted in multiple and often cumulative negative impacts. These impacts, ranging from simple habitat loss to complex, interactive changes in ecosystem function, continue to accelerate even as the need grows for the resources provided by this biome. This Sagebrush Conservation Strategy is intended to provide guidance so that the unparalleled collaborative efforts to conserve the iconic greater sage-grouse (*Centrocercus urophasianus*) by State and Federal agencies, academia, Tribes, nongovernmental organizations, and stakeholders can be expanded to the entire sagebrush biome to benefit the people and wildlife that depend on this ecosystem.

The Sagebrush Conservation Strategy will be presented in two parts. Part I, Challenges to Sagebrush Conservation (this volume), is an overview and assessment of the challenges facing land managers and landowners in conserving sagebrush ecosystems, including change agents such as invasive plants, altered fire regimes, climate, land use and development, and other challenges associated with conservation, including restoration, communication, adaptive management, and monitoring. Part I updates and extends to other sagebrush-obligate, near-obligate, and -dependent species and human communities the information and content provided by the two-part "Science Framework for the Conservation and Restoration of the Sagebrush Biome—Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to Long-Term Strategic Conservation Actions," incorporating aspects of the threat assessments, habitat prioritization methods, and resistance and resilience concepts. When completed, Part II will summarize conservation needs at ecoregional scales, provide an analysis of barriers and impediments to successful conservation of the sagebrush biome at those scales, and present nonregulatory strategies developed through a stakeholder engagement process to overcome these challenges.

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Conversion Factors

International System of Units to U.S. customary units

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)
millimeter (mm)	0.03937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
kilometer (km)	0.5400	mile, nautical (nmi)
meter (m)	1.094	yard (yd)
	Area	
square meter (m^2)	0.0002471	acre
hectare (ha)	2.471	acre
square hectometer (hm^2)	2.471	acre
square kilometer (km^2)	247.1	acre
square centimeter (cm^2)	0.001076	square foot (ft^2)
square meter (m^2)	10.76	square foot (ft^2)
square centimeter (cm^2)	0.1550	square inch (ft^2)
square hectometer (hm^2)	0.003861	section (640 acres or 1 square mile)
hectare (ha)	0.003861	square mile (mi^2)
square kilometer (km^2)	0.3861	square mile (mi^2)
	Mass	
gram (g)	0.03527	ounce, avoirdupois (oz)
kilogram (kg)	2.205	pound avoirdupois (lb)
metric ton (t)	1.102	ton, short [2,000 lb]
metric ton (t)	0.9842	ton, long [2,240 lb]
	Pressure	
kilopascal (kPa)	0.009869	atmosphere, standard (atm)
kilopascal (kPa)	0.01	bar
kilopascal (kPa)	0.2961	inch of mercury at 60 °F (in Hg)
kilopascal (kPa)	0.1450	pound-force per inch (lbf/in)
kilopascal (kPa)	20.88	pound per square foot (lb/ft^2)
kilopascal (kPa)	0.1450	pound per square inch (lb/ft^2)
	Energy	
joule (J)	0.0000002	kilowatthour (kWh)
	Application rate	
kilogram per hectare per year ([kg/ha]/yr)	0.8921	pound per acre per year ([lb/acre]/yr)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:
°F = (1.8 × °C) + 32.

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:
°C = (°F – 32) / 1.8.

Common and Scientific Names of Animal Species in this Report

Common name	Latin name	Chapter
Amargosa toad	*Anaxyrus nelsoni*	I
American bison	*Bison bison*	B, E
American kestrel	*Falco sparverius*	O
ant	*Formicidae* spp.	J
Arizona black rattlesnake	*Crotalus cerberus*	I
Arizona toad	*Anaxyrus microscaphus*	I
badger	*Taxidea taxus*	P
Baja California treefrog	*Pseudacris hypochondriaca*	I
bald eagle	*Haliaeetus leucocephalus*	P
barred tiger salamander	*Ambystoma mavortium*	I
beetle	*Coleoptera*	J
Bell's sparrow	*Artemisiospiza belli*	C
bighorn sheep	*Ovis canadensis*	N
black toad	*Anaxyrus exsul*	I
black-footed ferret	*Mustela nigripes*	H
black-necked gartersnake	*Thamnophis cyrtopsis*	I
black-tailed deer	*Odocoileus hemionus sitkensis* *Odocoileus hemionus columbianus*	G
black-tailed jackrabbit	*Lepus californicus*	A, E, H
black-tailed prairie dog	*Cynomys ludovicianus*	H
black-tailed rattlesnake	*Crotalus molossus*	I
boreal chorus frog	*Pseudacris maculata*	I
Brewer's sparrow	*Spizella breweri*	A, C, J, L, M, O, P, Q
burro	*Equus asinus*	front matter, L, N, R
burrowing owl	*Athene cunicularia*	O
canyon treefrog	*Hyla arenicolor*	I
chestnut-collared longspur	*Calcarius ornatus*	P
Chihuahuan spotted whiptail	*Aspidoscelis exsanguis*	I
chukar	*Alectoris chukar*	O
Clark's spiny lizard	*Sceloporus clarkii*	I
coachwhip	*Coluber flagellum*	I
Columbia Plateau pocket mouse	*Perognathus parvus*	O
Columbia spotted frog	*Rana luteiventris*	I, Q, S
Columbian sharp-tailed grouse	*Tympanuchus phasianellus columbianus*	O
common chuckwalla	*Sauromalus ater*	I
common raven	*Corvus corax*	O, P
common sagebrush lizard	*Sceloporus graciosus*	A, I, Q
common side-blotched lizard	*Uta stansburiana*	I

Common name	Latin name	Chapter
cottontail rabbits	*Sylvilagus* spp.	E
coyote	*Canis latrans*	P
dark kangaroo mouse	*Microdipodops megacephalus*	A, H, J, K
deer (general)	*Odocoileus* spp.	B
deer mouse	*Peromyscus maniculatus*	L, P
desert horned lizard	*Phrynosoma platyrhinos*	I
desert night lizard	*Xantusia vigilis*	I
desert nightsnake	*Hypsiglena chlorophaea*	A, I
desert spiny lizard	*Sceloporus magister*	I
domestic cat	*Felis catus*	P
domestic cow	*Bos taurus*	G
domestic dog	*Canis lupus familiaris*	P
domestic sheep	*Ovis aries*	B
eastern collared lizard	*Crotaphytus collaris*	I
eastern milksnake	*Lampropeltis triangulum*	I
elk	*Cervus canadensis*	B, G, J, L, O, P, N
flower weevil	*Larinus* spp.	K
Gilbert's skink	*Plestiodon gilberti*	I
golden eagle	*Aquila chrysaetos*	front matter, P, O
gophersnake	*Pituophis catenifer*	I
grasshopper sparrow	*Ammodramus savannarum*	Q
grasshoppers	*Orthoptera* spp.	J
gray flycatcher	*Empidonax wrightii*	A, C, J, M
Great Basin collared lizard	*Crotaphytus bicinctores*	I
Great Basin pocket mouse	*Perognathus mollipilosus*	A, O, M
Great Basin spadefoot	*Spea intermontana*	A, I, Q
Great Plains toad	*Anaxyrus cognatus*	I
greater sage-grouse	*Centrocercus urophasianus*	front matter, A, B, D, E, I, J, K, L, M, N, O, P, Q, R, S, T
greater short-horned lizard	*Phrynosoma hernandesi*	A, I, Q
green-tailed towhee	*Pipilo chlorurus*	A, B, C, M, Q
Gunnison sage-grouse	*Centrocercus minimus*	front matter, A, B, D, J, L, M, P, Q
hoary bat	*Lasiurus cinereus*	O
horned lark	*Eremophila alpestris*	O, P
horse	*Equus caballus*	front matter, A, L, N, R
Idaho giant salamander	*Dicamptodon aterrimus*	I
Idaho ground squirrel	*Urocitellus brunneus*	H
Inyo Mountains salamander	*Batrachoseps campi*	I
jackrabbit	*Lepus* spp.	H, P

Common name	Latin name	Chapter
Jemez Mountains salamander	*Plethodon neomexicanus*	I
kinglet	*Regulus* spp.	O
little brown bat	*Myotis lucifugus*	O
little striped whiptail	*Aspidoscelis inornata*	I
long-nosed leopard lizard	*Gambelia wislizenii*	I
long-nosed snake	*Rhinocheilus lecontei*	I
long-toed salamander	*Ambystoma macrodactylum*	I
many-lined skink	*Plestiodon multivirgatus*	I
Merriam's shrew	*Sorex merriami*	A, H
Mexican spadefoot	*Spea multiplicata*	I
mosquitoes	*Culex* spp.	L
Mount Lyell salamander	*Hydromantes platycephalus*	I
mule deer	*Odocoileus hemionus*	front matter, A, G, J, K, L, M, O, P, Q, R
North American racer	*Coluber constrictor*	I
Northern Idaho ground squirrel	*Urocitellus brunneus brunneus*	H
northern leopard frog	*Lithobates pipiens*	I, Q
northern pocket gopher	*Thomomys talpoides*	H
northern rubber boa	*Charina bottae*	I
northern tree lizard	*Urosaurus ornatus wrightii*	I
Ord's kangaroo rat	*Dipodomys ordii*	A, H
Oregon spotted frog	*Rana pretiosa*	I
Ornate tree lizard	*Urosaurus ornatus*	I
Pacific tree frog	*Pseudacris sierra*	I
Pai striped whiptail	*Aspidoscelis pai*	I
panamint alligator lizard	*Elgaria panamintina*	I
pinyon jay	*Gymnorhinus cyanocephalus*	A, C, M, Q, R
pinyon mouse	*Peromyscus truei*	M
plains gartersnake	*Thamnophis radix*	I
plains hog-nosed snake	*Heterodon nasicus*	I
Plains spadefoot	*Spea bombifrons*	I, Q
plateau fence lizard	*Sceloporus tristichus*	I
plateau striped whiptail	*Aspidoscelis velox*	I
prairie dog	*Cynomys* spp.	L
prairie rattlesnake	*Crotalus viridis*	I
Preble's shrew	*Sorex preblei*	A, F, H
pronghorn	*Antilocapra americana*	front matter, A, B, F, G, J, N, O, P, Q
pygmy rabbit	*Brachylagus idahoensis*	front matter, A, D, E, J, M, P, Q, R, S
pygmy short-horned lizard	*Phrynosoma douglasii*	A, I, Q

Common name	Latin name	Chapter
red fox	*Vulpes vulpes*	O
red-spotted toad	*Anaxyrus punctatus*	I
red-tailed hawk	*Buteo jamaicensis*	O
ring-necked pheasant	*Phasianus colchicus*	O
Rocky Mountain tailed frog	*Ascaphus montanus*	I
sage thrasher	*Oreoscoptes montanus*	A, C, J, L, M, O, P, Q
sagebrush lizard	*Sceloporus graciosus*	A, Q
sagebrush sparrow	*Artemisiospiza nevadensis*	A, C, J, L, M, O, P, Q
sagebrush vole	*Lemmiscus curatus*	A, D, H, O
sage-grouse (general)	*Centrocercus* spp.	B, O, P, Q, S
savannah sparrow	*Passerculus sandwichensis*	Q
sidewinder	*Crotalus cerastes*	I
Sierra garter snake	*Thamnophis couchii*	I
Sierra Nevada yellow-legged frog	*Rana sierra*	I
Sierran treefrog	*Pseudacris sierra*	I
Smith’s black-headed snake	*Tantilla hobartsmithi*	I
smooth greensnake	*Opheodrys vernalis*	I
Sonoran Mountain kingsnake	*Lampropeltis pyromelana*	I
Sonoran pronghorn	*Antilocapra americana sonoriensis*	E
Southern Idaho ground squirrel	*Urocitellus endemicus*	A, O
speckled rattlesnake	*Crotalus mitchellii*	I
striped whipsnake	*Coluber taeniatus*	I
sylvatic plague	*Yersinia pestis*	H, L
terrestrial gartersnake	*Thamnophis elegans*	I
thick-billed longspur	*Rhynchophanes mccownii*	O
tiger whiptail	*Aspidoscelis tigris*	I
vesper sparrow	*Pooecetes gramineus*	M
West Nile virus	*Flavivirus* spp.	L
western banded gecko	*Coleonyx variegatus*	I
western fence lizard	*Sceloporus occidentalis*	I
western groundsnake	*Sonora semiannulata*	I
western lyre snake	*Trimorphodon biscutatus*	I
western meadowlark	*Sturnella neglecta*	O
western patch-nosed snake	*Salvadora hexalepis*	I
western pond turtle	*Actinemys marmorata*	I
western rattlesnake	*Crotalus oreganus*	I
western skink	*Plestiodon skiltonianus*	I
western toad	*Anaxyrus boreas*	I, S
white-tailed deer	*Odocoileus virginianus*	G, L
white-tailed jackrabbit	*Lepus townsendii*	H
white-tailed prairie dog	*Cynomys leucurus*	A, H

Common name	Latin name	Chapter
Woodhouse's toad	*Anaxyrus woodhousii*	I
woodland caribou	*Rangifer tarandus caribou*	L
Wyoming ground squirrel	*Urocitellus elegans*	A, H
Wyoming pocket gopher	*Thomomys clusius*	H
Wyoming toad	*Anaxyrus baxteri*	I
yellow-backed spiny lizard	*Sceloporus uniformis*	I
Yosemite toad	*Anaxyrus canorus*	I
zebra mussel	*Dreissena polymorpha*	T
zebra-tailed lizard	*Callisaurus draconoides*	I

Common and Scientific Names of Plant Species in this Report

Common name	Latin name	Chapters
alfalfa	*Medicago sativa*	D, H, P
alkali sagebrush	*Artemisia arbuscula longiloba*	D
antelope bitterbrush	*Purshia tridentata*	A, E, H, J
aspen	*Populus tremuloides*	L
basin big sagebrush	*Artemisia tridentata tridentata*	A, D, J, K
big sagebrush	*Artemisia tridentata*	A, B, C, E, H, L, M, O, P, R
big sagebrush, related	*Artemisia, subgenus Tridentatae*	A
black sagebrush	*Artemisia nova*	A, D, E, L, J
bluebunch wheatgrass	*Pseudoroegneria spicata*	L, R
brome grass	*Bromus* spp.	front matter
Canada thistle	*Cirsium arvense*	K
cheatgrass	*Bromus tectorum*	B, C, E, F, H, I, J, K, L, M, N, P, Q
clover species	*Trifolium* spp.	L
common crupina	*Crupina vulgaris*	K
corn	*Zea mays*	O
creosote bush	*Larrea tridentata*	H
crested wheatgrass	*Agropyron cristatum*	A, C, I, K, L, R
curl-leaf mountain mahogany	*Cercocarpus ledifolius*	E
currant	*Ribes* spp.	A
dalmation toadflax	*Linaria dalmatica*	K
diffuse knapweed	*Centaurea diffusa*	K
Douglas-fir	*Pseudotsuga menziesii*	M, R
dyer's woad	*Isatis tinctoria*	K
field brome	*Bromus arvensis*	K

Common name	Latin name	Chapters
fir	*Abies* spp.	M
four-wing saltbush	*Atriplex canescens*	A, H
fringed sagebrush	*Artemisia frigida*	D
Gardner's saltbush	*Atriplex gardneri*	H
greasewood	*Sarcobatus vermiculatus*	A, H
green rabbitbrush	*Ericameria teretifolia*	J
halogeton	*Halogeton glomeratus*	K
hardheads	*Rhaponticum repens*	K
hoary cress	*Lepidium draba*	K
horsebrush	*Tetradymia* spp.	H
Iberian starthistle	*Centaurea iberica*	K
juniper	*Juniperus* spp.	front matter, C, H, I, J, K, L, M, Q, R, S
knapweed	*Centaurea* spp.	K
leafy spurge	*Euphorbia esula*	K
Lewis' flax	*Linum lewisii*	L
limber pine	*Pinus flexilis*	L
lodgepole pine	*Pinus contorta*	H
low sagebrush	*Artemisia arbuscula*	A, D, E, K, L, J
Mediterranean sage	*Salvia aethiopis*	K
medusahead rye	*Taeniatherum caput-medusae*	E, F, K, M
mountain big sagebrush	*Artemisia tridentata vaseyana*	A, D, J, R
mountain mahogany	*Cercocarpus* spp.	C, H
musk thistle	*Carduus nutans*	K
North Africa grass	*Ventenata dubia*	K
oak	*Quercus* spp.	H
pinyon pine	*Pinus edulis* and *Pinus monophylla*	front matter, C, H, I, J, K, L, M, R
ponderosa pine	*Pinus ponderosa*	M, R
prickly lettuce	*Lactuca serriola*	K
prickly pear cactus	*Opuntia* spp.	E, F
prickly phlox	*Linanthus pungens*	J
prickly Russian thistle	*Salsola tragus*	J, K
purple starthistle	*Centaurea calcitrapa*	K
rabbitbrush	*Chrysothamnus* spp., *Ericameria* spp., *Lorandersonia* spp.	A, E, H
red brome	*Bromus rubens*	K, L
rubber rabbitbrush	*Ericameria nauseosa*	J
rush skeletonweed	*Chondrilla juncea*	K
Russian knapweed	*Rhaponticum repens*	K

Common name	Latin name	Chapters
Russian wildrye	*Psathyrostachys junceus*	R
sagebrush	*Artemisia* spp.	front matter, A, B, C, D, E, F, G, H, I, J, K, L, M, O, P, Q, R, S, T
Sandberg's bluegrass	*Poa secunda*	R
Scotch thistle	*Onopordum acanthium*	K
serviceberry	*Amelanchier* spp.	A
shadscale	*Atriplex confertifolia*	A, H
Siberian wheatgrass	*Agropyron fragile*	R
silver sagebrush	*Artemisia cana*	A, D, L
Snake River wheatgrass	*Elymus wawawaiensis*	K
snowberry	*Symphoricarpos* spp.	A
snowfield big sagebrush	*Artemisia tridentata spiciformis*	A
spiny hopsage	*Grayia spinosa*	A, H
spotted knapweed	*Centaurea stoebe and Centaurea maculosa*	K
squarrose knapweed	*Centaurea virgata*	K
Sulphur cinquefoil	*Potentilla recta*	K
thistle	*Cirsium* spp.	K
threetip sagebrush	*Artemisia tripartita*	A, E, J
timothy hay	*Phleum pratense*	H
ventenata	*Ventenata dubia*	K
western juniper	*Juniperus occidentalis*	M, Q
wheat	*Triticum* spp.	P
whitetop	*Lepidium draba*	K
willow	*Salix* spp.	H, R
winterfat	*Krascheninnikovia lanata*	A, H
Wyoming big sagebrush	*Artemisia tridentata wyomingensis*	A, D, J, K, R
yellow toadflax	*Linaria vulgaris*	K
yellow-star thistle	*Centaurea solstitialis*	K
Yucca	*Yucca* spp.	H

Abbreviations

AHM	adaptive harvest management
AIM	assessment inventory and monitoring
ALE	agricultural land easement
AML	appropriate management level
APE	area of potential effects
APIA	Animal Protection Institute of America
APLIC	Avian Powerline Interaction Committee
ARM	adaptive resource management
AUM	animal unit month
BACI	before-after control-impact
BAR	burned area rehabilitation
BAER	burned area emergency response
BBS	breeding bird survey
BEA	bank enabling agreement
BIA	Bureau of Indian Affairs
BLM	Bureau of Land Management
BSCC	biological soil crust community
CCAA	Candidate Conservation Agreement with Assurances
CED	conservation efforts database
CPW	Colorado Parks and Wildlife
CWD	chronic wasting disease
COSEWIC	Committee on the Status of Endangered Wildlife in Canada
COT	conservation objectives team
CRM	customer relationship management
CRP	conservation reserve program
CWMA	cooperative weed management area
CWPP	community wildfire protection plan
DEQ	Department of Environmental Quality
DOE	U.S. Department of Energy
DOI	U.S. Department of the Interior
DPS	Distinct Population Segment
EDDMapS	Early Detection and Distribution Mapping System
EDRR	early detection and rapid response

EIS	environmental impact statement
EMODIS	enhanced moderate resolution imaging spectroradiometer
EO	executive order
EQIP	environmental quality incentive program
ES	emergency stabilization
ESA	Endangered Species Act of 1973 (16 U.S.C. 1531 et seq.)
ESD	ecological site description
ESR	emergency fire stabilization and rehabilitation
FWS	U.S. Fish and Wildlife Service
FIA	Forest Inventory and Analysis
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act of 1996 (7 U.S.C. 136 et seq.)
FLPMA	Federal Land Policy and Management Act of 1976 (43 U.S.C. 1701–1785)
FY	fiscal year
FS	U.S. Department of Agriculture, Forest Service
GAP	Gap Analysis Program
GCM	general circulation model
GeoMAC	Geospatial Multi-Agency Coordination
GHG	greenhouse gas
GHMA	general habitat management area
GIS	geographic information system
GPS	global positioning system
HA	herd area
HAF	habitat assessment framework
HMA	herd management area
HQT	habitat quantification tool
IBLA	Interior Board of Land Appeals
ICCATF	Interagency Climate Change Adaptation Task Force
IIRH	Interpreting Indicators of Rangeland Health
IM	instruction memorandum
IMBCR	Integrated Monitoring in Bird Conservation Regions
IPM	integrated population model
IRFMS	integrated rangeland fire management strategy
IRMA	integrated resource management application
IUCN	International Union for Conservation of Nature

LMF	landscape monitoring framework
LTDL	Land Treatment Digital Library
MDWG	Mule Deer Working Group
MOA	minimum occupied area
MOU	memorandum of understanding
MRLC	Multi-Resolution Land Characteristics
MTBS	Monitoring Trends in Burn Severity
MW	megawatt
MZ	management zone
NAIP	National Agriculture Imagery Program
NABCI	North American Bird Conservation Initiative
NDVI	normalized difference vegetation index
NEPA	National Environmental Policy Act (42 U.S.C. 4321 et seq.)
NFMA	National Forest Management Act of 1976 (16 U.S.C. 1600–1614)
NFWF	National Fish and Wildlife Foundation
NGO	nongovernmental organization
NLCD	National Land Cover Dataset
NPS	U.S. Department of the Interior National Park Service
NRCS	Natural Resources Conservation Service
NRI	national resources inventory
NSO	no surface occupancy
NTT	National Technical Team
DNA	deoxyribonucleic acid
OHV	off-highway vehicle
OSMRE	Office of Surface Mining Reclamation and Enforcement
PAC	priority areas for conservation
PEIS	programmatic environmental impact statement
PHMA	priority habitat management area
PIF	Partners in Flight
PSM	plant secondary metabolites
PRIA	Public Rangelands Improvement Act of 1978 (43 U.S.C. 1901)
PRISM	parameter-elevation regressions on independent slopes model
PUP	pesticide use plan

RAWS	remote automatic weather station
RCP	representative concentration pathway
REA	rapid ecoregional assessment
RFPA	Rangeland Fire Protection Association
ROD	records of decision
RMP	resource management plan
ROW	right-of-way
SDM	species distribution model
SEPA	State Environmental Protection Act
SFA	sagebrush focal area
SGI	Sage Grouse Initiative
SMCRA	Surface Mining Control and Reclamation Act
SNOTEL	snow telemetry
SO	secretarial order
STM	state-and-transition model
SWAP	State Wildlife Action Plan
SWP	soil water potential
TBGPEA	Thunder Basin Grasslands Prairie Ecosystem Association
TOC	total organic carbon
USDA	U.S. Department of Agriculture
USGS	U.S. Geological Survey
WAFWA	Western Association of Fish and Wildlife Agencies
WBT	wild burro territories
WFMI	Wildland Fire Management Information
WFRHBA	Wild Free-Roaming Horses and Burros Act of 1971 (16 U.S.C. ch. 30 1331 et seq.)
WGA	Western Governors' Association
WHB	wild horses and burros
WHBT	wild horse and burro territory
WHT	wild horse territory
WILD	Wildlife Innovation and Longevity Driver Act
WRI	Watershed Restoration Initiative
WSB	weed suppressive bacteria

Executive Summary

The sagebrush (*Artemisia* spp.) biome has provided important natural resources to inhabitants of the West since before Euro-American settlement. Sagebrush now occupies less than 55 percent of its historical extent, and more than 350 species of plants and animals associated with sagebrush are considered species of conservation concern. Several species considered sagebrush obligates have been petitioned for listing under the Endangered Species Act of 1973 (16 U.S.C. 1531 et seq.), including greater sage-grouse (*Centrocercus urophasianus*), Gunnison sage-grouse (*Centrocercus minimus*; listed as threatened), and pygmy rabbit (*Brachylagus idahoensis*). Other sagebrush-dependent species, such as pronghorn (*Antilocapra americana*) and mule deer (*Odocoileus hemionus*), have experienced significant population declines.

The loss and degradation of sagebrush continues because of a variety of change agents including altered fire regimes, invasive plant species, conifer expansion, overabundant free-roaming equids, and human land uses, including energy development, cropland conversion, infrastructure, and improper livestock grazing. Climate changes, including warmer temperatures and altered amounts and timing of precipitation, have and will likely increasingly compound negative effects to sagebrush ecosystems from all these threats. Warming climates, and associated decreases in rainfall during the growing season, are expected to increase the frequency, size, and intensity of wildfires in much of the sagebrush biome. The expansion of annual grass communities has resulted in large-scale wildfires that have consumed large expanses of sagebrush in recent years, threatening efforts to conserve sage-grouse and other sagebrush-associated wildlife. Since 2000, 20.6 percent of greater sage-grouse priority habitat management areas within the Great Basin has burned. Similarly, 17 percent of areas deemed highly suitable for pygmy rabbits burned within the Great Basin from 2000 to 2018. In the eastern portion of the sagebrush biome, the invasive annual grass and fire cycle is not yet a significant concern, but invasive brome (*Bromus* spp.) grass species are well established, and fire frequency is expected to increase.

Expansion of conifers, principally pinyon pine (*Pinus* spp.) and juniper (*Juniperus* spp.), into sagebrush shrublands is a pervasive cause of loss and degradation of sagebrush, with negative effects to hydrology, forage available for grazing, and sagebrush-associated wildlife. Efforts to restore ecosystem function and wildlife occupancy through removal of early phase conifer expansion have been successful and are ongoing across the sagebrush biome. The large majority (87 percent) of conifer reduction efforts within the sagebrush biome in the last 4–6 years has occurred in Nevada, Oregon, and Utah through State and Federal initiatives, although this represents only 1.6 percent of the area supporting trees across the sagebrush biome.

Overabundant free-roaming equids (wild horses [*Equus caballus*] and burros [*Equus asinus*]) are increasingly degrading sagebrush ecosystem function and reducing the forage and water available for domestic and native wildlife species. In March 2019, appropriate management level for U.S. Department of the Interior, Bureau of Land Management-administered herds was 26,690, but an estimated 88,090 wild horses and burros were inhabiting designated herd management areas. Without management to reduce growth rates, wild horse and burro populations could more than double within 4 years.

Mining and energy development are significant causes of loss and degradation of sagebrush where those activities occur. Approximately 8 percent of all sagebrush habitats are directly affected by oil and gas development, with greater than 20 percent of sagebrush habitats

affected in the Rocky Mountain area. Loss and degradation of habitat and disturbance associated with these activities have significant effects on sage-grouse, mule deer, and other sagebrush-dependent and -associated wildlife. Federal and State regulations, policies, and programs have recently been developed to mitigate impacts of energy development and other permitted activities to sage-grouse, but the effectiveness of these approaches for sage-grouse or other sagebrush-associated species is largely unknown.

Approximately 10 percent of the sagebrush biome has been converted to cropland, typically at low elevations with deep, fertile soils. Conversion to cropland remains a significant cause of loss of sagebrush in some areas, with the slightly wetter and more productive soils of eastern Washington, eastern Montana, and Wyoming experiencing the most conversion. Sagebrush-obligate species may abandon or be extirpated from areas if the proportion of sagebrush on the landscape falls too low.

All human uses of sagebrush landscapes impact ecological processes and wildlife, but effects can be positive or negative and vary tremendously in degree depending on the land use, site conditions, and species. For example, well-managed grazing can foster productive rangeland for cattle and wildlife; however, poorly managed grazing can lead to a reduction in grass cover and soil erosion and compaction. Also, tall structures and other infrastructure can fragment habitat leading to avoidance by some species, such as ground-nesting birds, but can also provide additional perching habitat for species of concern such as golden eagles (*Aquila chrysaetos*).

Regulatory and voluntary approaches are being implemented across the sagebrush biome to help reduce negative impacts from human land use. Federal land management agencies have established range condition targets to support sustainable grazing practices on public lands. State, Federal, and not-for-profit partners are providing voluntary protection mechanisms (for example, conservation easements) and cost-share opportunities (for example, the U.S. Department of Agriculture, Natural Resources Conservation Service, Environmental Quality Incentive Program) to help private landowners conserve and maintain resilient rangelands. Federal, State, and county entities are closing roads, constructing wildlife road crossing structures, and managing recreational activities to minimize human and wildlife conflicts. Mitigation programs are also active in many States to help avoid and offset adverse impacts from ongoing land use development, such as new pipelines or transmission lines. At present, mitigation programs within the sagebrush biome are directed towards sage-grouse conservation, and their effectiveness at addressing cumulative effects or conserving other sagebrush-dependent species is unknown.

The current management focus within the sagebrush biome is primarily on sage-grouse conservation, and there are significant State and Federal efforts and collaborations with private landowners and industry to address threats and restore degraded sagebrush habitats. Greater sage-grouse are widely considered a conservation umbrella, meaning efforts for this wide-ranging species may also conserve habitats of other sagebrush-obligate, -dependent, or -associated species. An analysis of the coverage for the sage-grouse umbrella indicates that conservation efforts may serve more as a model of an effective collaborative conservation approach for conserving sagebrush species rather than as a replacement for broader conservation efforts. For instance, only 22 percent of sagebrush occurs within priority habitat management areas for sage-grouse where most regulatory protections are in place, and threats like invasive species and fire are not well addressed through regulatory means alone. Conservation efforts would likely be more efficient and effective if sagebrush habitats are prioritized for conservation emphasis within an ecological context of resistance and resilience and in a manner that better captures important seasonal habitats for sagebrush-dependent and associated species and sage-grouse, while respecting human needs for natural resources.

Meeting conservation goals for sage-grouse, mule deer, pygmy rabbits, and other sagebrush-associated wildlife will require extensive restoration of sagebrush communities already converted or degraded by the change agents previously discussed. This will be a daunting task given the amount of habitat in need of restoration (about half of remaining sagebrush landscapes are considered degraded to some degree), vast geographies involved, limited native seed availability, and confounding and interacting effects of weather, climate change, invasive plants, and recurrent fire. Restoration at the landscape scale (ecoregion to planning unit) will require collaboration with partners across jurisdictional boundaries. The "Science Framework for the Conservation and Restoration of the Sagebrush Biome—Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to Long-Term Strategic Conservation Actions," Parts 1 and 2, developed through Federal and State collaboration, describes tools and approaches to prioritize sagebrush landscapes for restoration (and other conservation actions). Improvements in planning that now prioritize areas needing and likely to have a positive response to intervention, adaptive management approaches that incorporate learning, involvement of multiple stakeholders that allows for repeated interventions over longer time periods, and current research improving the understanding of factors affecting restoration success and restoration techniques are and will continue to improve restoration success. However, opportunities remain to better incorporate current knowledge into restoration practice. The greatest challenge in restoration of sagebrush landscapes will likely be obtaining resources to scale up efforts to the degree necessary to meet restoration objectives.

Adaptive management informed by monitoring is recognized as important and desirable for managing natural resources, yet it is seldom implemented effectively. Management of the sagebrush biome to retain natural resources for human use and conserve associated wildlife across 14 States and complex ownership patterns will require a coordinated and adaptive management construct. Adaptive resource management is an evolving process involving a sequential cycle of learning and adaptation. Although adaptive resource management approaches sound complex and difficult to implement, State and Federal governments have experience with them including harvest management under the North American Waterfowl Management Plan and big game management programs within State wildlife agencies.

Communication, outreach, and engagement are crucial components of successful natural resource management. Effective, strategic communication can enhance grassroots conservation efforts and build the next generation of managers, practitioners, scientists, and communicators who will care for the sagebrush ecosystem and stimulate or sustain public participation in sagebrush conservation issues. With more than 50 percent of the sagebrush ecosystem managed by Federal and State agencies, public support is essential to ensure a sustainable future for this ecosystem.

Successful and sustainable sagebrush conservation will depend on active engagement from entities that are currently active in sagebrush ecosystem management efforts (for example, those contributing to this strategy), those deriving their income from sagebrush landscapes, extractive industries, and outdoor recreationists, as well as various sectors of the broader American public. This report, "Sagebrush Conservation Strategy—Challenges to Sagebrush Conservation," provides an overview of the issues facing the sagebrush biome and the needs of the humans and wildlife that depend on this ecosystem.

PART I. Importance of the Sagebrush Biome to People and Wildlife

Chapter A. Introduction to the Sagebrush Biome

By Thomas E. Remington,[1] David L. Tart,[2] Mary E. Manning,[3] Justin L. Welty,[4] and David S. Pilliod[4]

Executive Summary

Management perspectives toward sagebrush (*Artemisia* spp.) and associated policies and regulations have evolved during the last century. Early management focused on removing sagebrush to create croplands or grasslands, the latter of which were often seeded with nonnative browse species for livestock. Because of cumulative impacts of historical grazing practices, sagebrush removal, and conversion efforts, the sagebrush habitats we manage today bear little resemblance to those occurring before European settlement. Loss of native species diversity and an influx of nonnative species have reduced the resilience of sagebrush ecosystems. More recent management perspectives have attempted to balance conservation and restoration of sagebrush communities with agriculture, resource extraction, and recreation. The focus of sagebrush management and conservation shifted abruptly when concern for sage-grouse (*Centrocercus* spp.) coalesced State and Federal agencies, nongovernmental organizations, and landowners in formal and informal partnerships to keep greater sage-grouse (*Centrocercus urophasianus*) from being listed under the Endangered Species Act of 1973 (ESA; 16 U.S.C. 1531 et seq.).

There is no single coordinated effort or plan to effect greater sage-grouse conservation, rather there are 11 different State plans and 98 Federal land use plans. These plans are implemented by Federal and State land and wildlife management agencies across the biome and supplemented by conservation practices implemented through State programs; the U.S. Department of Agriculture, Natural Resources Conservation Service, Sage Grouse Initiative; and the efforts of numerous nongovernmental agencies, working groups, and individual landowners. Future management of the sagebrush biome may be more effective with a move toward maintenance of ecosystem resilience and resistance and conservation of the entire suite of sagebrush-dependent and -associated species.

[1]Western Association of Fish and Wildlife Agencies.

[2]U.S. Department of Agriculture, Forest Service (retired).

[3]U.S. Department of Agriculture, Forest Service.

[4]U.S. Geological Survey.

Introduction

Big sagebrush (*Artemisia tridentata*) and related sagebrushes (*Artemisia* subgenus *Tridentatae*; Shultz, 2009) are uniquely North American plants. Studies of fossil pollen suggest that the earliest woody lineage of sagebrush appeared in the region of the Columbia Basin in Oregon (Davis, 1998), with widespread dominance of sagebrush taxa across much of the arid West occurring as recently as about 12,000 years ago (Shultz, 2009). Sagebrush now occupies an estimated 651,316 square kilometers (km^2; 160.1 million acres) over portions of 14 western States (fig. A1). Where sagebrush occurs, it shapes the community ecology of other plants and influences wildlife diversity and abundance.

Sagebrush occurs as a dominant or codominant shrub in many plant communities in the western United States, often interspersed with other plant communities (for example, desert, grassland, mountain shrub, deciduous and coniferous forests, and alpine systems). Despite these mixed patterns of sagebrush distribution across the landscape, the terms “sagebrush biome” and “sagebrush ecosystem” are useful to describe the extent of this community type.

Overview of Sagebrush Taxonomy

Criteria for identifying sagebrush species have changed over the past century, as have the number of sagebrush taxa recognized. The following summary is taken from Shultz’s 2009 monograph and 2012 field guide describing the *Artemisia* subgenus *Tridentatae*. Within this subgenus, there are 13 species of sagebrush, 10 of those within the section *Tridentatae* (Shultz, 2009; table A1), which comprise the sagebrush biome. The big sagebrush (*A. tridentata* ssp.) subspecies are closely related, and natural hybridization among them is common (Beetle, 1960; McArthur and others, 1979). Several taxa originated through hybridization and polyploidy (McArthur and Sanderson, 1999a, b). Some sagebrush hybridization is inconsequential (Beetle, 1977; Shultz, 2009), but taxa can form stable hybrid zones along ecotones. These zones may harbor populations that can expand into new habitats (McArthur and Sanderson, 1999b) such as dry lakebeds (Winward and McArthur, 1995; McArthur and Sanderson, 1999b) and abandoned croplands (Garrison and others, 2013). Polyploid taxa, such as Wyoming big sagebrush (*A. t.* ssp. *wyomingensis*) are usually more drought-tolerant than diploid taxa, such as basin big sagebrush (*A. t.* ssp. *tridentata*) and mountain big sagebrush (*A. t.* ssp. *vaseyana*).

Overview of Sagebrush Ecology

Sagebrush Shrublands and Sagebrush-Steppe

The sagebrush biome includes sagebrush semidesert shrublands and sagebrush-steppe ecosystems. They differ in vegetation structure, floristic composition, site productivity, and geographic distribution, which in turn affect which wildlife species use them. Undisturbed sagebrush-steppe plant communities have an equal or greater proportion of native herbaceous understory than shrubs. Sagebrush cover ranges from 10 to 50 percent. Sagebrush species and subspecies include mountain big sagebrush, snowfield big sagebrush, (*A. t. spiciformis*), silver sagebrush (*A. cana*), threetip sagebrush (*A. tripartita*), Wyoming big sagebrush, basin big sagebrush, black sagebrush (*A. nova*), and low sagebrush (*A. arbuscula*). Other shrubs, such as antelope bitterbrush (*Purshia tridentata*), snowberry (*Symphoricarpos* spp.), serviceberry (*Amelanchier* spp.), currant (*Ribes* spp.), and rabbitbrush (*Chrysothamnus* spp., *Ericameria* spp.,

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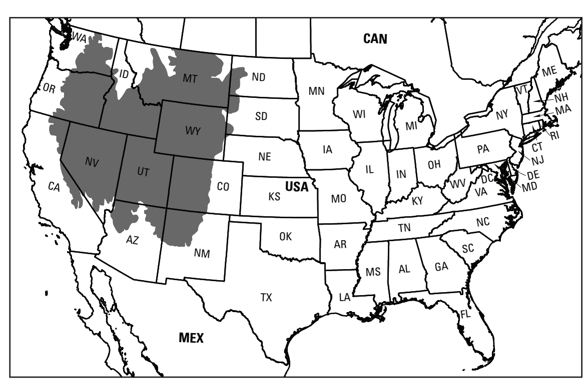

Figure A1. Extent of big sagebrush (*Artemisia tridentata*) and related sagebrushes (*Artemisia* subgenus *Tridentatae*) in the western United States (Jeffries and others, 2019).

Table A1. Species and subspecies of *Artemisia*, subgenus *Tridentatae*, section *Tridentatae,* which comprise the sagebrush biome (follows Shultz, 2009).

Common name	Genus/species/subspecies
Low or little sagebrush	*Artemisia arbuscula* ssp. *arbuscula*
Alkali sagebrush	*Artemisia arbuscula* ssp. *longiloba*
Hot springs sagebrush	*Artemisia arbuscula* ssp. *thermopola*
Bigelow sagebrush	*Artemisia bigelovii*
Black sagebrush	*Artemisia nova*
Pygmy sage	*Artemisia pygmaea*
Stiff sagebrush	*Artemisia rigida*
California silver sagebrush	*Artemisia cana* ssp. *bolanderi*
Plains silver sagebrush	*Artemisia cana* ssp. *cana*
Mountain silver sagebrush	*Artemisia cana* ssp. *viscidula*
Timberline sagebrush	*Artemisia rothrockii*
Snowfield sagebrush	*Artemisia spiciformis*
Basin big sagebrush	*Artemisia tridentata* ssp. *tridentata*
Mountain big sagebrush	*Artemisia tridentata* ssp. *vaseyana*
Wyoming big sagebrush	*Artemisia tridentata* ssp. *wyomingensis*
Parish or Mohave sagebrush	*Artemisia tridentata* ssp. *parishii*
Wyoming three-tip sagebrush	*Artemisia tripartita* ssp. *rupicola*
Three-tip sagebrush	*Artemisia tripartita* ssp. *tripartita*

Lorandersonia spp.) are typically present with variable cover. Sagebrush-steppe ecosystems occur in the northern Great Basin, Columbia Plateau, northern Great Plains, the Rocky Mountains, and at higher elevations in the southern Great Basin and the Colorado Plateau.

Sagebrush shrublands have a much lower proportion of graminoids and forbs to shrubs, often with a very sparse herbaceous layer. Sagebrush cover ranges from 10 to 40 percent. Sagebrush species include Wyoming and basin big sagebrush and all the dwarf sagebrush taxa. Along ecotones with salt desert shrublands, greasewood (*Sarcobatus vermiculatus*), shadscale (*Atriplex confertifolia*), fourwing saltbush (*Atriplex canescens*), spiny hopsage (*Grayia spinosa*), and winterfat (*Krascheninnikovia lanata*) are common. Sagebrush shrublands are dominant in the southern Great Basin, Colorado Plateau, and Wyoming Basin. They also occur on drier shallow and rocky soils in the northern Great Basin, Columbia Plateau, northern Great Plains, and the Rocky Moutains.

Sagebrush-steppe ecosystems were historically the most abundant vegetation type in the semidesert vegetation of North America (West, 1983). Much current sagebrush shrubland was once sagebrush-steppe as the herbaceous layer has been depleted under past management practices. In practice, it can be difficult to distinguish degraded sagebrush-steppe from sagebrush shrubland.

History of the Sagebrush Biome

The sagebrush biome is a working landscape, hosting a variety of land uses such as grazing, transmission line corridors, mining, and oil and gas development. Current ownership patterns across the sagebrush biome and the amount and condition of sagebrush habitat are artifacts of past policy and practices (Knick and Rotenberry, 2000; Morris and others, 2011). Contemporary management and conservation strategies for sagebrush ecosystems will need to consider this legacy of land use, ownership, and past management practices locally and across landscapes.

Settlement Through the 1930s

The geographic extent of the sagebrush biome prior to settlement is uncertain, but it is clear that the area occupied by sagebrush has declined since European settlement owing to urban and agricultural use and conversion to other vegetation types, such as pinyon-juniper woodlands or annual grasslands (Miller and others, 2011). Using 2006 LANDFIRE maps, Miller and others (2011) estimated that 55 percent of the area delineated as potentially dominated by sagebrush prior to settlement, based on mapping conducted by Küchler, was occupied by sagebrush (many known sagebrush areas, including all sagebrush habitats in eastern portions of Montana and Wyoming, were not mapped in the Küchler habitat types). This same analysis (Miller and others, 2011) estimated that sagebrush occupied 59 percent of the original extent of Küchler's Sagebrush Steppe type, 46 percent of the Great Basin sagebrush type, and 59 percent of the wheatgrass-needlegrass shrub steppe type. The amount of sagebrush loss ranged from 34 percent in Wyoming to 76.3 percent in Washington (Miller and others, 2011).

Following government land acquisitions, including the Louisiana Purchase of 1803 and the Oregon Treaty of 1846, the sagebrush biome was under Federal ownership. Subsequent public land policies designed to convert these lands to private ownership resulted in a mosaic of land ownership and land uses in sagebrush areas in the West (see review by Knick, 2011). Dozens of Federal legislative acts from 1785 through the mid-1900s (summarized in Knick, 2011) granted lands to States to support schools; to encourage homesteading, agricultural conversion, irrigation, and mining; and to transfer land to State or private entities, including railroads. Grazing by domestic livestock was unrestricted on Federal lands until a series of legislative acts between 1891 and 1934 placed restrictions on, initially U.S. Department of Agriculture (USDA) Forest Service (Forest Service) and U.S. Department of the Interior (DOI; now the Bureau of Land Management [BLM]) lands (Knick, 2011). The passage of the Taylor Grazing Act of 1934 (43 U.S.C. 315) introduced grazing districts, a permit system to limit numbers of livestock, and grazing management to reduce grazing impacts.

From the 1850s through the 1930s, the plant communities of the biome were heavily altered by excessive livestock grazing and repeated early spring grazing (Sampson, 1914; Clapp, 1936; Stoddart and others, 1938; Ellison, 1960; Miller and others, 1994; Miller and Eddleman, 2001; Crawford and others, 2004). By the 1930s, grazing capacity across the sagebrush-steppe was likely 60 to 90 percent less than presettlement conditions (McArdle and others, 1936; Stoddart and others, 1938). The reduction of grazing capacity was from reduced herbaceous growth as well as a change in herbaceous species, including the reduction and loss of native species and the introduction of many nonnative species.

Declines in herbaceous cover were followed by increases in sagebrush cover (McArdle and others, 1936; Stoddart and others, 1938; Ellison, 1960; Branson, 1985; Miller and others, 1994). This was because of release of sagebrush seedlings from herbaceous competition and decreased size and frequency of wildfires owing to the loss of herbaceous fuels. In many areas, these changes were accompanied by soil loss. The increase in sagebrush density and cover, along with soil loss and compaction, limited seedling establishment of native graminoids and forbs (McArdle and others, 1936; Shantz and Piemeisel, 1940). In response to the dust bowl and passage of the Taylor Grazing Act of 1934 (43 U.S.C. 315), a comprehensive range condition assessment was conducted for 295 million ha (728 million acres) of grazing land (including all habitat types) in the western United States, and the results were reported to Congress (U.S. Department of Agriculture, Forest Service, 1936). Presentation of their findings begins with the following:

> “There is perhaps no darker chapter nor greater tragedy in the history of land occupancy and use in the United States than the story of the western range. * * * The major finding of this report * * * is range depletion so nearly universal under all conditions of climate, topography, and ownership that the exceptions serve only to prove the rule.” (Clapp, 1936).

The report goes on to say,

> “Widespread, continuous, and exhaustive use of the forage has changed the whole character of the virgin range.” (McArdle and others, 1936).

The report states that forage productivity, or grazing capacity, of the presettlement rangelands was reduced by more than half. The greatest concern was the shift in species composition of the grasses and forbs:

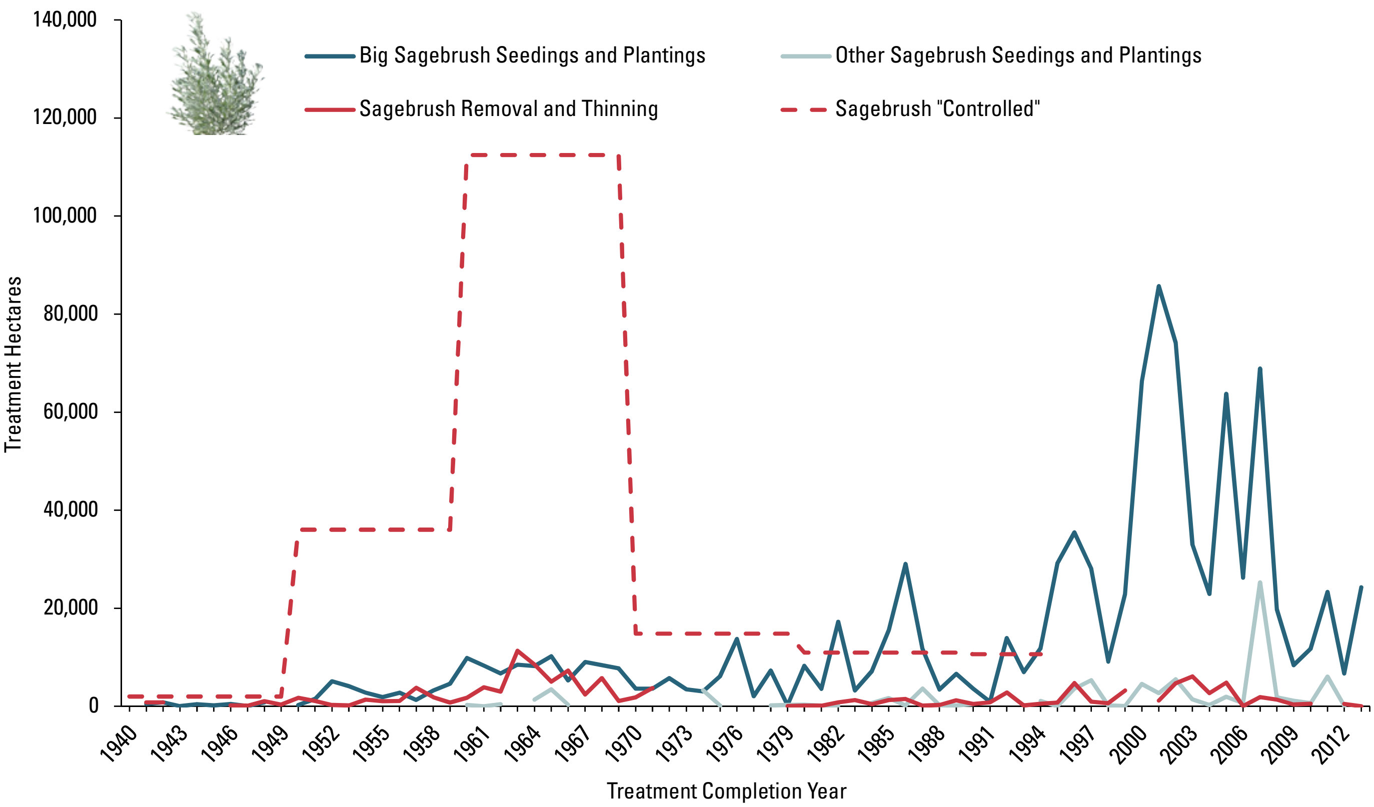

Figure A2. Hectares of big sagebrush (*Artemisia tridentata*) seedings and plantings, other sagebrush (*Artemisia* spp.) seedings and plantings, and sagebrush removal and thinning treatments on lands administered by the U.S. Department of the Interior, Bureau of Land Management (BLM; Pilliod and others, 2020b). Hectares of sagebrush control treatments on BLM lands annually from 1940 to 1994 averaged across total hectares treated per decade (Miller and Eddleman, 2001). These data represent the minimum hectares treated on BLM lands because many records have been lost or are incomplete.

"By far the most significant departure from virgin range conditions is the change in plant cover. * * * the plant cover in every type is depleted to an alarming degree. Many valuable forage species have disappeared entirely. Palatable plants are being replaced by unpalatable ones. Worthless and obnoxious weeds from foreign countries are invading every type" (McArdle and others, 1936). "*Only by restoring the vegetation as nearly as possible to its original composition and vigor can the productivity and stabilization of the soil and vegetation again be obtained*" (emphasis added; Stoddart and others, 1938).

1940s–1990s

The emphasis on homesteading of Federal lands and transfer to private ownership began to change in the mid-1900s to sustained use under Federal ownership (Dombeck and others, 2003). Management for multiple uses, including resource extraction, outdoor recreation, and habitat conservation for fish and wildlife began in 1960 on national forests and in 1964 on DOI lands (Bean and Rowland, 1997). The National Environmental Policy Act of 1969 (NEPA; 42 U.S.C. 4321 et seq.) required that potential environmental impacts of any activity or land use that could affect the environment be evaluated prior to approval. The Federal Land Policy and Management Act of 1976 (FLPMA; 43 U.S.C. 1701–1785) directed Federal lands be retained under Federal ownership, be managed for multiple uses and sustained use, be based on an inventory of natural resources, and follow a public planning process.

The management goal of restoration was replaced with the goal of increasing forage for livestock and reducing soil erosion. Attempts to eliminate sagebrush in an effort to cultivate or increase grass for livestock forage production were common. Initially, sagebrush was removed through mechanical means, with herbicides such as 2,4-D becoming more prevalent after the mid-1940s (Baker and others, 1976; Miller and others, 1994; Knick, 2011). Estimated minimum area of sagebrush treated on BLM lands was 18,000 km^2 (4.45 million acres) between 1940 and 1994 (Miller and Eddleman, 2001). Sagebrush removal peaked in the 1950s and 1960s (Miller and Eddleman, 2001; Pilliod and others 2017b; fig. A2). Spraying with 2,4-D eliminated sagebrush and also reduced or eliminated native forb species and antelope

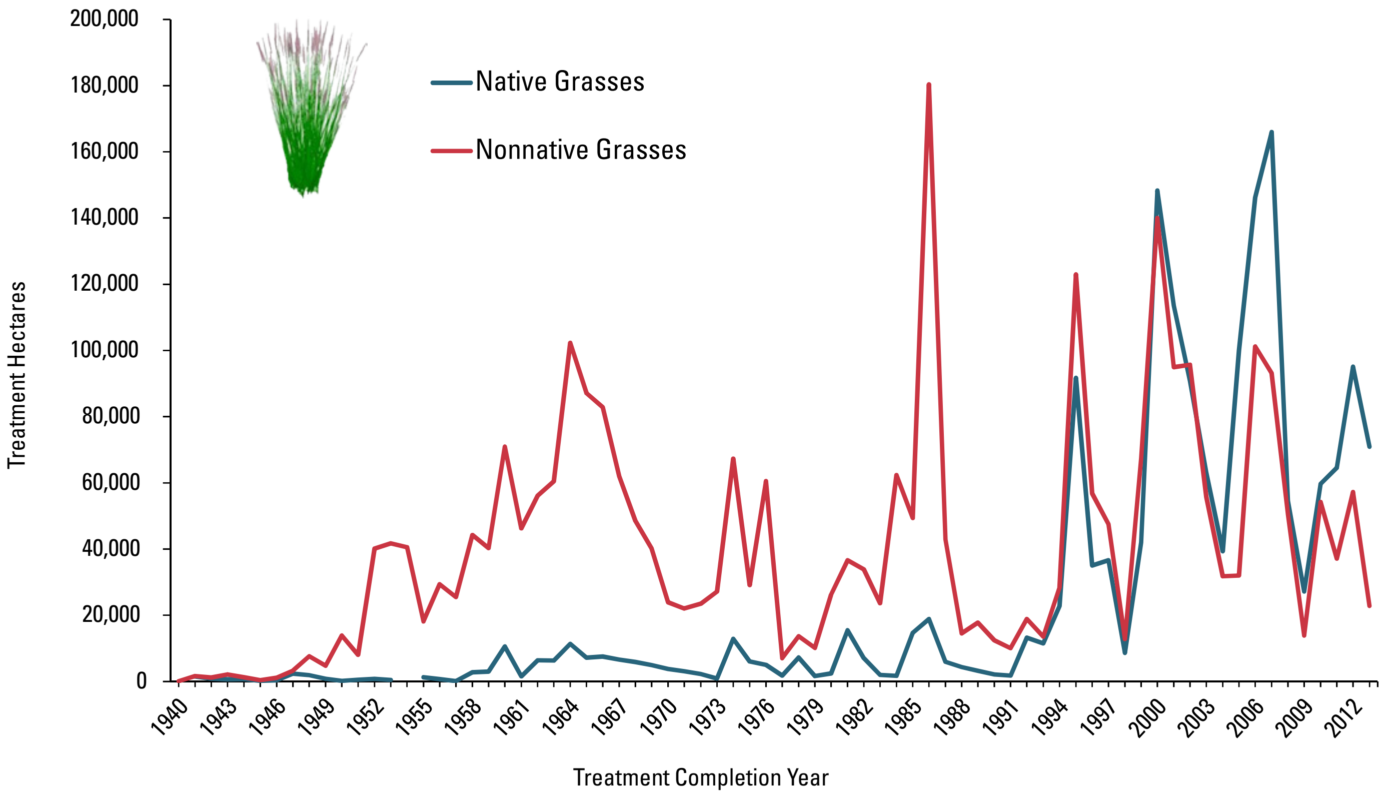

Figure A3. Hectares of native and nonnative grasses planted on U.S. Department of the Interior, Bureau of Land Management (BLM) lands by year (Pilliod and others, 2020b). These data represent the minimum hectares treated on BLM lands because many records have been lost or are incomplete.

bitterbrush, a key wildlife browse species. A newly introduced Eurasian species, crested wheatgrass (*Agropyron cristatum*), was initially used to stabilize soils in abandoned cropland in the 1930s (Young, 1994). Crested and other nonnative perennial wheatgrasses were used extensively following World War II to reseed areas after sagebrush removal because native species were difficult to seed, success rates were low, and seed sources were limited (fig. A3; Young, 1994). Seeding nonnative species with the goal of increasing forage for livestock did not restore the original species diversity or composition of the sagebrush plant communities, and often resulted in a monoculture of nonnative grasses that were less desirable to livestock, and thus, lightly or rarely grazed.

2000–2020—Sagebrush Management Becomes Sage-Grouse Management

Current objectives for sagebrush treatments typically involve thinning of high density or decadent sagebrush to promote grass and forb growth while maintaining some sagebrush canopy cover as habitat for sagebrush-obligate species. Native grass species are beginning to be favored for restoration, but nonnative species are still seeded at relatively high rates, particularly following larger fires, because of their availability (fig. A3; see also chap. R, fig. R1, this volume; Pilliod and others, 2017b; Copeland and others, 2018).

Concern over declining populations and a potential listing of sage-grouse under the Endangered Species Act of 1973 (ESA; 16 U.S.C. 1531 et seq.) began to influence management and conservation of sagebrush on public and private lands in the early 2000s. This management emphasis followed nine petitions between 1998 and 2005 to list various populations or presumed subspecies of greater sage-grouse and Gunnison sage-grouse under the ESA.

Recent sagebrush management has focused on sage-grouse. Funding and support for mechanical or chemical sagebrush elimination treatments have been minimized, and stipulations designed to protect sage-grouse breeding and nesting habitats from development or other disturbances were updated and waived less often. Restoration efforts on burned or otherwise degraded habitats increased (Pilliod and others, 2017b). Research efforts on sage-grouse and attempts to better understand and map seasonal habitats were accelerated. The Western Association of Fish and Wildlife Agencies (WAFWA), with U.S. Fish and Wildlife Service (FWS) funding produced “The Conservation Assessment of Greater Sage-Grouse and Sagebrush Habitats” in 2004 (Connelly and others, 2004), followed by the “Greater Sage-Grouse Comprehensive Conservation Strategy” in 2006 (Stiver and others, 2006). The USDA Natural Resource Conservation Service began the Sage Grouse Initiative in 2010, a cost-share program to incentivize landowners and public land management agencies to adopt positive conservation measures for sage-grouse.

Conservation and planning efforts by Federal and State agencies, private landowners, and others resulted in a not warranted ESA finding for greater sage-grouse in 2015 (U.S. Department of the Interior, 2015c). These and additional conservation efforts, including Federal land management documents, form the basis for sagebrush management today. The nature and likely effectiveness of the current State, Federal, and private conservation efforts for sage-grouse at conserving sagebrush and sagebrush-obligate, -dependent and -associated wildlife is reviewed in chapter Q, this volume.

Sagebrush Benefits, Sagebrush Wildlife

The sagebrush biome supports people and communities in the West (reviewed in chap. B, this volume) and provides habitat for more than 350 species of plants and animals considered species of conservation concern, including 63 vertebrates (Wisdom and others, 2005). The relationship of animals to sagebrush habitats varies widely, from those with an absolute dependence on sagebrush, such as greater sage-grouse or pygmy rabbits (*Brachylagus idahoensis*), to other species that have large ranges across multiple habitat types, including sagebrush, such as gray flycatcher (*Empidonax wrightii*; table A2). The following definitions were developed for use in this strategy to clarify relationships of species to sagebrush for management purposes.

Sagebrush obligate.—Complete dependence on sagebrush or associated sagebrush plant community to meet one or more seasonal habitat requirements. If sagebrush is lost, habitat functionality is lost for obligate species. Sage-grouse and sagebrush sparrow (*Artemisiospiza nevadensis*) are examples of sagebrush-obligate species.

Sagebrush near-obligate.—Breeding distribution almost entirely within sagebrush communities and highest densities achieved within sagebrush communities. Pronghorn (*Antilocapra americana*) and green-tailed towhee (*Pipilo chlorurus*) are examples of sagebrush near-obligate species.

Sagebrush dependent.—Species with seasonal distributions in a variety of habitats including sagebrush, but there is a strong overlap with the distribution of sagebrush. Where ranges overlap, sagebrush provides an important seasonal habitat. Mule deer (*Odocoileus hemionus*) are an example of a sagebrush-dependent species.

Sagebrush associated.—Species that occur in other communities such as grassland, woodland, or shrublands but may also breed and forage or meet other habitat requirements in sagebrush.

Table A2. Sagebrush (*Artemisia* spp.) obligate; near-obligate; and dependent birds, mammals, reptiles, and amphibians.

Birds	Mammals	Reptiles and amphibians
Brewer's sparrow (*Spizella breweri*)[1]	Pygmy rabbit (*Brachylagus idahoensis*)[1]	Sagebrush lizard (*Sceloporus graciosus*)[2]
Greater sage-grouse (*Centrocercus urophasianus*)[1]	Sagebrush vole (*Lemmiscus curtatus*)[1]	Desert nightsnake (*Hypsiglena chlorophaea*)[3]
Gunnison sage-grouse (*Centrocercus minimus*)[1]	Great Basin pocket mouse (*Perognathus mollipilosus*)[2]	Great Basin spadefoot (*Spea intermontana*)[3]
Sage thrasher (*Oreoscoptes montanus*)[1]	Merriam's shrew (*Sorex merriami*)[2]	Greater short-horned lizard (*Phrynosoma hernandesi*)[3]
Sagebrush sparrow (*Artemisiospiza nevadensis*)[1]	Preble's shrew (*Sorex preblei*)[2]	Pygmy short-horned lizard (*Phrynosoma douglasii*)[3]
Gray flycatcher (*Empidonax wrightii*)[2]	Pronghorn (*Antilocapra americana*)[2]	
Green-tailed towhee (*Pipilo chlorurus*)[2]	Wyoming ground squirrel (*Urocitellus elegans*)[2]	
Pinyon jay (*Gymnorhinus cyanocephalus*)[4]	Dark kangaroo mouse (*Microdipodops megacephalus*)[3]	
	Mule deer (*Odocoileus hemionus*)[3]	
	Ord's kangaroo rat (*Dipodomys ordii*)[3]	
	Southern Idaho ground squirrel (*Urocitellus endemicus*)[3]	
	White-tailed prairie dog (*Cynomys leucurus*)[3]	
	Wyoming pocket gopher (*Thomomys clusius*)[4]	
	Black-tailed jackrabbit (*Lepus californicus*)[3]	
Total = 8	**Total = 14**	**Total = 5**

[1]Sagebrush obligate.

[2]Sagebrush near-obligate.

[3]Sagebrush dependent.

[4]Sagebrush associated, conservation concern, and likely to be affected.

Chapter B. Human Dimensions of Sagebrush

By Drew E. Bennett[1] and Julie Suhr Pierce[2]

Executive Summary

Humans have, and continue to derive, multiple benefits from sagebrush (*Artemisia* spp.) ecosystems. The concept of ecosystem services provides a framework to discuss these benefits—both market and nonmarket—to human beings. Beneficiaries of these services include not only those now living within or visiting the sagebrush biome, such as farmers, ranchers, and recreationists but also people in distant towns and cities as well as future generations. The sagebrush biome provides water filtration, improved timing of water flows, flood attenuation, irrigation water supply, enhanced connectivity between subsurface and surface water flows, and more. Intact sagebrush ecosystems reduce wildfire return intervals; they also provide forage for both livestock and wildlife and host many species of wildlife, including animals we hunt, as well as sensitive, threatened, and endangered species. Healthy sagebrush ecosystems sequester carbon, which can be enhanced through conservation efforts on public lands as well as on privately owned rangelands. Ranchers have participated in voluntary conservation projects aimed at protecting and restoring sagebrush landscapes as well as providing habitat to sage-grouse (*Centrocercus* spp.). In some sagebrush ecosystems, there are considerable mineral deposits. Solid minerals mining contributes multiple types of economic benefits and costs to local and regional economies, and at the same time, it presents challenges to public land managers who must balance the interests of public stakeholders and those of mining companies. Recreationists benefit from the sagebrush biome through hunting, fishing, bicycling, hiking, wildlife viewing, bird watching, horseback and off-highway vehicle (OHV) riding, and multiple other activities. Indigenous peoples and more recent arrivals enjoy cultural benefits from sagebrush landscapes. Threats to sagebrush ecosystems such as invasive species, wildfire, and many others also directly threaten the ecosystem services that people derive from sagebrush and pose an indirect threat because of the potential listing of species under the Endangered Species Act of 1973 (16 U.S.C. 1531 et seq.).

As people work together to protect and restore the sagebrush biome, the ecosystem services provided by the biome can be secured for both current and future generations.

Introduction

Human beings have lived in and depended on the sagebrush (*Artemisia* spp.) biome for thousands of years. Sagebrush ecosystems have been a backdrop for—and have contributed to—human social, spiritual, and cultural well-being. People have visited and used these vast landscapes for recreation, contemplation, religious practices, artistic and literary work, and cultural activities. Humans have benefitted socially and economically from sagebrush landscapes in multiple ways.

Historical Relationship of Humans to Sagebrush

Big sagebrush (*Artemisia tridentata*) has 216 documented traditional uses by Native Americans, including medicinal, ceremonial, building (fiber), and clothing materials (Moerman, 1998). Tribes that use big sagebrush are numerous and include the Paiute, Shoshoni, and Washoe (Moerman, 1998). The Northern Cheyenne included sagebrush in their Sun Dance ceremony (Liberty, 1967). Hunter-gatherer Tribes hunting American bison (*Bison bison*), deer (*Odocoileus* spp.), elk (*Cervus canadensis*), and pronghorn (*Antilocapra americana*) were indirectly dependent on sagebrush for sustenance as big sagebrush was, and remains, an important seasonal component of diets for these species. This indirect dependence on sagebrush continued when, as early as 400 years ago, some Native American Tribes transitioned into using domestic sheep (*Ovis aries*) as a source of fiber, food, and income, replacing hunting to some extent in their way of life. Sagebrush would have been an important food source for domestic sheep, particularly during winter. Inputs derived from domestic sheep became important to artistic and textile activities that were culturally significant while also serving as potential income streams in Tribal communities. Since colonization of the West by descendants of European immigrants, the relationship that humans have with sagebrush landscapes has become more complex, and the understanding of these relationships has expanded.

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[1]University of Wyoming.

[2]U.S. Department of the Interior, Bureau of Land Management.

Ecosystem Services

Ecosystem services provide a context within which to discuss the various types of benefits derived by people from the sagebrush biome. Ecosystem services, or the benefits that people receive from nature, are commonly classified within four major categories: regulating, provisioning, cultural, and supporting (Millennium Ecosystem Assessment, 2005). Sagebrush environments provide numerous tangible ecosystem services, such as food products from livestock production; hunting; other recreational opportunities; and the provision of water for municipal, industrial, and irrigation uses (table B1). Some ecosystem services come from direct use on public lands. Other ecosystem services are provided by privately owned sagebrush landscapes. Still other ecosystem services, such as clean water, flood control, irrigation, and other benefits are enjoyed outside of the sagebrush landscape. Management of each ownership type presents its own set of challenges, and what occurs on one can affect the others.

Multiple regulating ecosystem services are provided by sagebrush landscapes. Hydrologic services provided by sagebrush ecosystems influence water quantity and quality and the timing and location of flows. Intact sagebrush environments can slow the flow of surface runoff from precipitation, increasing the infiltration into soils and groundwater that supply localized drinking water and irrigation supplies, as well as maintaining subsurface flows to surface water (Brauman and others, 2007). This serves to attenuate flows, reducing the probability of flooding downstream as infiltration increases subsurface flows. These subsurface-to-surface connections are critical for maintaining late season flows that moderate downstream flooding events, support fisheries, and provide recreational opportunities. Large portions of sagebrush range are also within the source watersheds that supply most of the drinking water to several major cities, including Las Vegas, Nevada; Los Angeles, California; and San Diego, California (McDonald and Shemie, 2014). Maintaining healthy sagebrush ecosystems helps support the filtration, storage, and soil stabilization services and the ongoing provision of municipal and industrial water to over 6.5 million people.

In addition to municipal and industrial water, sagebrush ecosystems are an upstream source of irrigation water for agriculture in the western United States. Some of the sagebrush landscape has been converted to cropland. While making up a relatively small part of the western United States economy, agriculture plays an important role in the economies and cultures of rural communities in the West. Farms across a large portion of the United States benefit indirectly from healthy sagebrush ecosystems, as runoff from the sagebrush biome feeds into the greater Arkansas, Colorado, Columbia, and Missouri River systems, providing water to both farmers and ranchers in parts of the Northwest, Great Plains, Rocky Mountain and Great Basin, and Mississippi River Basin. Farms and ranches benefit from supporting services provided by intact sagebrush landscapes. Production of native grasses, forbs, and other forage for livestock and wildlife, as well as nutrient cycling services, support and augment provisioning values generated by the sagebrush biome.

Sagebrush also provides climate stabilization services through carbon sequestration, primarily in the form of soil carbon. Broadly speaking, public and private rangelands in the United States, of which sagebrush areas make up a large percentage, sequester a significant volume of carbon and hold the potential to mitigate carbon emissions through restoration of degraded rangelands or conversion of marginal cropland to native vegetation (Follett and others, 2001; Olander and others, 2012). Despite this mitigation potential, participation of sagebrush rangelands in current voluntary or potential future compliant carbon markets (for example, cap-and-trade programs) presents significant economic obstacles because of the low volume of additional carbon that can be sequestered per unit of land area relative to the transaction costs involved (Joyce and others, 2013). However, a survey of 495 ranchers in Utah showed that while only 10 percent of ranchers perceived climate mitigation as a benefit of adopting practices to sequester rangeland carbon, 39 percent perceived these practices as promoting environmentally sound land management (Ma and Coppock, 2012). Additionally, a majority (71 percent) of those surveyed stated that they were open to engaging in carbon sequestering practices (Ma and Coppock, 2012). Ranchers are unlikely to receive significant financial incentives to adopt carbon sequestration practices within the sagebrush biome in the foreseeable future. They

Table B1. Examples of services provided by the sagebrush (*Artemisia* spp.) ecosystem.

Ecosystem service categories	Examples
Regulating	Water purification, water infiltration and flood attenuation, carbon sequestration, wildfire resistance
Provisioning	Products from livestock (for example, beef, lamb, leather, wool); water for municipal, industrial, and irrigation use; mineral extraction; food from hunting wildlife
Cultural	Recreational opportunities such as cycling, hiking, hunting, and wildlife viewing (for example, sage-grouse [*Centrocercus* spp.] leks); sense of place; spiritual benefits
Supporting	Production of grasses, nutrient cycling

may contribute to climate mitigation by their willingness to adopt carbon sequestering practices perceived as beneficial to sound range management. Another regulating service provided by healthy sagebrush ecosystems is resistance to wildfires. Maintaining a thriving, wildfire-resistant sagebrush landscape provides both economic benefits and benefits that are more difficult to quantify, such as human life and safety. When an ecosystem transitions from sagebrush dominated to cheatgrass (*Bromus tectorum*)-dominated landscapes, fire-return intervals shorten from as long as a 100 or more years to as little as 3 to 5 years (chap. I, this volume). These shortened return intervals result in higher firefighting costs, greater losses in terms of wildlife mortality, burned buildings and infrastructure, and loss of livestock and potentially human lives. Fuels management in sagebrush landscapes is not always seen by stakeholders in a positive light. Sense of attachment to specific views and landscapes comes into conflict with fuel break creation and other landscape-altering management activities.

The ranching community benefits from multiple sagebrush ecosystem services, including supporting services that provide forage for livestock, which in turn allow for provisioning services, such as beef production. In a survey of 645 ranchers in the western United States (not exclusive to sagebrush areas), participants identified the cultural values of maintaining their families' traditions and values as the most important reason for owning a ranch, followed by passing on the ranch and ranch lifestyle to future generations (Tanaka and Maczko, 2017). Provisioning services, like providing food and fiber, and economic reasons, like obtaining a good return on investment, were ranked below cultural services. Additionally, the survey found that the vast majority of ranchers did not allow access or charge a fee for recreation on their lands, suggesting that there is opportunity to capitalize on these tangible cultural services (Tanaka and Maczko, 2017). These findings demonstrate the importance of multiple ecosystem services to the ranching community that go beyond the economically valuable provisioning services.

Managing with ecosystem services in mind can also support ranchers in addressing business risks and opportunities (Toombs and others, 2011) within operational, regulatory, reputational, market and product, and financing categories (Hansen and others, 2018). For instance, ranchers managing for ecosystem services that prevent soil erosion can reduce regulatory risks while potentially being able to access cost-share programs that provide operational opportunities. Similarly, managing in ways that provide habitat for wildlife species of concern can help manage reputational risks by demonstrating that livestock production can be compatible with wildlife conservation (Toombs and others, 2011). Although opportunities remain few to directly monetize ecosystem services on sagebrush rangelands, the potential to do so through habitat mitigation, carbon sequestration, and niche marketing of meat products may grow in the future (Goldstein and others, 2011). Providing venues for compensatory mitigation of impacts to greater sage-grouse (*Centrocercus urophasianus*) may be a particular area of growth in the future as Colorado, Idaho, Montana, Nevada, Oregon, Utah, and Wyoming have, or are developing, State-sponsored mitigation programs for this purpose.

Development of successful and more benign grazing systems in sagebrush ecosystems following European settlement required decades of trial and error and changes in management regimes. In addition to sagebrush landscapes, typical western forage systems often also include higher elevation forests, irrigated pastures, base property corrals, and feedlots. Domestic livestock joined wildlife in the sagebrush biome niche, although the relative benefits that livestock provides to land health depend on how herds and flocks are managed.

Livestock operations provide important socioeconomic benefits for rural western communities. Through multiplier effects, ranch expenditures ripple through local and regional economies. Ranching provides employment, labor income, value added, and output benefits as sales of output and purchases of inputs push revenues outward from ranches into their communities. In rural towns where summer tourism is key to economic activity, ranch purchases of supplies of all types—fuel, hard goods, food, and so on—can provide a stabilizing stream of revenue to small businesses in the off-season. From a cultural standpoint, ranching has come to be associated with traditional life in the West. In many rural communities, the so-called cowboy way of life is appreciated by both residents and visitors and is an integral part of the cultural aspects of ranching communities. Having a chance to see or work alongside actively working ranch hands has become a key feature of tourism across the western United States. A prime example is found in southern Utah, where cowboy culture is highly valued.

> "The cowboy culture that once was widespread within the American West, but that is no longer as prevalent as it once was in some of the West's more urbanized places, is still a central part of life within the Grand Staircase-Escalante National Monument area. It is important to many long-time residents of the region to preserve and celebrate the traditional cowboy lifestyle and the skills, knowledge, and cultural arts that are connected with it." (Bureau of Land Management, 2015, p. 8).

Because the long-term success of ranching in the sagebrush biome depends on landscape health and adequate forage for livestock, ranchers have often been key partners in conservation efforts in the West over recent decades. Some ranchers have demonstrated a collective commitment to making their operations more compatible with protection of wildlife, particularly to avoid listings under the ESA. Separate efforts by ranchers to protect sage-grouse by marking fences and engaging in other conservation practices have demonstrated that ranchers can actively engage in conservation activities beyond their livestock management actions. According to the U.S. Department of Agriculture, Natural Resources Conservation Service (NRCS),

> "Ranchers are part of a range-wide collaborative effort to voluntarily aid the sage-grouse and the sagebrush landscape, an effort credited with enabling the U.S. Fish and Wildlife Service to determine in 2015 that protections under the ESA were not needed for the species. The NRCS is working with nearly 1,500 landowners in 11 western States to improve habitat for sage-grouse while also improving ranching operations." (Natural Resources Conservation Service, 2019a).

The willingness of government agencies and private landowners to cooperate in restoring sagebrush ecosystem services is evidence of how highly those services are valued.

Not all activities that improve sage-grouse habitat are universally viewed as positive. Some stakeholders are upset by herbicide treatments, chaining projects, and other management practices that alter the landscape and affect nontargeted species (Shindler and others, 2011; Gordon and others, 2014). Conifer expansion is seen by some as a natural process that should not be controlled; it is seen by others as a cause of ecosystem decline that needs to be set back in order to protect fragile systems from the loss of native grasses and forbs.

Landowners, government agencies, and universities have participated in studying sagebrush ecosystems and determining what conservation practices would provide the greatest benefits. Participants in local and regional sage-grouse working groups contributed to planning efforts and facilitated funding of many sagebrush habitat protection and restoration projects across the sagebrush biome (Belton and Jackson-Smith, 2010; Belton and others, 2017). In Wyoming, a statewide plan for sage-grouse was adopted in 2002, after which two statewide and eight local citizen working groups were established to provide stakeholder input in the development and implementation of conservation efforts. Multiple stakeholders have perceived these efforts to be successful and attributed success to sound science, maintaining funding, and long-term commitment from working group participants (Christiansen and Belton, 2017).

Recreational activities within the sagebrush biome are generally classified as "cultural" ecosystem services. These include activities such as cycling, hiking, hunting, fishing, wildlife viewing, tourist activities, off-highway vehicle (OHV) use, shed-antler hunting, dark-sky viewing, and so on. Each of these activities in turn can generate a variety of benefits, including economic activity, social connections, and personal well-being. Yet these recreational uses can also conflict with each other or other management goals and need to be managed holistically to minimize user conflicts and damage to resources (Switalski, 2018).

Measuring the value of recreational and other uses of sagebrush landscapes can be challenging. Economists regularly quantify the value of ecosystem goods and services in dollar terms. Methods vary and can consider many factors including

- market prices based on the activities and choices made by actual people, and the contribution of environmental or ecosystem services to the price of other goods and or services (for example, better viewsheds can increase the price of homes on otherwise comparable properties),
- the amount of money people either are willing to spend or actually spend on visits to a particular place,
- surveys asking how much people are willing to pay to obtain an ecosystem good or service or how much they would have to be compensated in dollars in exchange for giving up an ecosystem good or service,
- the cost to provide a specific ecosystem good or service by means of a human-built method,
- estimating the value of a healthy ecosystem by identifying the cost of treatment for ecological damages where treatment or mitigation is required,
- assessing the value of something as a minimum equal to the value of the next best use, and
- estimating the value of an ecosystem when a damaging activity is either proposed or has already occurred.

One of the greatest challenges associated with managing public lands is the variety of opinions and values of stakeholders (Brunson and Shindler, 2004). Local knowledge regarding all aspects of the sagebrush biome can inform land management decisions. Fostering trust between land management agencies and local citizens is an ongoing process that requires engagement, communication, and good-faith decision making. Shindler and others (2011) surveyed public opinions and perceptions of sagebrush management and found low levels of trust in land management agencies responsible for implementing management actions. When the process does not go well, local citizens sometimes resort to appealing to political figures in order to influence (or stop) decisions that do not match their own values. Gordon and others (2014) suggest that in order to gain public support for management practices, stakeholders should focus on building trust and establishing relationships with communities rather than simply providing more or better information.

Mining presents an example regarding the degree of conflict and disagreement between stakeholders in planning and decision making. The planning processes that currently exist allow members of the public in general to participate via public meetings and comment opportunities. However, corporations—whose economic well-being depends on the success of their proposals, from a claim initially being filed to retirement of a mine and reclamation—have legal standing that often supersedes the interests and values of people who enjoy the nonconsumptive ecosystem services provided by the same geographic locations. Balancing the economic, social, and cultural interests of the full range of stakeholders can be daunting to public land managers. In addition to providing revenue to the owners, mining projects provide direct, indirect, and induced economic benefits to the local and regional economies. These include jobs, labor income, demand for products and services sold by local and regional wholesalers and retailers, and secondary economic activity caused by recirculation of these dollars spent as employees, wholesalers, and retailers spend their incomes. This process is repeated throughout the economy, causing ripples of economic activity through the region. Fiscal inputs such as royalty payment receipts, tax receipts, and, on occasion, civic infrastructure sponsorship are additional economic benefits that are provided by mining companies to the benefit of communities nearby.

As these benefits are generated, there are community costs associated with hardrock mining. These include increased demand for social and civic services; pressure on housing and other markets, which can drive up prices and make the cost of living difficult to afford for nonmining families; increased traffic and wear on local and regional transportation systems; and degraded environmental quality. Perhaps the greatest cost is the loss of all other ecosystem services provided by the landscape where the mine is developed. Unfortunately, obtaining the ecosystem services provided by sagebrush landscapes often forces managers to choose between mutually exclusive values. Some of the benefits provided by hardrock mining are nationally strategic and contribute to the well-being of citizens across the United States. Local and regional sacrifices in sagebrush ecosystems provide essential minerals for economic activities nationwide.

Two of the most difficult values to measure for sagebrush ecosystems are sense of place and spiritual benefits. Because these are personal in nature and not often acknowledged during management or project planning, it is difficult to assess these benefits. One way to evaluate these types of ecosystem services, and their benefits to people, is through studying their appearance in literature and fine art. Many authors and artists have expressed an appreciation for the beauty, quiet, and sense of the sublime provided by vast sagebrush landscapes. Native works of art and cowboy art have frequently been set in sagebrush ecosystems. Nonfiction books, particularly those focused on westward migration during the 19th century, as well as novels, such as “The Green Grass of Wyoming” by Mary O’Hara, have often used a sagebrush landscape backdrop. To a degree, these artistic works capture the place and spiritual value of these landscapes to the public.

Overall, the social and economic benefits provided by a thriving sagebrush biome are extensive and highly valued. Sometimes these benefits are explicit and easy to identify; other times they are subtle and not easy to pinpoint. At times, the benefits are enjoyed directly on the landscape itself, and in other cases, the benefits are enjoyed far downstream from where they are generated. Recognizing these benefits and managing landscapes to protect them into the future will be challenging but worthwhile to current and future generations.

Threats

Chapters J–P of this volume describe the numerous threats to the sagebrush biome, which, individually and cumulatively over time, affect the ability of individuals and communities to benefit from sagebrush ecosystem services. These chapters also summarize the social costs of these threats, when this information is known. Impacts to people can be direct, such as reduced weight gains in calves when invasive plants or overabundant free-roaming equids degrade rangelands, or when structures are destroyed by wildland fire. They can be indirect, for instance when grazing is deferred when agencies attempt to restore burned landscapes after fire. Economic impacts can extend beyond curtailment of ecosystem services to include extractive industries when land uses are restricted because of threats to species of conservation concern such as greater sage-grouse or Gunnison sage-grouse (*C. minimus*). The cost of failure to conserve the sagebrush biome could be severe. The total economic impact in the State of Wyoming—if greater sage-grouse were listed as threatened—was estimated to be a loss of $1.5–5.4 billion (2–6 percent of total economic output for the State), total employment could decrease by 8,019 to 24,307 jobs, total labor earnings could decrease by $500.6 million to $1.5 billion, and State/local government revenue could decrease by $96.1 million to $287.5 million per year (Stoellinger and Taylor, 2016).

Chapter C. Sagebrush Birds

By T. Luke George,[1] Aaron M. Sidder,[2] and Brittany J. Woiderski[1]

Executive Summary

Five passerine bird species dependent upon or strongly associated with sagebrush (*Artemisia* spp.) at landscape scales have been identified as likely to be impacted by sagebrush management activities: Brewer's sparrow (*Spizella breweri*), sagebrush sparrow (*Artemisiospiza nevadensis*), sage thrasher (*Oreoscoptes montanus*), gray flycatcher (*Empidonax wrightii*), and green-tailed towhee (*Pipilo chlorurus*). Additionally, one corvid species, pinyon jay (*Gymnorhinus cyanocephalus*), is also affected by sagebrush management decisions, although it is not considered a sagebrush obligate. Population vulnerability for each of these bird species was assessed considering size of breeding range, population size, population trends, and threats to breeding areas. While the response of sagebrush-dependent or -associated bird species to specific threats varies based on their ecology and behavior, any activity that eliminates, degrades, or reduces connectivity among sagebrush patches can reduce population size and occupancy of an area. Many sagebrush-associated bird species respond negatively to disturbances such as infrastructure development; type conversion to grasslands as a result of fire; or removal of sagebrush by mechanical thinning, mowing, or herbicide application, as it reduces sagebrush cover that provides nesting and foraging habitat. Actions that replace sagebrush or shrubs with grasses have a consistently negative effect, especially when introduced Eurasian species, such as crested wheatgrass (*Agropyron cristatum*), are seeded. The pinyon jay is a nonmigratory corvid that nests and feeds primarily in pinyon (*Pinus* spp.)-juniper (*Juniperus* spp.) forests, but its habitat often overlaps sagebrush communities. This species is of concern to managers of sagebrush landscapes because populations are declining, and management efforts to remove conifer can have negative consequences for pinyon jays.

[1]Bird Conservancy of the Rockies.

[2]Red Beard Science LLC.

Introduction

The Brewer's sparrow (*Spizella breweri*), sagebrush sparrow (*Artemisiospiza nevadensis*), sage thrasher (*Oreoscoptes montanus*), gray flycatcher (*Empidonax wrightii*), and green-tailed towhee (*Pipilo chlorurus*) are dependent upon or strongly associated with sagebrush (*Artemisia* spp.) at landscape scales. These five passerine species plus one additional species, the pinyon jay (*Gymnorhinus cyanocephalus*), are likely to be affected by sagebrush management decisions.

Although many other avian species occur in sagebrush for at least a part of their annual life cycles, this chapter focuses on these 6 species because of their dependence on sagebrush and potential to be impacted by sagebrush management actions.

In the following discussions, species range data from BirdLife International and NatureServe (2015) were used to estimate the proportion of each species' breeding range that occurs within sagebrush (derived from LANDFIRE 1.4.0 Existing Vegetation Cover; U.S. Geological Survey, 2014b).

Habitat Selection and Dependency on Sagebrush

Brewer's Sparrow

The Brewer's sparrow range overlaps about 98 percent of the sagebrush extent (table C1; fig. C1). However, because it uses habitats other than sagebrush, only about 46 percent of its entire range occurs within sagebrush (table C1). Thus, while sagebrush management may impact this species, it is likely to have less impact on its overall population than other sagebrush obligates. Where Brewer's sparrows occur within sagebrush dominated habitats, they require large patches of sagebrush with dense shrub cover (Knick and Rotenberry, 1995; Reinkensmeyer and others, 2007) and exist in highest densities in areas that include some taller patches of sagebrush (Chalfoun and Martin, 2007). They nest in shrubs, particularly sagebrush (Rich, 1980; Reynolds, 1981) and prefer high densities of suitable nest shrubs for territories and nest sites (Chalfoun and Martin, 2007, 2009). Brewer's sparrows generally are not abundant at poor condition sites with less than 25 percent cover in climax vegetation. This suggests they are associated with stands approaching climax conditions and do not thrive in seral communities (Vander Haegen and others, 2000).

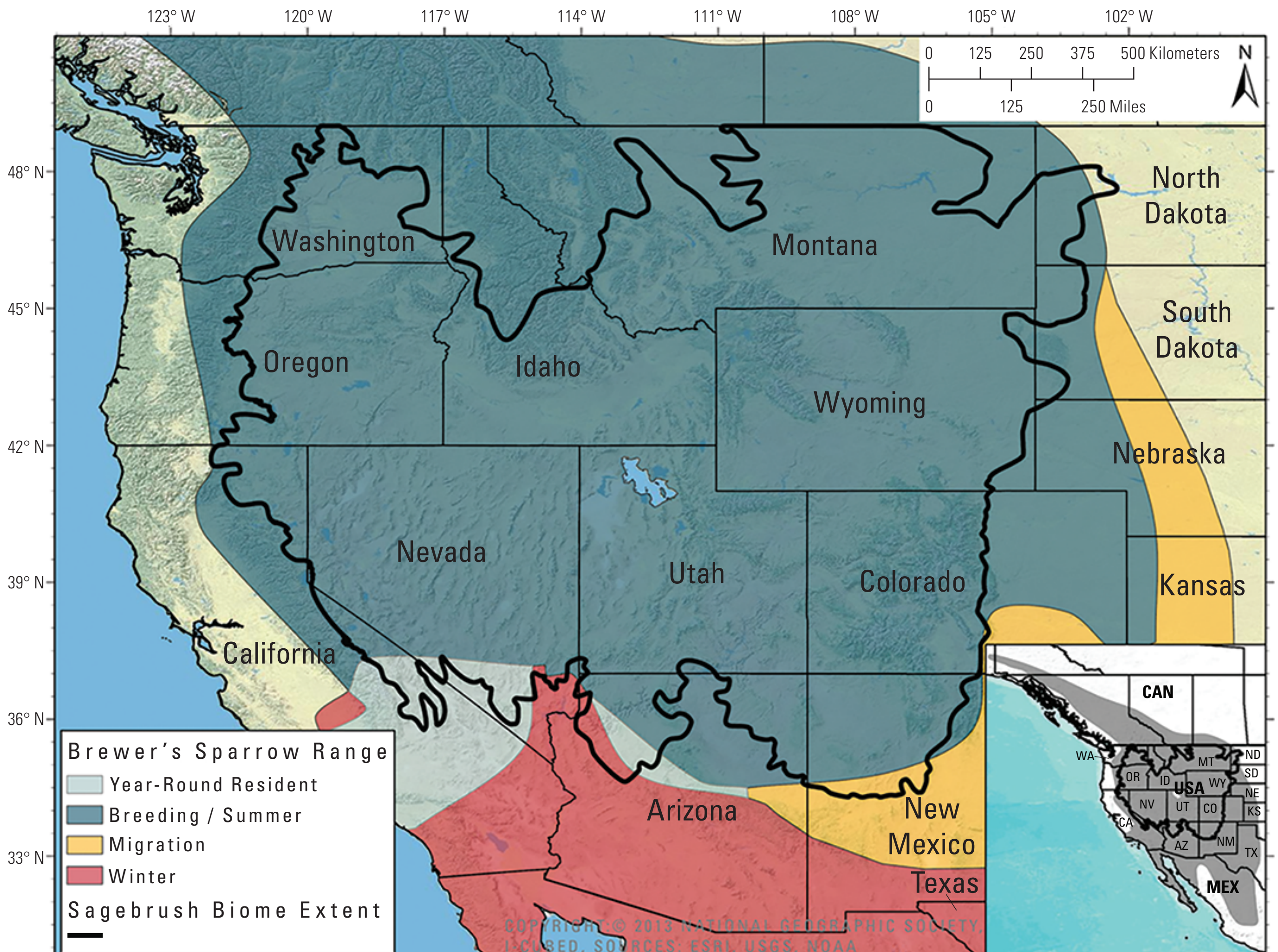

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Figure C1. Range of the Brewer's sparrow (*Spizella breweri*) in the sagebrush (*Artemisia* spp.) biome. Data were obtained from BirdLife International and NatureServe (2015).

Sagebrush Sparrow

The sagebrush sparrow is a migratory sagebrush-obligate songbird that breeds in sagebrush shrublands. The sagebrush sparrow breeding range covers more than 1 million square kilometers (km^2, 386,100 square miles [mi^2]; Rosenberg and others, 2016; fig. C2; table C1), and occupies about 76 percent of the sagebrush extent (table C1). Sagebrush sparrows generally thrive in habitats with relatively tall big sagebrush (*A. tridentata*) cover and high horizontal heterogeneity (Wiens and Rotenberry, 1981). The species prefers large and contiguous areas of tall and dense sagebrush and generally nests in the interior of sagebrush stands (Hansley and Beauvais, 2004), placing nests at the base of sagebrush shrubs or within the sagebrush canopy (Rich, 1980; Reynolds, 1981; Petersen and Best, 1985). As grass cover increases, sagebrush sparrow abundance decreases (Rotenberry and Wiens, 1980). Management actions that reduce shrub cover and increase grass cover or bare ground, such as mine reclamation (Krementz and Sauer, 1982), generally reduce sagebrush sparrow densities.

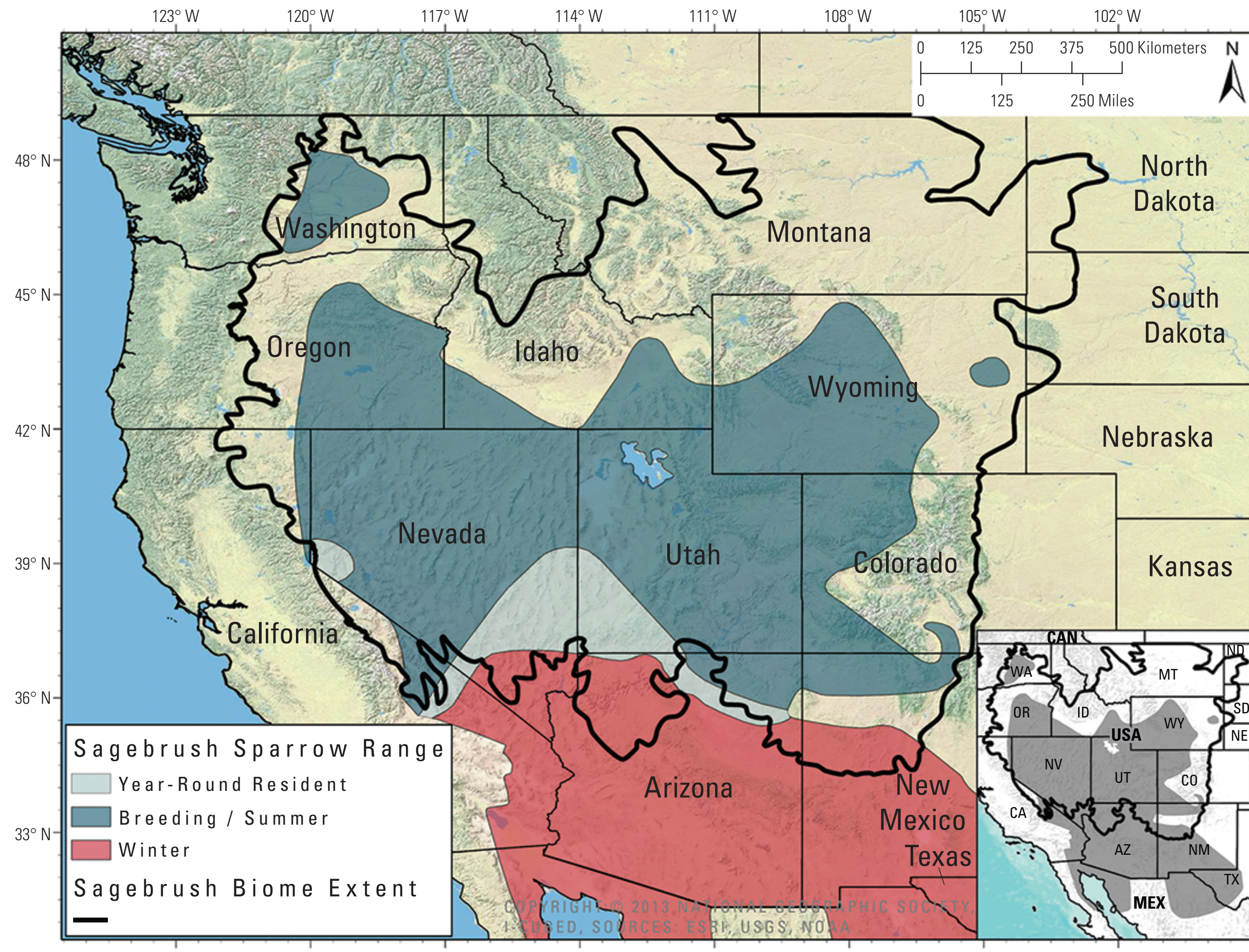

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Figure C2. Range of the sagebrush sparrow (*Artemisiospiza nevadensis*) in the sagebrush (*Artemisia* spp.) biome. Data were obtained from BirdLife International and NatureServe (2015).

Sage Thrasher

A sagebrush obligate, the sage thrasher inhabits sagebrush shrublands, especially tall, mature stands of big sagebrush (Boyle and Reeder, 2005). The breeding range of the sage thrasher includes almost all (93 percent) of the sagebrush ecosystem (fig. C3), but it also inhabits other vegetation types, and therefore sagebrush only makes up 56 percent of its breeding range (table C1). Almost all nests are located within or under big sagebrush plants (Rich, 1980; Reynolds, 1981; Petersen and others, 1991). Multiple studies found that the replacement of sagebrush with grasses, whether through invasion by cheatgrass (*Bromus tectorum*) or seeding of introduced grasses such as crested wheatgrass (*Agropyron cristatum*) for reclamation and restoration, reduces sage thrasher densities (Reynolds and Trost, 1980; Krementz and Sauer, 1982; McAdoo and others, 1989; Brandt and Rickard, 1994).

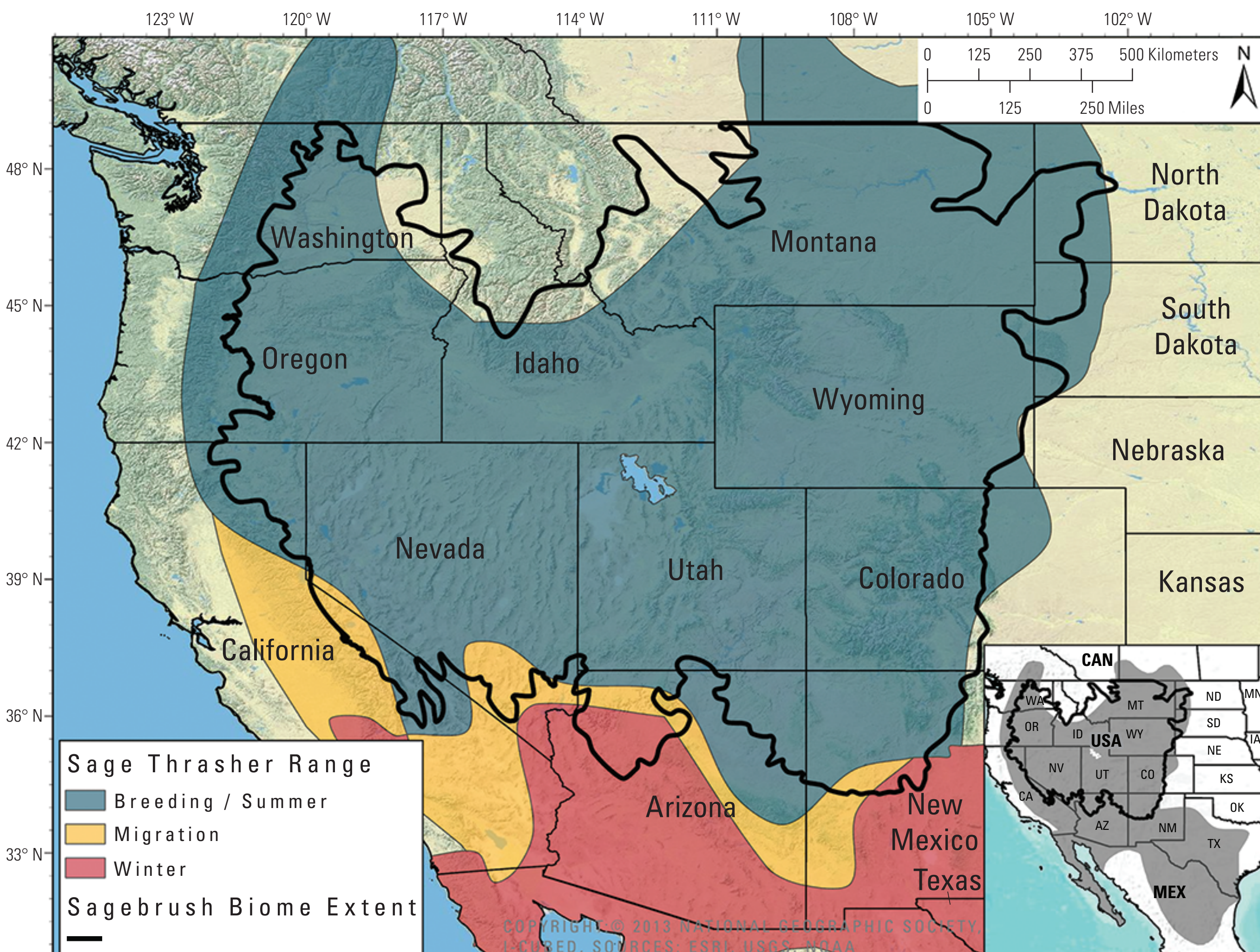

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Figure C3. Range of the sage thrasher (*Oreoscoptes montanus*) in the sagebrush (*Artemisia* spp.) biome. Data were obtained from BirdLife International and NatureServe (2015).

Gray Flycatcher

The gray flycatcher is a migratory sagebrush near-obligate that breeds in sagebrush or mountain mahogany (*Cercocarpus* spp.) shrublands and pinyon (*Pinus* spp.)-juniper (*Juniperus* spp.) woodlands. The gray flycatcher's breeding range occupies the southern and western portion of the sagebrush range and overlaps 63 percent of the sagebrush ecosystem (BirdLife International and NatureServe, 2015; fig. C4; table C1). The gray flycatcher is closely tied to arid woodlands and shrublands, including pinyon-juniper with a sagebrush understory (Gillihan, 2006). Birds use sites that combine high overstory juniper cover, pinyon pine presence, some senescent trees, and an understory of seedlings and saplings (tall sagebrush and late-successional pinyon-juniper woodlands; Pavlacky and Anderson, 2001).

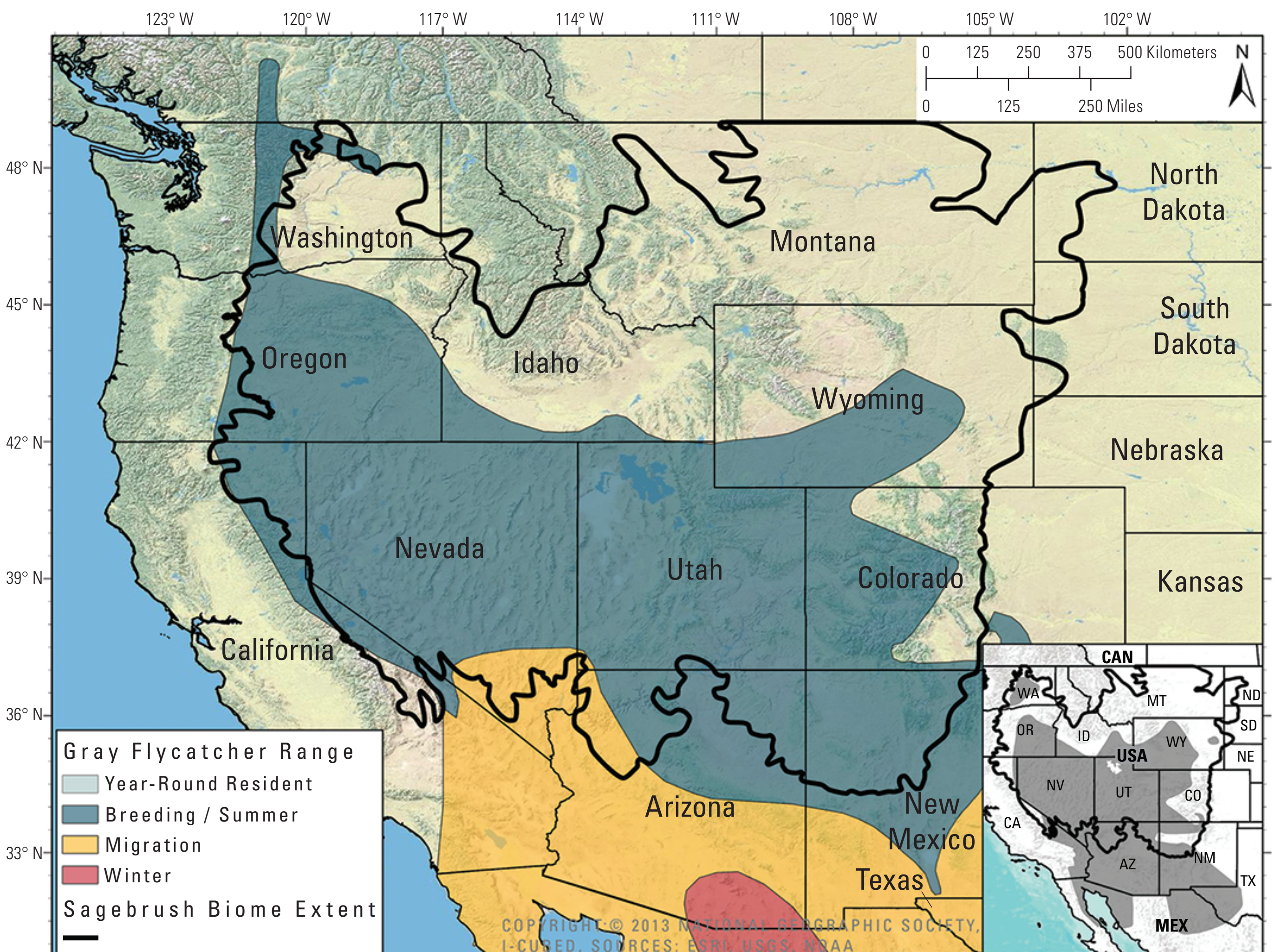

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Figure C4. Range of the gray flycatcher (*Empidonax wrightii*) in the sagebrush (*Artemisia* spp.) biome. Data were obtained from BirdLife International and NatureServe (2015).

Green-Tailed Towhee

The green-tailed towhee is a migratory sagebrush near-obligate that inhabits sagebrush shrublands, woodlands, and riparian areas. Green-tailed towhees inhabit a large (82 percent) proportion of the sagebrush ecosystem (fig. C5), but because they use other habitat types, sagebrush only constitutes 58 percent of their breeding range. This species prefers habitats with tall, dense shrubs and a diverse shrub community (Dobkin and Sauder, 2004), which provide cover for their bulky nests.

Pinyon Jay

The pinyon jay is a nonmigratory corvid that nests and feeds primarily in pinyon-juniper forests, but its habitat often overlaps sagebrush communities (fig. C6) where the two are adjacent. This species is of concern to managers of sagebrush landscapes because populations are declining, and management efforts to remove conifer can have negative consequences for pinyon jays (chap. M, this volume).

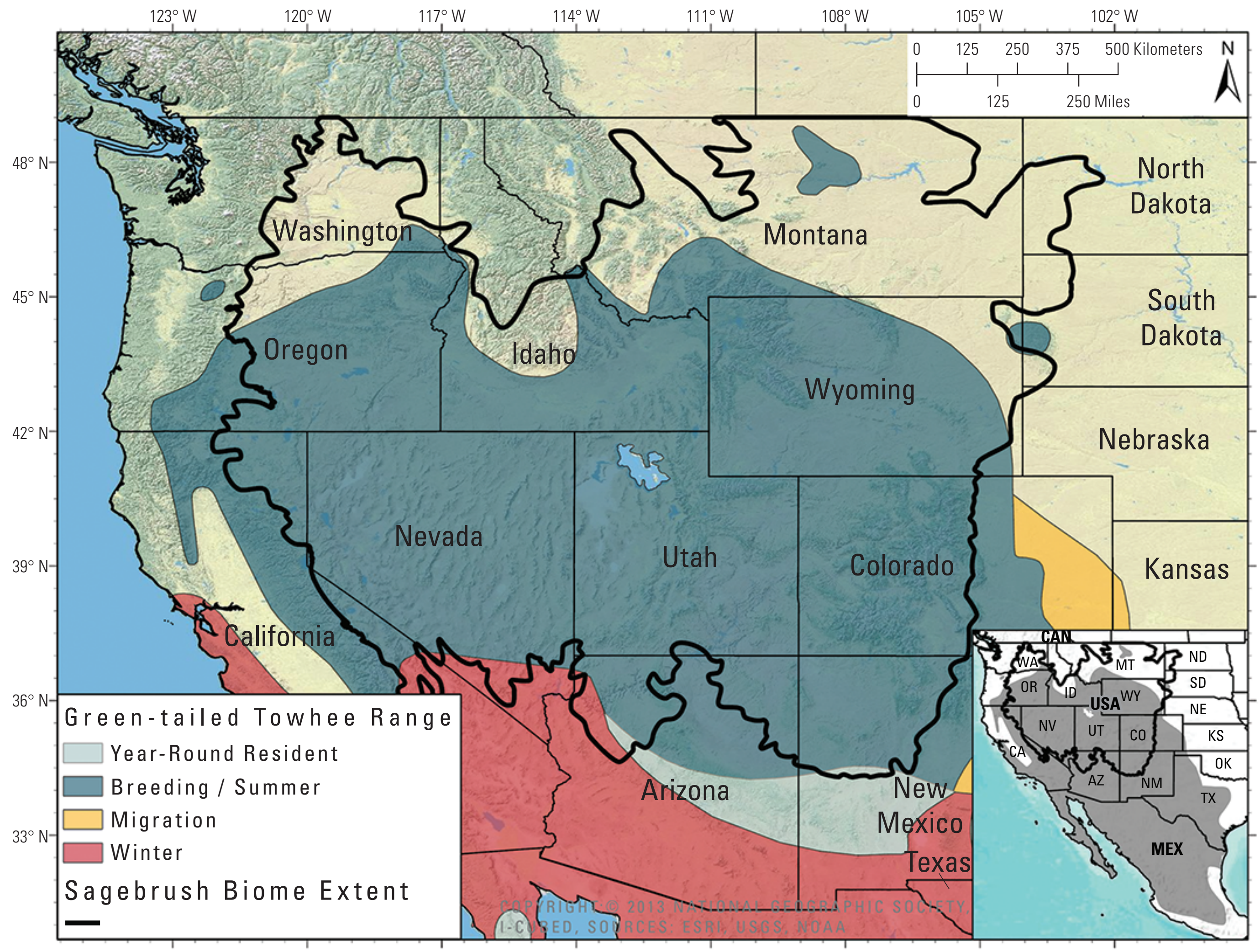

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Figure C5. Range of the green-tailed towhee (*Pipilo chlorurus*) in the sagebrush (*Artemisia* spp.) biome. Data were obtained from BirdLife International and NatureServe (2015).

Population Trends and Conservation Status

Population vulnerability for each of these six bird species was assessed considering size of breeding range, estimated population size, population trends, and threats to breeding areas. Population vulnerability information was obtained from Partners in Flight (PIF; Rosenberg and others, 2016), and trend data over the period 1966–2015 was obtained from the Breeding Bird Survey (BBS; Sauer and others, 2017). BBS trends were used because they provide the only regionwide, long-term trend data for most avian species in North America.

Of these six species, only pinyon jay is included on the PIF “Watch List,” which identifies species of highest conservation concern at the continental (rangewide) scale based on analyses of population size and trend, breeding and nonbreeding distribution, and threats. Pinyon jay is described as having population declines and moderate to high threats (Rosenberg and others, 2016). Over the period 1966–2015, three of the six species experienced significant declines in counts on BBS routes in the western United States: Brewer’s sparrow, sage thrasher, and pinyon jay, whereas gray flycatchers increased over this same time period (table C1).

With a global population of 16 million, Brewer’s sparrow is among the most numerous of the sagebrush songbirds, though its population has been declining across the western United

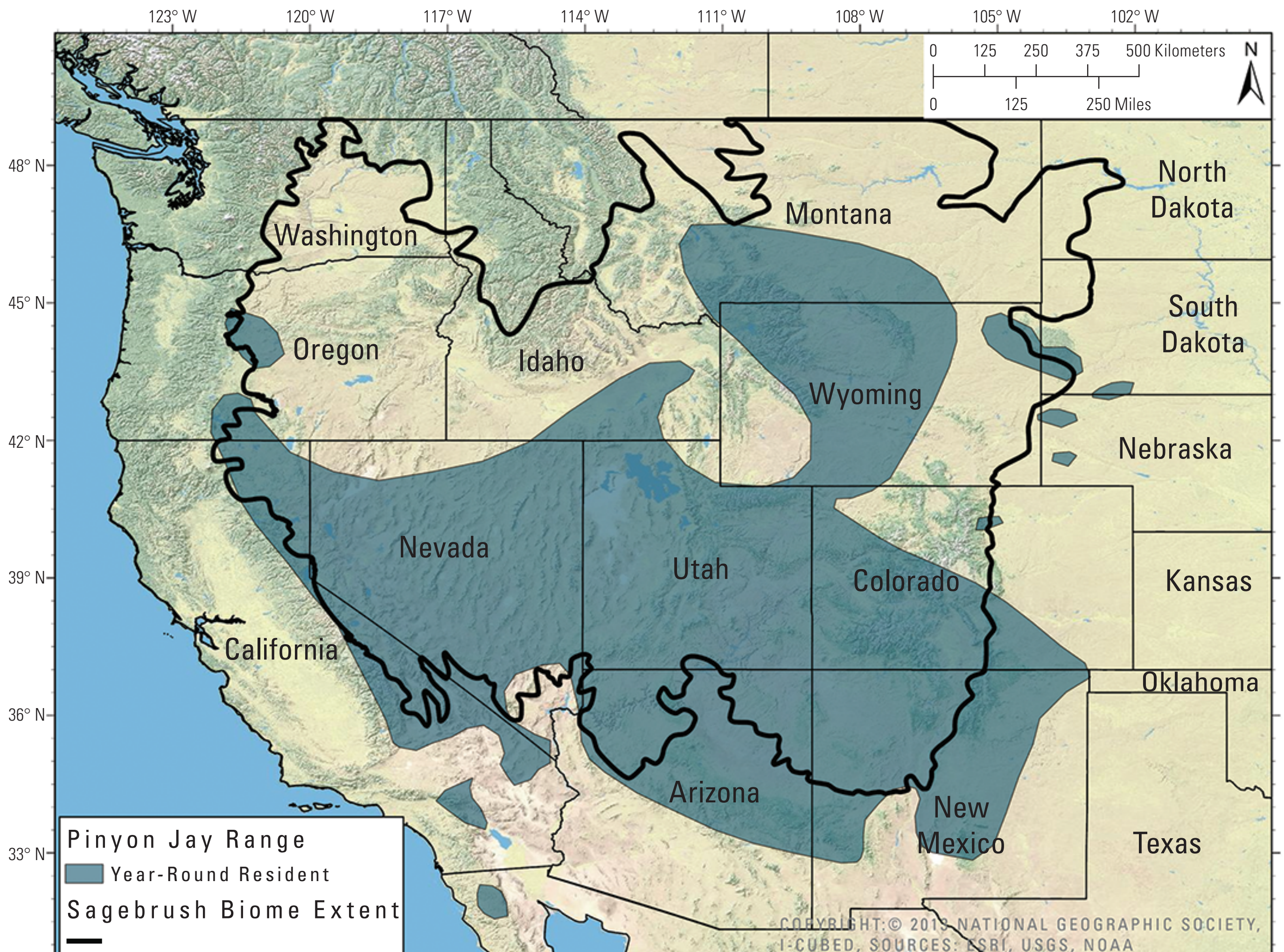

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Figure C6. Range of the pinyon jay (*Gymnorhinus cyanocephalus*) in the sagebrush (*Artemisia* spp.) biome. Data were obtained from BirdLife International and NatureServe (2015).

States (table C1; Sauer and others, 2017). The species is on five State's species of concern lists: Idaho, Montana, Nevada, South Dakota, and Wyoming. The Brewer's sparrow is currently declining most dramatically in shortgrass prairie and badlands and is experiencing population loss throughout its breeding range (Sauer and others, 2017). The species decline is most pronounced in Colorado, which has large areas of shortgrass prairie. Population decline is also pronounced in Oregon and California with a slightly lower, but significant, decline in Idaho (Sauer and others, 2017). The species has lost approximately 60 percent of its population since 1970 (Rosenberg and others, 2016). A slight to moderate decline in the future suitability of its breeding conditions was predicted by the PIF Landbird Conservation Plan (Rosenberg and others, 2016).

The sagebrush sparrow has an estimated global population of 4.7 million birds (table C1; Rosenberg and others, 2016) but is declining throughout its range (Note: the analysis includes data for Bell's sparrow [*Artemisiospiza belli*], which was split from the sagebrush sparrow in 2013; Sauer and others, 2017). The species is declining most rapidly in the Southern Rockies/Colorado Plateau region and in the northern Great Basin in southeastern Oregon and southern Idaho. Population decline is greatest in Idaho and Oregon where the species is estimated to be losing more than 4 percent of its population annually, although data deficiencies may be affecting accuracy of the trend estimate in Idaho (Sauer and others, 2017). A slight to moderate decline in the future suitability of its breeding conditions has been predicted (Rosenberg and others, 2016). The sagebrush sparrow is listed as a species of concern in three western States: Montana, Washington, and Wyoming.

The sage thrasher has an estimated population of 6.6 million individuals (Rosenberg and others, 2016). Sage thrashers have experienced large population declines, losing approximately 1.2 percent of their population annually since 1966 (table C1; Sauer and others, 2017). The species is listed on five State species of concern lists: Montana, Nevada, South Dakota, Washington, and Wyoming. The sage thrasher is declining throughout the western United States, but its losses are most pronounced in the Great Basin (Sauer and others, 2017). Sage thrashers are declining most rapidly in Utah (−1.9 percent/year) with smaller but significant declines in Nevada, Idaho and Oregon (−1.5, −1.4, and −1.4 percent/year, respectively; Sauer and others, 2017). Over the past 50 years, sage thrasher populations have declined by 44 percent, but the number of years until it loses half its current population is estimated at more than 50 years (PIF half life; Rosenberg and others, 2016). Breeding threats to the sage thrasher are described as slight to moderate (Rosenberg and others, 2016).

The population of gray flycatchers is estimated at around 3 million, and it is the only bird species reviewed in this chapter with a positive population trend since the 1960s (table C1; Rosenberg and others, 2016; Sauer and others, 2017). The gray

Table C1. Summary of conservation-related information for six sagebrush (*Artemisia* spp.)-associated bird species.

[Statistically significant Breeding Bird Survey (BBS) trend counts are indicated by *, with a 95-percent confidence interval that does not overlap 0. km^2, square kilometer; %, percent; mi^2, square mile; N/A, not applicable]

Species	Sagebrush dependence[1]	Global population[2]	Breeding range (km^2)[3]	Species range overlap sagebrush (%)[3]	Sagebrush range overlap species (%)[3]	BBS trend[4]
Brewer's sparrow (*Spizella breweri*)	Obligate	16,000,000	2,821,742 (1,089,481 mi^2)	46	98	−1.01*
Sagebrush sparrow[5] (*Artemisiospiza nevadensis*)	Obligate	4,700,000	1,087,895 (420,039 mi^2)	74	60	−0.11
Sage thrasher (*Oreoscoptes montanus*)	Obligate	6,600,000	1,752,289 (676,562 mi^2)	56	93	−1.2*
Gray flycatcher (*Empidonax wrightii*)	Near obligate	3,000,000	1,143,266 (441,417 mi^2)	63	64	2.43*
Green-tailed towhee (*Pipilo chlorurus*)	Near obligate	4,800,000	1,735,151 (669,946 mi^2)	58	82	−0.31
Pinyon jay (*Gymnorhinus cyanocephalus*)	Associated conservation concern	690,000	1,335,717 (515,723 mi^2)	N/A	N/A	−3.69*

[1]Sagebrush dependence was assessed by scientists at the Western Association of Fish and Wildlife Agencies meeting in June 2016.

[2]Estimates from 2016 Partners in Flight report (Rosenberg and others, 2016).

[3]Species range data from BirdLife International and NatureServe (2015); sagebrush cover derived from LANDFIRE (U.S. Geological Survey, 2014b) and cover data.

[4]Annual trend in counts on Breeding Bird Survey (BBS) routes in the western United States, 1966–2015 (Sauer and others, 2017).

[5]The sage sparrow was split into the Bell's sparrow (*Artemisiospiza belli*) and the sagebrush sparrow in 2013. Trend data are for both species combined.

flycatcher is not listed as a species of concern in any of the western States. BBS data for the gray flycatcher are deficient in many States and ecoregions, but its population appears to be increasing annually by 2.5 percent throughout the western United States., with even higher increases in California, Nevada, and Utah and within the Northern Rocky Mountain Bird Conservation Region (Sauer and others, 2017). These trends translate to a 185 percent population increase over the past 35 years (Rosenberg and others, 2016). A slight to moderate decline in the future suitability of breeding conditions for the gray flycatcher is predicted (Rosenberg and others, 2016).

The green-tailed towhee population is estimated at 4.8 million birds (Rosenberg and others, 2016). Over the past 40 years, the green-tailed towhee has experienced modest annual (−0.3 percent per year) population declines and is estimated to have lost 17 percent of its population during this period (Sauer and others, 2017; table C1). However, this trend is not consistent across the West. In Utah, for example, towhees have experienced a 3.3 percent annual increase since 1966, whereas Idaho's towhee population declined by 3.0 percent per year (Sauer and others, 2017). The Great Basin has experienced population declines that exceed the overall trend (−1.1 percent/year; Sauer and others, 2017). Threats to breeding range were identified as slight to moderate (Rosenberg and others, 2016). With its modest population declines, the green-tailed towhee's population half-life estimate is >50 years (Rosenberg and others, 2016).

The pinyon jay has declined at a rate of 4.3 percent/year, the largest rate of decline of birds reviewed in this chapter. Pinyon jays have declined 84 percent over the period 1966–2015 (Rosenberg and others, 2016). Pinyon jays are projected to lose half of their current population within 19 years if the current rate of decline continues, and, consequently, PIF listed the species on its "D" Yellow Watch List, indicating it is a species with declining populations and moderate to high threats. (Rosenberg and others, 2016). Large regional declines are occurring in many areas of the western United States but are particularly severe in the Great Basin Ecoregion. The pinyon jay is listed as a species of concern in Colorado, Idaho, Montana, and Nevada.

Threats

Response of sagebrush-dependent or -associated bird species to specific threats varies based on their ecology and behavior. In general, for the sagebrush-obligate and near-obligate passerines (Brewer's sparrow, sagebrush sparrow, sage thrasher, gray flycatcher, and green-tailed towhee), any activity that eliminates, degrades, or reduces connectivity among sagebrush patches can reduce population size and occupancy of an area. For further discussion of the impacts of these threats to these six bird species, see chapters in this volume on individual threats in "Part II. Change Agents in the Sagebrush Biome—Extent, Impacts, and Efforts to Address Them."

Management Considerations

Many sagebrush-associated bird species respond negatively to the loss of sagebrush from wildfire, mechanical thinning, mowing, or herbicide application as they reduce sagebrush cover that provides nesting and foraging habitat (Norvell and others, 2014; Rottler and others, 2015; Carlisle and others, 2018a). Mowing sagebrush in Wyoming eliminated use of those areas by Brewer's sparrows and sage thrashers (Carlisle and others, 2018a). Mechanical thinning in Utah increased Brewer's sparrow density at treated plots in the first year posttreatment, though density reverted to nearly the reference condition by the fourth year following treatment (Norvell and others, 2014). In a sagebrush/grassland habitat, Brewer's sparrows declined when herbicide application resulted in a total kill of all sagebrush, but not in partial-kill plots, which included alternate spray strips and partial removal of sagebrush (Best, 1972). In another study, Brewer's sparrow densities decreased by 67 percent after 1 year and by 99 percent after 3 years in herbicide-treated big sagebrush areas (Schroeder and Sturges, 1975). Brewer's sparrow appears adaptable to some sagebrush removal regardless of method, provided some dense sagebrush islands remain on the landscape. Actions that replace sagebrush or shrubs with grasses have a consistently negative effect, especially when introduced Eurasian species, such as crested wheatgrass, are seeded (Reynolds and Trost, 1980; Krementz and Sauer, 1982; McAdoo and others, 1989).

Both mechanical thinning and herbicide application, which reduced shrub cover, reduced sagebrush sparrow density, though effects may be slightly delayed by 1 or 2 years (Wiens and Rotenberry, 1985; Norvell and others, 2014). Sage thrasher density was reduced following sagebrush and shrub cover reduction from chaining or mechanical thinning (Castrale, 1982; Norvell and others, 2014).

Acknowledgments

We would like to thank Anna Chalfoun, U.S. Geological Survey, and Chris White, Bird Conservancy of the Rockies, for their edits and additions to this chapter.

Chapter D. Greater and Gunnison Sage-Grouse

By Thomas E. Remington[1]

Executive Summary

Greater and Gunnison sage-grouse (*Centrocercus urophasianus* and *C. minimus,* respectively) are iconic western species entirely dependent on sagebrush (*Artemisia* spp.) for food and cover. Both species co-evolved with sagebrush and have developed physiological and behavioral adaptations that allow them to feed exclusively on sagebrush leaves in the winter, but which also create a dependency on that diet. Sage-grouse can exhibit large seasonal and annual movements, which include migration between breeding and wintering areas for some populations. Gunnison sage-grouse are currently listed as a threatened species under the Endangered Species Act of 1973 (16 U.S.C. 1531 et seq.). Both the Bi-State distinct population segment of greater sage-grouse and greater sage-grouse rangewide were recently found not warranted for listing under the Endangered Species Act. Threats to sage-grouse are numerous and significant, including but not limited to invasive species, altered fire regimes, energy development, free-roaming equids, and a warming climate.

Introduction

Sage-grouse (*Centrocercus* spp.) only occur within the sagebrush (*Artemisia* spp.) ecosystem of the western United States and Canada. Greater sage-grouse (*C. urophasianus*) are distributed across 703,453 square kilometers (km^2 ; 271,604 square miles [mi^2]; U.S. Department of the Interior, 2015c) in portions of 11 States (California, Colorado, Idaho, Montana, Nevada, North Dakota, Oregon, South Dakota, Utah, Washington and Wyoming; fig. D1) and two Canadian provinces (Alberta and Saskatchewan). Gunnison sage-grouse (*C. minimus*) occurs in the Gunnison Basin of Colorado and five other small, isolated sagebrush areas in southwestern Colorado, plus one area that straddles southwestern Colorado and southeastern Utah (fig. D1). Approximately 2 percent of the total range of greater sage-grouse occurs in Canada, with the remainder in the United States (Knick, 2011). The Bi-State population of greater sage-grouse occurs along the California-Nevada border (fig. D1) and was designated a distinct population segment (DPS) by the U.S. Fish and Wildlife Service (FWS) based on genetic differentiation and distance to other sage-grouse populations. Unless otherwise specified, the term “sage-grouse” will refer to both species and the Bi-State DPS.

[1]Western Association of Fish and Wildlife Agencies.

Habitat Selection and Dependency on Sagebrush

Sage-grouse depend on large areas of contiguous sagebrush to meet all seasonal habitat requirements (Connelly and others, 2011a, b; Wisdom and others, 2011) and are considered sagebrush-obligate species. Sage-grouse occur across a diversity of sagebrush plant communities across the sagebrush biome. Consequently, sage-grouse distribution is strongly correlated with the distribution of sagebrush (fig. D1). Sage-grouse use a variety of sagebrush species including Wyoming big sagebrush (*A. tridentata wyomingensis*), mountain big sagebrush (*A. t. vaseyana*), basin big sagebrush (*A. t. tridentata*), black sagebrush (*A. nova*), fringed sagebrush (*A. frigida*), silver sagebrush (*A. cana*), and low sagebrush (*A. arbuscula*; Miller and others, 2011).

During the breeding season, male sage-grouse gather to perform courtship displays on areas called leks. Females visit leks for mating then travel to nesting areas characterized by sagebrush with an understory of native grasses and forbs that provide cover, an insect prey base, and herbaceous forage for prelaying and nesting females (Connelly and others, 2000a; Connelly and others, 2004). Females typically move from 1.3 to 5.1 kilometers (km; 0.8 to 3.2 miles [mi]) from leks to nest (Connelly and others, 2011a, b; Dahlgren and others, 2016a), although the juxtaposition of habitats, amount of disturbance, and the extent of habitat fragmentation may influence the distance that nests are located from leks (Connelly and others, 2011b, and references therein). Most nests are located under sagebrush plants, and nests under sagebrush tend to be successful at higher rates than nests under other substrates (but see Wallestad and Pyrah, 1974; Connelly and others, 1991; Gregg and others, 1994; Sveum and others, 1998).

Females rear their broods near the nest site for the first 2 to 3 weeks following hatching. Forbs and insects are essential nutritional components for chicks (Connelly and others, 2004). Chick growth and survival is enhanced when early brood-rearing habitat provides adequate cover adjacent to areas with abundant forbs and insects (Connelly and others, 2004; Thompson and others, 2006; Huwer and others, 2008; Casazza and others, 2011). Sage-grouse gradually move from sagebrush uplands to more mesic (wet) areas during the late brood-rearing period (Peterson, 1970), as herbaceous vegetation dries and senesces (Connelly and others, 2000a). Summer use areas can include sagebrush habitats as well as riparian areas, wet meadows, alfalfa (*Medicago sativa*) fields (Schroeder and others, 1999), and fields enrolled in the Conservation Reserve Program (CRP; Schroeder and Vander Haegen, 2011).

During the winter, sage-grouse depend on sagebrush stands for both food and cover (Patterson, 1952; Dalke and others, 1963; Eng and Schladweiler, 1972; Wallestad and Eng, 1975; Beck, 1977; Remington and Braun, 1985; Thacker and others, 2012). Winter areas are characterized by large expanses of big sagebrush, predominantly located on relatively gentle south- or west-facing slopes that provide favorable thermal conditions and above-snow forage (Hupp and Braun, 1989; Doherty and others, 2008; Carpenter and others, 2010; Hagen and others, 2011; Dzialak and others, 2013). Sage-grouse exhibit fidelity to winter sites (Berry and Eng, 1985); however, birds may change habitat use in response to severe conditions (Smith, 2012).

Adaptations to a Sagebrush Diet

Sage-grouse feed on the leaves of several species of sagebrush year-round and exclusively during winter (Wallestad and Eng, 1975; Connelly and others, 2011b). Species and subspecies fed upon include Wyoming, mountain, and basin big sagebrush and black, low, silver, and alkali (*A. arbuscula longiloba*) sagebrush (Remington and Braun, 1985; Welch and others, 1988, 1991; Gregg and others, 2008; Frye and others, 2013).

Sage-grouse have evolved adaptations, including selective feeding and specialized gut morphology that allow them to be one of only three herbivores, along with pygmy rabbits (*Brachylagus idahoensis*) and sagebrush voles (*Lemmiscus curtatus*), that can, at least seasonally, survive on a diet of 100-percent sagebrush leaves. This specialization

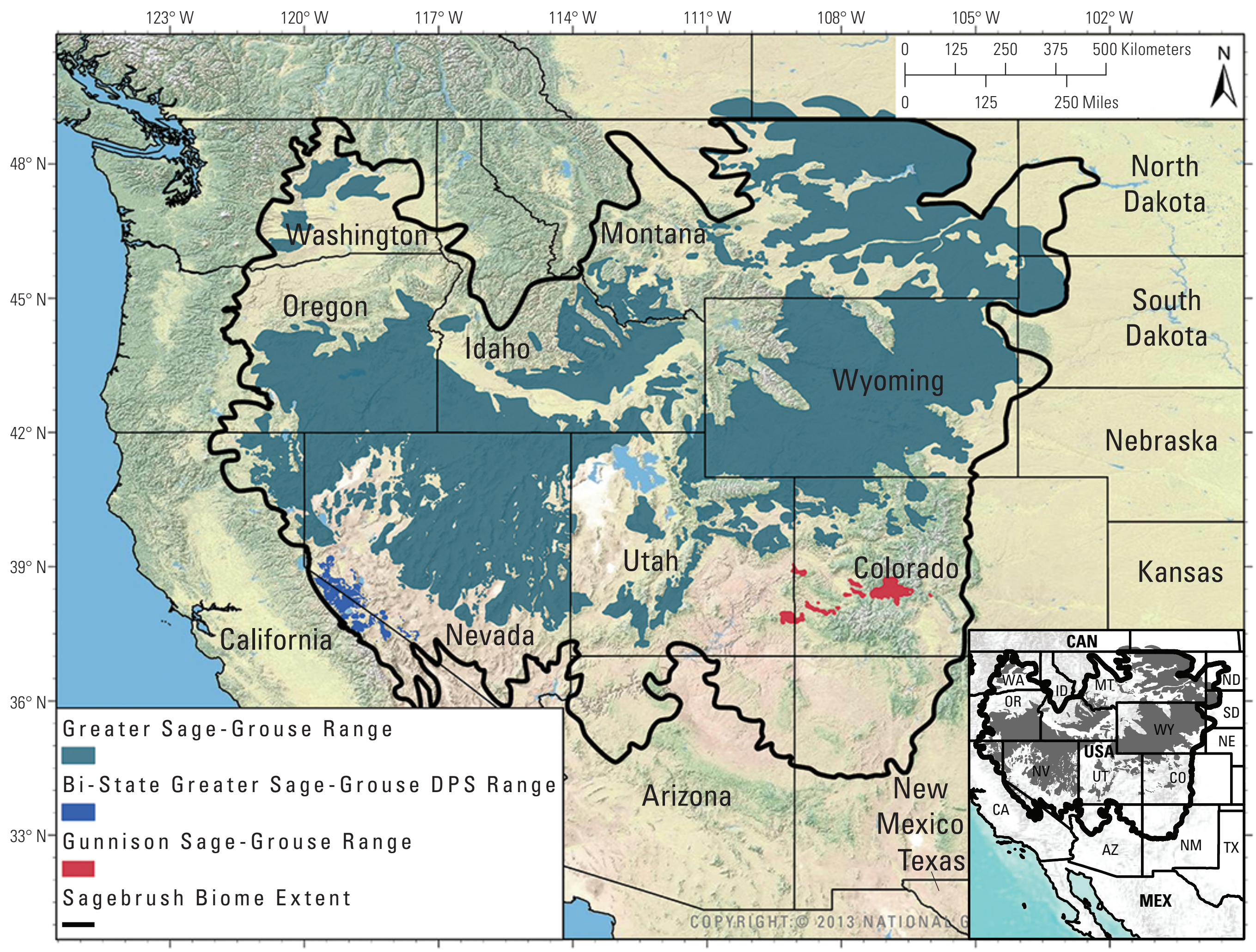

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Figure D1. Range of greater sage-grouse (*Centrocercus urophasianus*), the Bi-State Distinct Population Segment of greater sage-grouse, and the Gunnison sage-grouse (*C. minimus*) in the sagebrush (*Artemisia* spp.) biome (Jeffries and Finn, 2019). Greater sage-grouse and Bi-State Distinct Population Segment data obtained from U.S. Department of the Interior (2014). The Gunnison sage-grouse range utilized data from Braun and others (2014). DPS, distinct population segment.

also means sage-grouse can only survive where sagebrush leaves are available as a winter diet. Sage-grouse are highly selective feeders, preferring leaves of particular species, subspecies, accessions, and even growth forms of sagebrush (Remington and Braun, 1985; Welch and others, 1988, 1991; Gregg and others, 2008; Frye and others, 2013). The reasons for preferences among sagebrush taxa and plants have been attributed to selection for protein, avoidance of plant secondary compounds, or both (Remington and Braun, 1985; Sauls, 2006; Ulappa, 2011; Frye and others, 2013). In addition to selective feeding to reduce intake of secondary compounds, sage-grouse have evolved other specialized mechanisms to cope with a diet that is highly nutritious yet toxic. These include hosting symbiotic gut bacteria that eliminate monoterpenes (Sauls, 2006; Kohl and others, 2016), digestive enzymes that are resistant to inhibition by monoterpenes (Kohl and others, 2015), pathways for detoxification and excretion of plant secondary metabolites (Remington and Hoffman, 1997), and the loss of a heavily muscularized and grinding gizzard (ventriculus) which reduces the absorption of toxic plant secondary compounds.

Movements and Home Ranges

The distances sage-grouse move between seasonal habitats are highly variable across the occupied range (Connelly and others, 1988). Sage-grouse may migrate between two or three distinct seasonal ranges or not at all. Migratory populations may travel over 100 km (62 mi) between breeding and wintering areas (Tack and others, 2012). Long-distance movements from breeding to wintering areas appear to be motivated by the lack of suitable or available (because of snow depth) sagebrush for food and cover during winter in some breeding areas. For example, sage-grouse along the Montana (United States)-Saskatchewan (Canada) border nested and raised broods in silver sagebrush habitats but moved up to 122 km (76 mi) south to winter in Wyoming big sagebrush habitats, which provided a more reliable food source (Tack and others, 2012).

Little information is available regarding minimum sagebrush patch sizes required to support populations of sage-grouse. Home range calculations range from 4 to 615 km^2 (1.5 to 237.5 mi^2; Connelly and others, 2011b), and populations that move long distances between seasonal ranges may use areas exceeding 2,700 km^2 (1,042 mi^2; Leonard and others, 2000; Davis and others, 2014). Large seasonal and annual movements emphasize the landscape scale nature of the species (Connelly and others, 2011a, b).

Population Trends and Conservation Status

Numerous population trend analyses for greater sage-grouse have been performed at a variety of scales (see Garton and others, 2011, 2015; U.S. Department of the Interior, 2014, 2015c; Western Association of Fish and Wildlife Agencies, 2015). Additionally, a hierarchical integrated population model (IPM) for the Bi-State DPS of greater sage-grouse was created by Coates and others (2014a). These analyses depended in whole or in part on counts of males on leks during spring. Lek count data have numerous potential sources of bias and variability, and results should be interpreted cautiously. However, analyses based on lek data indicate a long-term decline since 1965, with declines flattening in recent years.

Concern about long-term declines of sage-grouse led to nine petitions to list greater, Gunnison, and Bi-State sage-grouse under the ESA. In the most recent (2015) finding, greater sage-grouse were found not warranted for listing (as threatened or endangered) under the ESA in 2015 (U.S. Department of the Interior, 2015c). The International Union for Conservation of Nature (IUCN) Red List has listed greater sage-grouse as near-threatened since 2004. The Bi-State DPS was found not warranted for listing under the ESA in 2015 (U.S. Department of the Interior, 2015d), but this determination has been remanded by the courts and is now under review. Since 2012, the Bi-State DPS has experienced multiple years of drought conditions associated with periods of population decline across multiple populations (Mathews and others, 2018). Gunnison sage-grouse were listed as threatened under the ESA in 2014 (U.S. Department of the Interior, 2014), and the IUCN has listed Gunnison sage-grouse as endangered since 2000.

Greater sage-grouse were designated a threatened species in Canada by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) in 1997, and redesignated as endangered in April 1998. The State of Washington listed greater sage-grouse as threatened in 1998.

Threats

Because sage-grouse are almost completely dependent on sagebrush habitats across landscape scales, they are affected to some degree by almost any perturbation within sagebrush ecosystems. The manner and the relative extent to which specific sagebrush change agents impact sage-grouse differs among greater, Bi-State, and Gunnison sage-grouse and regionally for greater sage-grouse (U.S. Department of the Interior, 2014, 2015c, d, and references therein; Chambers and others, 2017a). The most significant change agents at landscape scales include invasive plant species and the role they play in altered fire regimes, conifer expansion in the western part of the range, (chap. K, this volume; chap. L, this volume; chap. M, this volume) and oil and gas and other energy development in the eastern part of the range (chap. O, this volume). Other change agents, such as coal and hard rock mining (chap. O, this volume), free-roaming equids (chap. N, this volume), transmission lines (Gibson and others, 2018), and other infrastructure (chap. P, this volume), and cropland conversion (chap. O, this volume), may be impactful at more local scales. Grazing by domestic livestock is pervasive across the range but is considered a relatively minor impact where appropriately managed (U.S. Department of the Interior, 2015c; Smith and others, 2018b; chap. P, this volume).

Management Considerations

How to conserve, restore, and enhance sagebrush habitats for greater and Gunnison sage-grouse has been the subject of extensive research, analysis, planning, and litigation in recent years. The Science Framework, Parts I (Chambers and others, 2017a) and II (Crist and others, 2019), describes an ecological context that can be used to prioritize areas for management emphasis and guidance for determining which management strategies are likely to be effective in enhancing ecosystem resilience to disturbance and resistance to invasive plants across multiple scales. Chapter Q (this volume) describes the current management paradigm for sage-grouse, including numerous State and Federal agency conservation efforts for sage-grouse.

Chapter E. Pygmy Rabbit

By Janet L. Rachlow,[1] Ian T. Smith,[1] and Marjorie D. Matocq[2]

Executive Summary

The pygmy rabbit (*Brachylagus idahoensis*) is a habitat specialist that lives only in sagebrush (*Artemisia* spp.) landscapes, and although its geographic range spans much of the sagebrush biome, its known distribution is highly patchy. Habitat requirements include soils that are suitable for burrow construction and relatively dense stands of sagebrush shrubs. Four large and somewhat disjunct patches of primary core habitat were identified through modeling efforts, but only a small part of the predicted habitat is occupied. Because of their restricted distribution and habitat specialization, factors that remove and degrade sagebrush habitats can threaten pygmy rabbits. Population estimates are unavailable, and most monitoring efforts have focused on surveys to document presence or absence based on sign (for example, burrows and pellets). A rangewide estimate of occupied areas and predicted habitat provides a framework for population surveys, habitat conservation planning, and assessment of future changes in the species distribution.

Introduction

The pygmy rabbit (*Brachylagus idahoensis*) was originally described from a specimen collected in Idaho by Merriam in 1890 and placed in the genus of hares as *Lepus idahoensis* (Merriam, 1891). However, evidence based on dental and cranial characteristics (Hibbard, 1963; Kenner, 1965) and serum protein electrophoresis (Johnson and Wicks, 1964; Johnson, 1968) distinguished *Brachylagus* as a separate genus with only one living member. No subspecies are described for the pygmy rabbit, however, the fossil record suggests that the population in the Columbia Basin of central Washington was likely isolated from other populations for an estimated 10,000 years (Lyman, 1991, 2004), and genetic analyses have identified substantial variation among populations (Becker and others, 2011; DeMay and others, 2016, 2017). Ongoing work to apply next generation genetic analyses to this species will likely clarify patterns of diversity among populations that could represent subspecies.

The current and historical geographic range of the pygmy rabbit spans most of the Great Basin and adjoining intermountain regions and parts of the Columbia Basin in central Washington where they were reintroduced. Historically, the species was documented in eight States: California, Idaho, Montana, Nevada, Oregon, Utah, Washington, and Wyoming (U.S. Department of the Interior, 2010b). Recently, DNA evidence confirmed the presence of pygmy rabbits in northwestern Colorado (Estes-Zumpf and others, 2014). Within the geographic range, however, the distribution of pygmy rabbits is highly patchy, a pattern that also was noted for historical populations, and is likely attributable to their habitat specialization (for example, Green and Flinders, 1980; Dobler and Dixon, 1990).

Recent efforts to map the rangewide distribution of pygmy rabbits have produced the first estimates of the minimum known area occupied by this species. Occurrence data were compiled from across the range, resulting in 10,420 trusted records from the full extent of the species (Smith and others, 2019). This assessment did not include Washington because populations there are a result of ongoing reintroduction efforts following extirpation (Becker and others, 2011; DeMay and others, 2017). These records are locations where pygmy rabbits have been documented since 2000, and they can serve as a baseline for assessing the minimum area known to be occupied by the species. Assuming a 3-kilometer (km; 1.9 mile [mi]) buffer around point locations based on the median dispersal distance for females (Estes-Zumpf and Rachlow, 2009), the estimated minimum area occupied was 28,367 square kilometers (km^2; 10,953 square miles [mi^2]; fig. E1).

The known occurrences of pygmy rabbits reflect a highly patchy distribution throughout their range (Smith and others, 2019). The largest contiguous patches of occurrence are in the Wyoming Basin, but relatively large patches also occur in east-central Idaho and southwestern Montana, in southwestern Idaho, and near the intersection of the California, Nevada, and Oregon borders (fig. E1). The States with the greatest estimated occupied areas are Wyoming (8,595 km^2 [3,319 mi^2]), Idaho (7,766 km^2 [2,998 mi^2]), and Nevada (6,417 km^2 [2,478 mi^2]), representing 30 percent, 27 percent, and 23 percent of the minimum occupied area, respectively. Three other States (Montana, Oregon, and Utah) each have occupied areas greater than (>) 1,500 km^2 (579 mi^2), representing 6–7 percent of the estimated area, whereas both California (0.6 percent) and Colorado (0.3 percent) contain less than 200 km^2 (77 mi^2) each of the occupied area.

Recent efforts to create an inductive species distribution model have generated maps of primary habitat and suitable (or secondary) habitat for the species across the geographic range, excluding the reintroduced populations in the Columbia Basin (Smith and others, 2019). Predicted primary habitat for pygmy rabbits covered >132,000 km^2 (50,965 mi^2) across the range of the species, but much of this area consists of fragmented patches of varying sizes and isolation (fig. E1).

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Four relatively large core areas (>21,000 km^2 [8,108 mi^2]) of mostly contiguous primary habitat are apparent in eastern Idaho, southwestern Wyoming, northeastern Nevada, and south-central Oregon. These names reflect the geographic center of each core area, but the primary habitat expands from those areas, forming irregular patches and spanning State boundaries (for example, the eastern Idaho core also includes southwestern Montana and is bisected by mountain ranges). The general distribution pattern of the four core areas is spatially consistent with divisions among greater sage-grouse (*Centrocercus urophasianus*) populations and patterns of genetic diversity represented by microsatellite clusters (Oyler-McCance and others, 2005), suggesting that rangewide patterns of habitat distribution might affect both sagebrush (*Artemisia* spp.) obligates similarly. Landscape genetic analyses could help identify the degree to which isolation of the four core habitat areas also shapes patterns of genetic diversity across pygmy rabbits.

Areas identified as suitable habitat (224,820 km^2 [86,803 mi^2]) were generally located adjacent to primary habitat core areas, and in many cases, fill gaps between fragmented patches of primary habitat (fig. E1). In some areas, suitable habitat forms corridors joining patches of primary habitat (for example, in southeastern Idaho and central Utah). Such corridors of suitable habitat could be important in providing connectivity, especially over high-elevation mountain passes or across watershed divisions and along foothills between mountain valleys.

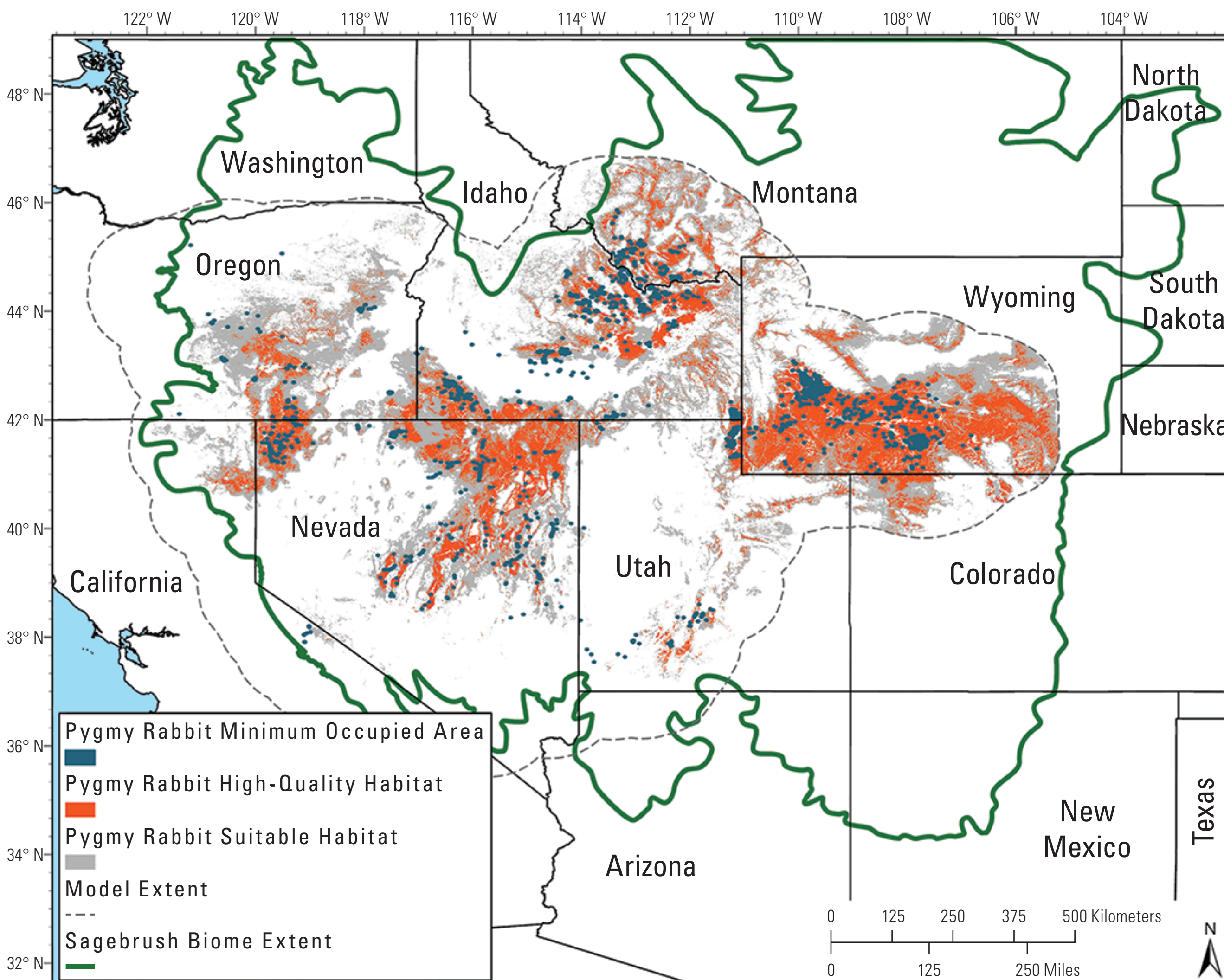

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Figure E1. Minimum occupied areas and modeled predicted habitat for pygmy rabbits (*Brachylagus idahoensis*), categorized as high-quality (or primary) habitat and suitable (or secondary) habitat in the sagebrush (*Artemisia* spp.) biome.

Habitat Selection and Dependency on Sagebrush

The pygmy rabbit is a true sagebrush obligate (Wilde, 1978; Green and Flinders, 1980; Weiss and Verts, 1984; Katzner and Parker, 1997; Gabler and others, 2001). Most often, pygmy rabbits occupy areas dominated by subspecies of big sagebrush (*A. tridentata* ssp.), but they also occur at sites with communities that contain other shrubs, including other sagebrush species (for example, three-tip sagebrush [*A. tripartita*], black sagebrush [*A. nova*], low sagebrush [*A. arbuscula*], curl-leaf mountain mahogany [*Cercocarpus ledifolius*], antelope bitterbrush [*Purshia tridentata*], and rabbitbrush [*Chrysothamnus* spp., *Ericameria* spp., and *Lorandersonia* spp.]). Strong selection for sites with relatively tall and dense sagebrush canopies has been documented in most, if not all, studies that have examined habitat relationships for this species; however, the absolute heights and percent of sagebrush cover differ across sites and studies. Although specific habitat characteristics used by pygmy rabbits vary somewhat throughout their broad geographic range, the presence of relatively tall and dense sagebrush vegetation is ubiquitous across areas occupied by this species.

Association of pygmy rabbits with sagebrush inclusions or islands of taller sagebrush surrounded by lower-stature shrubs has been noted in several sites (Larrucea and Brussard, 2008a; Ulmschneider and others, 2008). This pattern of vegetation heterogeneity is sometimes attributed to the presence of mima mounds (areas of raised microtopography) that are often distributed regularly across landscapes where they occur (Gahr, 1993; Ulmschneider and others, 2008; Parsons and others, 2016). Such patchy habitats could provide opportunities for rabbits to obtain hiding cover while also retaining visibility of the surrounding habitat to enhance detection of potential predators (Camp and others, 2013). Activity of pygmy rabbits at burrows was reduced within 100 meters (m; 328 feet [ft]) of sagebrush habitat edges where increased presence of predators and potential competitors (cottontail rabbits [*Sylvilagus* spp.] and black-tailed jackrabbits [*L. californicus*]) was documented (Pierce and others, 2011). Similarly, potential shifts of habitat use within home ranges in response to vegetation treatments were noted by Wilson and others (2011). Pygmy rabbits within and near pipeline right-of-way construction shifted patterns of space use and had reduced home range size (Edgel and others, 2018). Collectively, these studies suggest that pygmy rabbits might respond differently to naturally patchy sagebrush vegetation than to patchiness created through habitat or vegetation manipulations.

Pygmy rabbits, like sage-grouse, are sagebrush dietary as well as habitat specialists. Their winter diet consists almost exclusively of sagebrush, and although forbs and grasses are consumed during summer, sagebrush still makes up about half of their summer diets (Green and Flinders, 1980; Thines and others, 2004; Shipley and others, 2006). Pygmy rabbits exhibit strong preferences for types of sagebrush and even individual shrubs, and evidence suggests that variation in plant chemistry and nutrients influences foraging behavior (Crowell and others, 2018). Sagebrush plants with higher levels of crude protein were more likely to be browsed (Ulappa and others, 2014). Sagebrush shrubs are chemically defended by plant secondary metabolites (PSMs), which function to reduce herbivory (Kelsey and others, 1982; Frye and others, 2013). Although pygmy rabbits can tolerate relatively high levels of PSMs, concentrations of specific monoterpenes influenced foraging behavior by free-ranging pygmy rabbits (Ulappa and others, 2014), and captive individuals reduced consumption of sagebrush in response to PSMs (Shipley and others, 2006). Recent work suggests that pygmy rabbits have relatively high detoxification rates for toxins found in sagebrush, but the relationships between specific chemicals and foraging preferences are complex (Nobler and others, 2019).

In addition to selection for forage, pygmy rabbits use sagebrush vegetation for security. Pygmy rabbits have a diverse suite of terrestrial and aerial predators, and predation accounts for the majority of documented mortalities for both adults (Crawford and others, 2010) and juveniles (Price and others, 2010). Numerous studies have documented strong selection for habitats with dense sagebrush canopies presumably because such habitats allow pygmy rabbits to hide and escape from predators. Specific habitat properties associated with reducing predation risk, including concealment and visibility (that is, sightlines that provide opportunities to visually detect predators), have been linked with behavioral measures of predation risk (Camp and others, 2012; Crowell and others, 2016). Pygmy rabbits create and use burrow systems year-round. Both free-ranging and captive rabbits exhibited strong selection for proximity to burrows (Camp and others, 2012, 2017; Crowell and others, 2016), which are typically located around the base of relatively tall sagebrush shrubs (Dobler and Dixon, 1990; Gahr, 1993).

Sagebrush vegetation also provides thermal shelter, which is particularly important for this species. Pygmy rabbits are small-bodied (adults weigh approximately 500 grams [g]; about 1 pound [lb]) with relatively high ratios of surface area to volume and higher energy requirements in comparison to similar mammals (Shipley and others, 2006). In addition, because they do not hibernate or cache food, nor do lagomorphs store large fat reserves, pygmy rabbits must actively forage throughout the year, including during periods of thermal extremes (Milling and others, 2017). Dense sagebrush vegetation creates a highly heterogeneous thermal environment that facilitates behavioral thermoregulation, which is especially important during summer because rabbits, in general, are vulnerable to heat stress and hyperthermia (Marai and others, 2002). Sagebrush shrubs create microsites with significantly lower mean daily maximum temperatures and mean diurnal temperature ranges (Milling and others, 2018). During summer, pygmy rabbits selected cooler sites with lower levels of shortwave radiation (Milling and others, 2017).

Although both sage-grouse and pygmy rabbits are sagebrush obligates, their resource needs differ at least during some periods of the annual cycle (Smith, 2019). Unlike sage-grouse, pygmy rabbits do not exhibit seasonal changes in habitat use and have not been documented shifting ranges seasonally. In addition to relatively dense and tall sagebrush, soil characteristics that are conducive to burrowing, such as deep and loamy soils, are associated with year-round presence of pygmy rabbits (Wilde, 1978; Green and Flinders, 1980; Weiss and Verts, 1984; Dobler and Dixon, 1990). Their burrow systems often are associated with topographic features such as alluvial fans, drainages, and microtopography such as mima mounds, that tend to have relatively deep soils (Grinnell and others, 1930; Borell and Ellis, 1934; Weiss and Verts, 1984; Gahr, 1993; McMahon and others, 2017). Because pygmy rabbits are obligate burrowers, fine-scale heterogeneity in soil properties likely shapes their distribution to a greater degree than sage-grouse.

In addition to responding to the environment, pygmy rabbits also change it in multiple ways. They concentrate activities (digging and deposition of feces and urine) and forage heavily on sagebrush and other vegetation in proximity to burrows. Browsing by pygmy rabbits reduced sagebrush canopy cover and the percentage of individual shrubs that were alive over time (Parsons and others, 2016). However, both seedling recruitment and biomass of inflorescences increased with duration of burrow occupancy, suggesting that pygmy rabbits enhanced reproduction and recruitment of sagebrush shrubs. These results suggest that although pygmy rabbits are inconspicuous on the landscape, the species might play an important role in maintaining and augmenting heterogeneity in the sagebrush-steppe (Parsons and others, 2016).

Movements and Home Ranges

Pygmy rabbits occupy sagebrush habitats year-round and are not known to shift use of habitats or exhibit seasonal shifts in space use. However, males move over larger areas during the spring and summer breeding season, and both sexes exhibit more restricted movements during winter (Katzner and Parker, 1997; Burak, 2006; Crawford, 2008; Sanchez and Rachlow, 2008). Estimates of the sizes of home ranges vary among studies, depending on the sex, season, study area, and methods used to generate home ranges. Estimates of mean home ranges span from greater than1 hectare (ha; 2.47 acres) for females during the nonbreeding season (Burak, 2006; Crawford, 2008; Sanchez and Rachlow, 2008) to greater than (>) 12 ha (30 acres) for males during the breeding season (Sanchez and Rachlow, 2008).

Although pygmy rabbits generally concentrate activities around one or more burrow systems, several examples of long-distance movements have been reported. The longest movements were typically recorded for males during the breeding season, presumably related to reproductive status and activity (Wilde, 1978), although both male and female adult pygmy rabbits have been observed to make long-distance movements that can exceed 3.5 km (2 mi; Gahr, 1993; Katzner and Parker, 1998; Burak, 2006; Crawford, 2008; Sanchez and Rachlow, 2008).

Data from radio-tagged individuals in Idaho demonstrated that juvenile dispersal occurred between 8 and 12 weeks of age and that dispersal distances varied markedly among individuals (Estes-Zumpf and Rachlow, 2009). In that study, females were more likely to disperse than males, and the longest distance dispersal events documented (10–12 km; 6–7 mi) were undertaken by females typically in less than 1 week.

Population Trends and Conservation Status

In 2003, pygmy rabbits were petitioned for rangewide listing as threatened or endangered under the Endangered Species Act of 1973 (ESA; 16 U.S.C. 1531 et seq.; U.S. Department of the Interior, 2008). Two years later, a 90-day finding (U.S. Department of the Interior, 2005a) stated that listing was not warranted based on information provided in the petition. In response to a legal challenge, this decision was reversed, and a new 90-day finding (U.S. Department of the Interior, 2008) was published in 2008 indicating that listing may be warranted. A 12-month finding (U.S. Department of the Interior, 2010b) published in 2010, however, stated that rangewide listing for the species was not warranted.

The actions concerning listing of the species across its range did not include the isolated population of pygmy rabbits in central Washington. In 2001, that population, known as the Columbia Basin pygmy rabbit, was designated as a distinct population segment (DPS) and received emergency listing as endangered under the ESA (U.S. Department of the Interior, 2001). In 2003, a final ruling was published that replaced the emergency listing and designated the DPS as endangered (U.S. Department of the Interior, 2003). As the result of rapid declines, captive breeding was initiated for the Columbia Basin population in 2001 to produce individuals for reintroduction (Becker and others, 2011; Elias and others, 2013). Subsequently, translocation of pygmy rabbits from other States and onsite breeding efforts were implemented to support reintroduction and population recovery (DeMay and others, 2016, 2017). That population continues to face several challenges including disease and invasive plant species within onsite breeding pens, fires, and long-term resource limitation.

Population trends in pygmy rabbits are not known beyond research project scales, and monitoring emphasis is usually focused on detecting occupancy. Pygmy rabbits have many characteristics that make monitoring their populations challenging. They are uncommon across their range, occur at low densities in shrub-dominated habitats, have reclusive behaviors, and are well camouflaged for much of the year. Most monitoring efforts have focused on surveying for burrows and estimating densities of active or occupied burrow systems (for example, Thines and others, 2004; Sanchez and others, 2009; Wilson and others, 2010; Germaine and others, 2017). These methods, however, require evaluation of signs of animal activity (for example, fecal pellets, or digging) and correct attribution of both the burrow system and the signs to pygmy rabbits versus other burrowing mammals. Cameras placed at burrow entrances have helped to refine estimates of burrow occupancy (Larrucea and Brussard, 2008b; Pierce and others, 2011; Ellis and others, 2017), as have burrow surveys conducted during winter when tracks and pellets in fresh snow can help identify occupied burrow systems (Katzner and Parker, 1998; Price and Rachlow, 2011; DeMay and others, 2015). One challenge in using burrow systems for monitoring population trends is that individual rabbits use multiple burrows simultaneously (a behavior that might vary among sites, Price and Rachlow, 2011), and although pygmy rabbits are not group living, individual burrow systems are sometimes used by multiple rabbits (Wilde, 1978; Crawford, 2008; Sanchez and Rachlow, 2008; McMahon and others, 2017). Although one study in Idaho demonstrated that an index of active burrow density was monotonically related to density of individuals (Price and Rachlow, 2011), such relationships are likely influenced by environmental factors and need to be evaluated and calibrated in other areas before application. Genetic approaches have been used to monitor the breeding and reintroduction program for Columbia Basin pygmy rabbits in Washington. Those efforts included noninvasive genetic sampling of fecal pellets, which facilitated estimates of individual survival, dispersal, and reproduction (DeMay and others, 2016, 2017). Data on population estimates or trends are not available for most populations.

Threats

Primary threats to pygmy rabbits relate to loss and degradation of sagebrush habitats, especially in areas with relatively deep soils. Such factors include conversion of sagebrush communities to other vegetation states, particularly as a function of altered fire regimes (chap. J, this volume); invasion by nonnative plants (chap. K, this volume); expansion of native conifers at higher elevations (chap. M, this volume); and removal of sagebrush for agriculture, energy development and other land uses (chap. P, this volume). Some evidence also suggests that pygmy rabbits are potentially vulnerable to environmental changes associated with climate change (chap. L, this volume).

Management Considerations

Although the distribution of pygmy rabbits broadly overlaps that of greater sage-grouse, habitat restoration designed for sage-grouse cannot be assumed to also benefit pygmy rabbits. Because use of sagebrush habitats by the two species differs at local scales, priority sage-grouse habitats where treatments are likely to occur do not necessarily overlap highly suitable habitats for pygmy rabbits (chap. Q, fig. Q3, this volume). Indeed, soil properties, which are likely to strongly influence the distribution of pygmy rabbits, are not often considered in prioritizing sagebrush areas for management for sage-grouse. Additionally, pygmy rabbits might respond negatively to sagebrush treatments commonly conducted for sage-grouse or big game (Wilson and others, 2011). The maps of occupied areas and highly suitable habitat for pygmy rabbits (fig. E1; Smith and others, 2019) can provide a spatial framework for field surveys to refine information about their current distribution and prioritize areas for habitat conservation and restoration, similar to efforts for greater sage-grouse (Chambers and others, 2017a; Crist and others, 2019). Because the estimate of minimum occupied area (MOA) represents the current state of knowledge about the distribution of pygmy rabbits, it could be used to evaluate how fires and other disturbances might have shaped the species distribution, and it also serves as a baseline against which to evaluate changes in response to future land use.

Acknowledgments

Many institutions and individuals provided the records of pygmy rabbit locations used to estimate occupied area and model habitats. We thank the California Natural Diversity Database (Patrick McIntyre, Scott Osborn), Colorado Natural Heritage Program (Jeremy Siemers), Idaho Natural Heritage Program (Angie Schmidt, Leona Svancara), Montana Natural Heritage Program (Scott Blum), Nevada Natural Heritage Program and the Nevada Department of Wildlife (Chester Van Dellen, Eric Miskow, Bonnie Weller), Oregon Biodiversity Information Center (Lindsey Wise, Eleanor Gaines), Utah Conservation Data Center (Sarah Lindsey), Wyoming Natural Diversity Database (Melanie Arnett, Gary Beauvais), Stephen Germaine, Wendy Estes-Zumpf, and Michael McGee. Leona Svancara, Laura MacMahon and Sonya Knetter provided key contributions to the rangewide analyses. Justin Welty created the distribution figure. Financial support was provided through a U.S. Fish and Wildlife Service grant administered by the Western Association of Fish and Wildlife Agencies, the Idaho State Office of the Bureau of Land Management, Idaho Department of Fish and Game, and the University of Idaho.

Chapter F. Pronghorn

By Andrew F. Jakes[1]

Executive Summary

The pronghorn (*Antilocapra americana*) primarily occupies sagebrush (*Artemisia* spp.) and grassland habitats in Colorado, Montana, New Mexico, South Dakota, and Wyoming. Pronghorn feed on a variety of forage seasonally; however, during winter, they feed primarily on sagebrush. Pronghorn migrate seasonally to maximize access to high-nutrition vegetation, improve physical condition for increased reproductive success, and respond to changing environmental conditions. Populations fluctuate locally in response to annual variability in environmental gradients, such as precipitation (for example, lack of rain and extreme snowfall), and are impacted by specific and cumulative impacts from anthropogenic disturbances. In general, population trends are increasing in cultivated areas and are stable or decreasing in native grassland and sagebrush. Various tools can be used by stakeholders to mitigate these threats to allow for continued use of seasonal ranges and migratory pathways and aid in maintaining or increasing populations.

Introduction

The pronghorn (*Antilocapra americana*) is an ungulate indigenous to western North America with a range that extends across prairie, intermountain valley, sagebrush (*Artemisia* spp.) and desert habitats from northern Mexico to southern Canada (Yoakum, 2004). The pronghorn originated in the Pleistocene Era and is the only species extant in its taxonomic family (O'Gara and Janis, 2004). Pronghorn coevolved with fleet predators on the open landscapes of North America. Consequently, they have extremely keen eyesight and are the second-fastest land animal in the world (O'Gara and Janis, 2004). Currently, the distribution of pronghorn spans 23 jurisdictions in western North America, including 17 American States, 4 Mexican States, and 2 Canadian Provinces (fig. F1). There are five recognized subspecies of pronghorn across their range. Almost half of all pronghorn (47.1 percent) are found in Wyoming, and, together with Colorado, Montana, New Mexico, and South Dakota, these five States contain approximately 85 percent of the total pronghorn population (table F1; fig. F1).

Habitat Selection and Dependency on Sagebrush

Pronghorn have relatively small rumens that do not efficiently process low-quality, high-fiber foods (Schwartz and others, 1977). Pronghorn have physiological traits similar to both concentrate feeders (Van Soest, 1994) and intermediate or mixed feeders (Hofmann, 1989), suggesting they are adapted to feed on diets high in cell solubles, such as forbs and higher quality grasses. Pronghorn energetic requirements, stemming from their size and their vigilant behavior, keep them foraging throughout most of the day (Hofmann, 1989). During each season, pronghorn are forage adaptable and consume the highest available nutritional forage. For example, in the spring, pronghorn select developing grasses that provide the highest crude protein content (Schwartz and Nagy, 1976). From late spring to early summer, pronghorn show high fidelity to fawning areas and engage in birth synchrony, where they typically give birth to twins in areas where succulent forbs are selected (Gregg and others, 2001; Wiseman and others, 2006). As hiders, fawns select areas where short-distance movements are required and forage is vertically structured to promote camouflage and cover, including sagebrush (Barrett, 1984; Wiseman and others, 2006). During summer, forage quantity peaks, and pronghorn forage on diverse vegetation, including forbs, grasses, legumes, and perennial crops. When forage senescence occurs, generally in late summer, pronghorn may initiate exploratory movements to seek improved forage conditions or from social interactions during the rut (Kitchen, 1974; Hoskinson and Tester, 1980; Byers, 1997). In the fall, pronghorn select forbs and browse, including sagebrush, in addition to cultivated forbs and grasses that may still be developing. During winter, pronghorn gather on seasonal range where evergreens, namely sagebrush, provide a persistent source of nutrition. Across pronghorn range, winter ranges are larger in area than summer ranges, as during this time, individuals must continuously seek available forage to survive (Jacques and others, 2009; Suitor, 2011; Collins, 2016). In much of pronghorn range, sagebrush species are a relatively abundant source of forage that is high in protein and cell solubles and protrudes through snow, enabling pronghorn to survive through difficult and unpredictable winter periods (Schwartz and Nagy, 1976).

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[1]National Wildlife Federation.

Pronghorn resource selection is based both on environmental gradients and anthropogenic factors and is affected by the scale of selection (Jakes, 2015; Jones and others, 2019; Reinking and others, 2019). In general, across seasons, pronghorn select for natural cover types (such as grasslands and sagebrush) in areas that have the highest nutritional value during a particular season. In winter, these include visibly open landscapes, areas with less snow, and south-facing slopes. Anthropogenic features also influence selection patterns of pronghorn and, in general, higher road and fence densities are selected against (Sheldon, 2005; Gavin and Komers, 2006; Hebblewhite, 2011; Beckmann and others, 2012; Christie and others, 2015, 2017; Jakes, 2015; Jones and others, 2015, 2019).

Movements and Migration

To maintain healthy wildlife populations, species require suitable resources and the ability to move within and between suitable habitats or to new habitats (Dingle and Drake, 2007; Lowe, 2009). Animal movement provides connections between suitable habitats across spatiotemporal scales, such as daily foraging among patches, annual migrations between seasonal ranges, or dispersal events connecting populations. Migration in ungulates is an adaptive strategy that can be defined as repeated movements by individuals or population segments to discrete seasonal ranges used at different times of the year (Berger, 2004; Dingle and Drake, 2007). Because it is a repeated phenomenon, migration can be a useful focus for

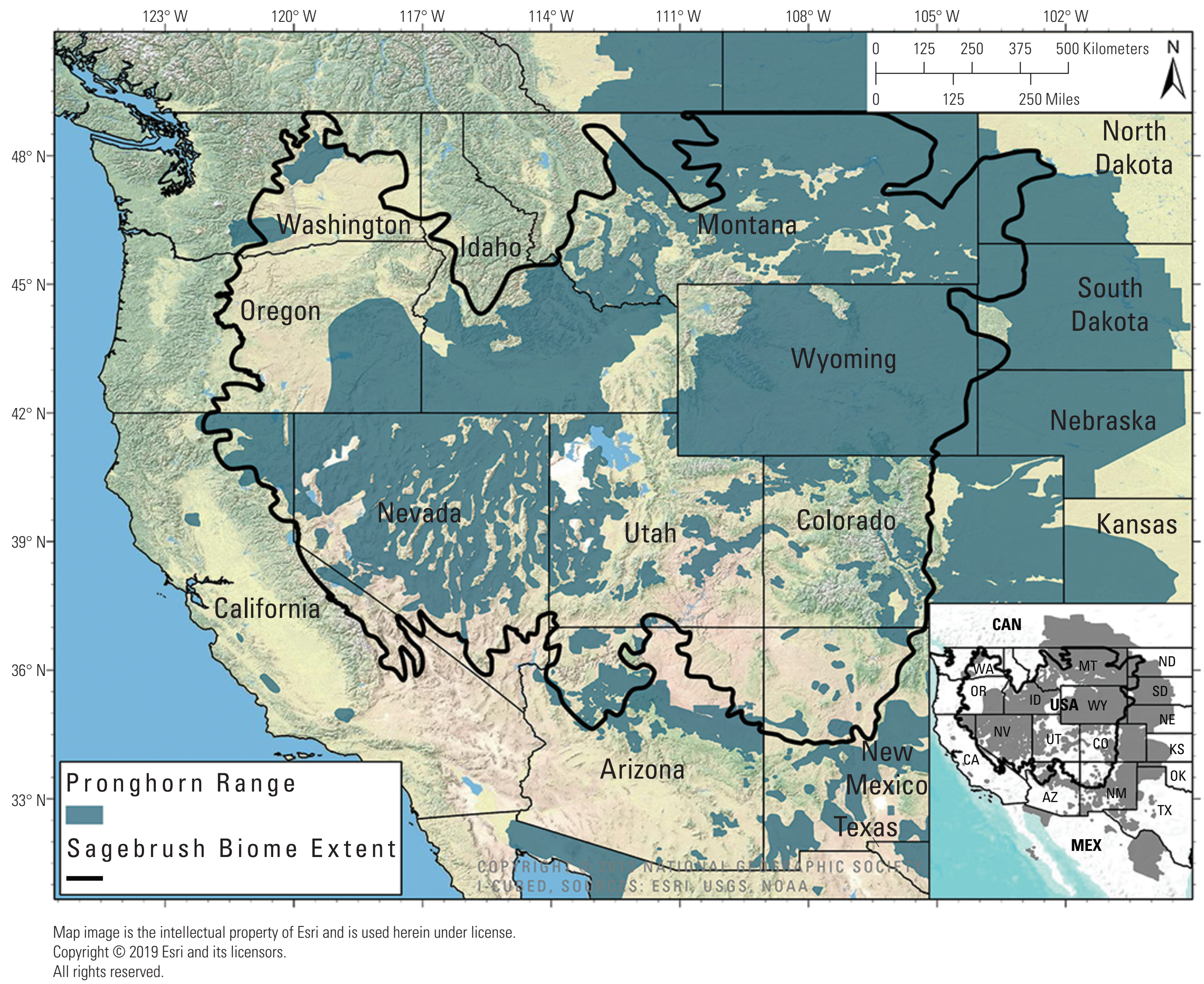

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Figure F1. Current distribution of pronghorn (*Antilocapra americana*) in relation to the sagebrush (*Artemisia* spp.) biome across western North America. State and provincial wildlife agencies within the United States, Mexico and Canada provided the jurisdictional pronghorn distributions used to create the rangewide map (inset). Generalized distributions identified at the 27th Biennial Western States and Provinces Pronghorn Workshop (Andersen and Newell, 2016) were used for States that did not provide current distributions, including Oregon and Washington (one population), and Sonora and Chihuahua in Mexico.

Table F1. Pronghorn (*Antilocapra americana*) population estimates across western North America in 2018. Pronghorn currently range in 23 jurisdictions across the United States, Mexico and Canada; however, five jurisdictions (Alberta, Canada; Chihuahua, Sonora, Coahuila, and Baja California Sur, Mexico) did not report estimates. Population estimates taken from Schroeder (2018) and directly provided by Nebraska Game and Parks.

State/province[1]	2018 population estimate	Percent of total population
Wyoming	436,800	47.1
Montana	157,965	17.0
Colorado	85,600	9.2
New Mexico	48,000	5.2
South Dakota	47,700	5.1
Nevada	30,000	3.2
Oregon	22,000	2.4
Texas	18,000	1.9
Utah	16,700	1.8
Saskatchewan	15,000	1.6
Idaho	13,000	1.4
Nebraska[2]	12,000	1.3
Arizona	11,000	1.2
North Dakota	6,038	0.7
California	3,055	0.3
Kansas	3,000	0.3
Oklahoma	1,840	0.2
Washington	150	0.0
Totals	**927,848**	**100**

[1]The Canadian province of Alberta and all States in Mexico (Baja California Sur, Chihuahua, Coahuila, and Sonora) did not report population estimates.

[2]Data provided by Nebraska Game and Parks.

identifying and maintaining landscape connectivity to sustain native ungulate populations. Pronghorn, like other native ungulates, use migration and other long-distance movements to maximize access to high-nutrition vegetation, improve physical condition for increased reproductive success, and respond to changing environmental conditions (Fryxell and Sinclair, 1988; Bolger and others, 2008, Hebblewhite and others, 2008, Avgar and others, 2014).

Pronghorn populations are often partially migratory (White and others, 2007; Jacques and others, 2009; Kolar and others, 2011; Jakes and others, 2018), which is defined as a population with a percentage of individuals that migrate (typically from summer range to winter range and back, fig. F2) and a percentage that remain residents (Dingle and Drake, 2007). Migration in pronghorn improves fawn condition by increasing their access to higher quality forage (Barnowe-Meyer and others, 2017). Depending on the length of migration, pronghorn may use stopover sites to energetically recover and amass reserves to complete the journey (Bolger and others, 2008; Sawyer and others, 2009a; Sawyer and Kauffman, 2011). At stopover sites, pronghorn select for areas of higher forage productivity with lower densities of anthropogenic features relative to migratory pathways (Poor and others, 2012; Jakes, 2015; Jakes and others, 2018). However, pronghorn have also been reported to stopover along suboptimal areas that are influenced by anthropogenic features (Seidler and others, 2015). Some individuals switched movement tactics from one year to the next (Jakes and others, 2018). This suggests that pronghorn exhibit plasticity in spatiotemporally variant systems and may learn movement tactics through social interactions, indicating that migration may not be a fixed behavior (Barnowe-Meyer and others, 2013; Jesmer and others, 2018).

Other long-distance movements by pronghorn have been observed. Both facultative winter migration, defined as migration from one winter range to another in response to extreme environmental conditions, and potential postfawning migration, defined as movement from an initial distinct fawning range during known parturition dates to a separate summer range, have been reported across pronghorn northern range (Jakes and others, 2018). In general, facultative winter migrations made by pronghorn occurred from winter range where sagebrush and other forage was unavailable to winter range where sagebrush was accessible (Jakes and others, 2018). Pronghorn may move to follow forage maturation and availability as opposed to exhibiting fidelity to any one area, although this is not well understood. Global positioning system (GPS) data from radio-collared pronghorn indicates that individuals migrating through low-quality habitat (for example, cropland) increase their movement rates to reach higher-quality forage locales (Jakes, 2015). Increased rates of movement were observed following periods where migrations were protracted by anthropogenic features (for example, roads and fences), which may also act as barriers (fig. F3; Jakes, 2015; Seidler and others, 2015).

Population Trends and Conservation Status

The pronghorn is relatively widespread and managed in the United States and Canada by State and provincial wildlife agencies. The Sonoran pronghorn (*A. a. sonoriensis*) is found in the southern portions of Arizona, New Mexico, and northern Mexico and is listed as endangered by the U.S. Fish and Wildlife Service. In addition to Sonoran pronghorn, two other subspecies of pronghorn reside in Mexico where no legal hunting of pronghorn has been allowed since 1922 (Yoakum and others, 2014). Among big game species managed by State agencies within the United States, pronghorn typically receive the least amount of attention with respect to research, monitoring, and management. This may be caused, in part, by relatively high social tolerance for pronghorn across their range. Only in highly cultivated areas are pronghorn discouraged from increasing in number. In general, the public wants to see pronghorn, and this tolerance is on an upward trend.

Though once thought to rival American bison (*Bison bison*) in sheer numbers at approximately 30 million individuals, there are an estimated 927,848 pronghorn today (Yoakum, 2004; Schroeder, 2018). Multiple pressures exerted during European settlement of western North America, particularly overharvesting, caused pronghorn distribution to contract considerably. By 1923, the population had dwindled to approximately 13,000 (Yoakum, 2004). Management efforts by State wildlife agencies, Federal agencies, private landowners, Tribes, and nonprofit organizations re-established pronghorn across most of their historical range. Pronghorn are an iconic North American species important to western State and provincial agencies for the viewing and hunting recreation they provide and for the associated economic benefits. Overall, rangewide population estimates are trending upward. The pronghorn population was estimated at 821,220 individuals in 2015, 909,848 individuals in 2017, and 927,848 in 2018, exclusive of pronghorn in Alberta and Mexico. Population sizes fluctuate locally in response to annual variability in environmental gradients, such as precipitation (for example, lack of rain and extreme snowfall), and to differing degrees of habitat conversion and anthropogenic development, which impact recruitment and mortality. Environmental variability may affect populations at the periphery of pronghorn range to a greater extent than those in core areas. When queried about population trends, biologists from management agencies generally indicated increasing population trends in cultivated areas, whereas native grassland and sagebrush habitats had either stable or downward population trends across pronghorn range (Schroeder, 2018).

Quality habitat is a primary driver in establishing harvest objectives, as well as monitoring annual recruitment and survival of pronghorn populations. The annual harvest of pronghorn throughout most of its range provides recreational and economic incentives for their management (table F2) and is regulated by State and provincial agencies.

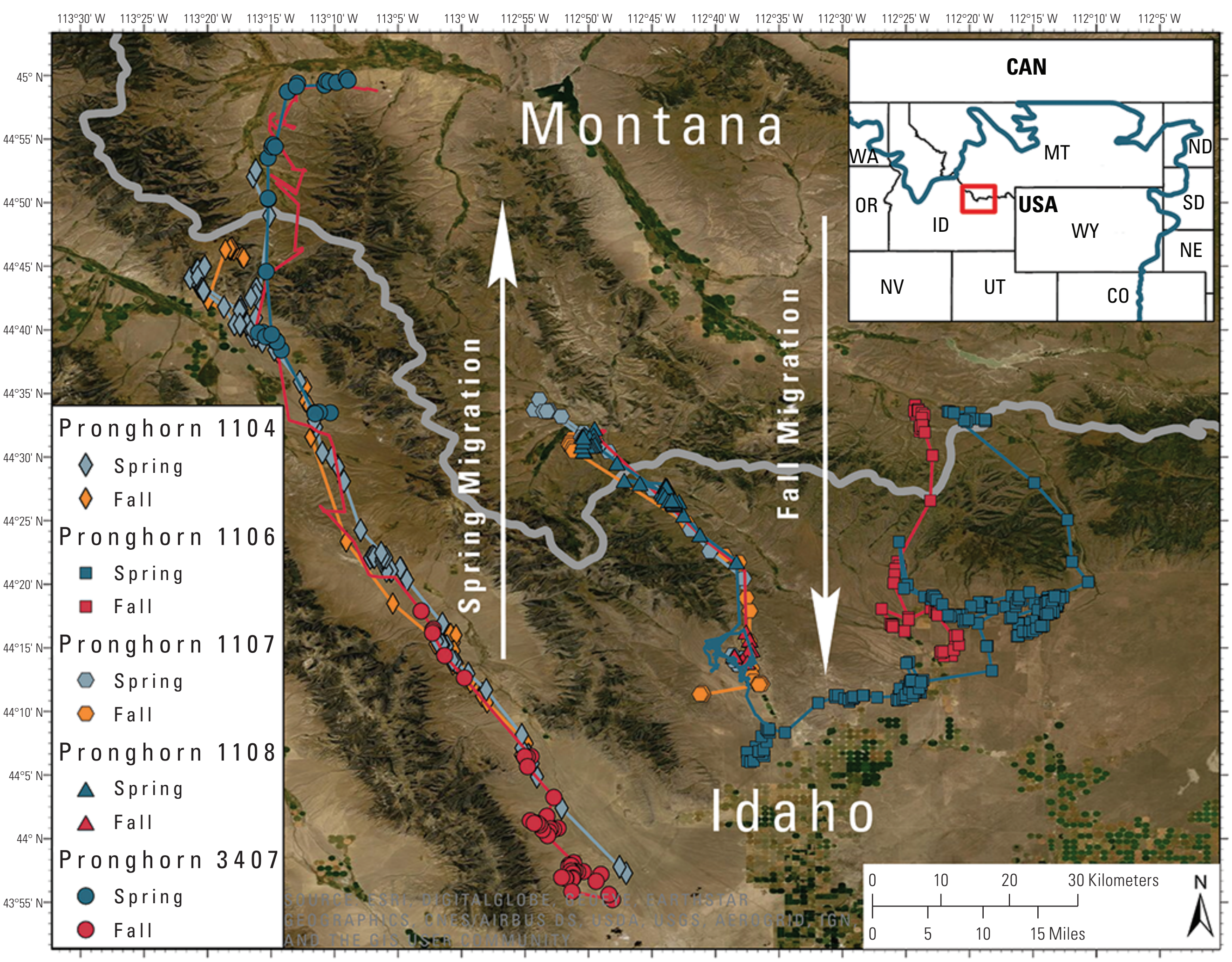

Figure F2. Pronghorn (*Antilocapra americana*) migration between summer range in Montana and winter range in Idaho. Global positioning system, or GPS, trackers for the individual pronghorn in this figure were deployed in 2004 (pronghorn 3407) and 2011 (pronghorn 1104, 1106–1108). GPS data courtesy of Scott Bergen, Idaho Department of Fish and Game.

Table F2. Pronghorn (*Antilocapra americana*) harvest estimates across western North America in 2018 (Schroeder, 2018).

State/province[1]	Bucks	Does	Totals
Arizona	548	0	548
California	190	0	190
Colorado	6,185	2,958	9,143
Idaho	1,389	406	1,795
Kansas	206	11	217
Montana	10,059	4,345	14,404
Nebraska[2]	823	263	1,086
Nevada	2,246	1,000	3,246
New Mexico	3,127	359	3,486
North Dakota	270	11	281
Oklahoma	63	65	128
Oregon	1,138	143	1,281
Saskatchewan	410	2	412
South Dakota	3,221	1,145	4,366
Texas	659	0	659
Utah	845	1,025	1,870
Wyoming	25,941	16,501	42,442
Totals	**57,320**	**28,234**	**85,554**

[1]Pronghorn are not hunted in Mexico or Washington.

[2]Data provided by Nebraska Game and Parks.

Threats

Pronghorn are impacted, to some degree, by any loss or fragmentation of sagebrush/grassland habitats and increasingly face threats from anthropogenic structures and disturbances that impact seasonal habitats and migrations. Significant threats include cropland conversion; development and other land uses such as roads and fences, which bar movement and increase mortality (chap. P, this volume); and mining and energy development (chap. O, this volume). Climate change (chap. L, this volume) is likely to be an increasing threat for populations at the southern periphery of their range.

Environmental, biological, and behavioral threats toward pronghorn abundance and distribution are also prevalent. Stochastic events, such as prolonged drought and wildfire, have consequences on fawn productivity, recruitment, and overall pronghorn mortality. Biological factors such as disease, predation, competition, and the encroachment and establishment of invasive plants, such as cheatgrass (*Bromus tectorum*) and medusahead rye (*Taeniatherum caput-medusae*), have an impact on pronghorn populations. In parts of the pronghorn range, some management practices or events are considered positive or useful, whereas some are considered negative in other areas of their range. For example, infrequent and moderately intense fire events can serve to release important nutrients into the soil and promote denser and higher-quality forage, as well as allow pronghorn access to the succulent lobes of pricklypear cactus (*Opuntia* spp.; Suitor, 2011). However, in areas with intense and frequent fires, nutrient recharge back into the system can be lost, and loss of sagebrush would be detrimental. Finally, migration is believed to be a learned trait passed down through generations (Jesmer and others, 2018). Loss of migratory corridors could result in the loss of plasticity in movement tactics for pronghorn (Jakes, 2015; Seidler and others, 2018).

Management Considerations

Landscape connectivity is paramount for native ungulates, including pronghorn, so that they can track spatiotemporal shifts in habitat, adapt to anthropogenic influences, and persist in altered landscapes that may become more suitable for colonization over time (Hilty and others, 2006). Subsequently, pronghorn may need corridors to cross anthropogenic impediments to movement to sustain connectivity across fragmented landscapes (Beier and Noss, 1998; Hilty and others, 2006). In general, natural landscapes are more connected than landscapes with anthropogenic development. Solutions toward providing safe passage of pronghorn across linear features, such as roads and fences, exist and are available for use in management efforts. For example, pasture fence designs and modifications are available that allow daily and seasonal movements of pronghorn while keeping livestock in desired pastures (Gates and others, 2012; Jones, P.F., and others, 2018). “Let down” fences are an effective design that provide gaps along a fence line to allow passage to moving pronghorn and other native ungulates (Paige, 2015). Fences along roadways can perform either as an opportunity to cross at a specific locale (if properly designed or modified) or as a funneling mechanism to a structure, depending on pronghorn use, traffic levels, and so on (O’Gara and McCabe, 2004; Sawyer and Rudd, 2005; Yoakum and others, 2014). In Wyoming, the construction of wildlife crossings has been effective in allowing for continued seasonal migrations of pronghorn and provides an additional option to communities and jurisdictions to allow for wildlife movement (Seidler and others, 2018). Finally, managing for landscape-scale connectivity is one option to combat the effects of climate change. Long-distance movements, including migrations, may be a particularly important adaptation for pronghorn at the periphery of their range because these movements offer escape from extreme environmental conditions, stochastic events, and habitat alterations.

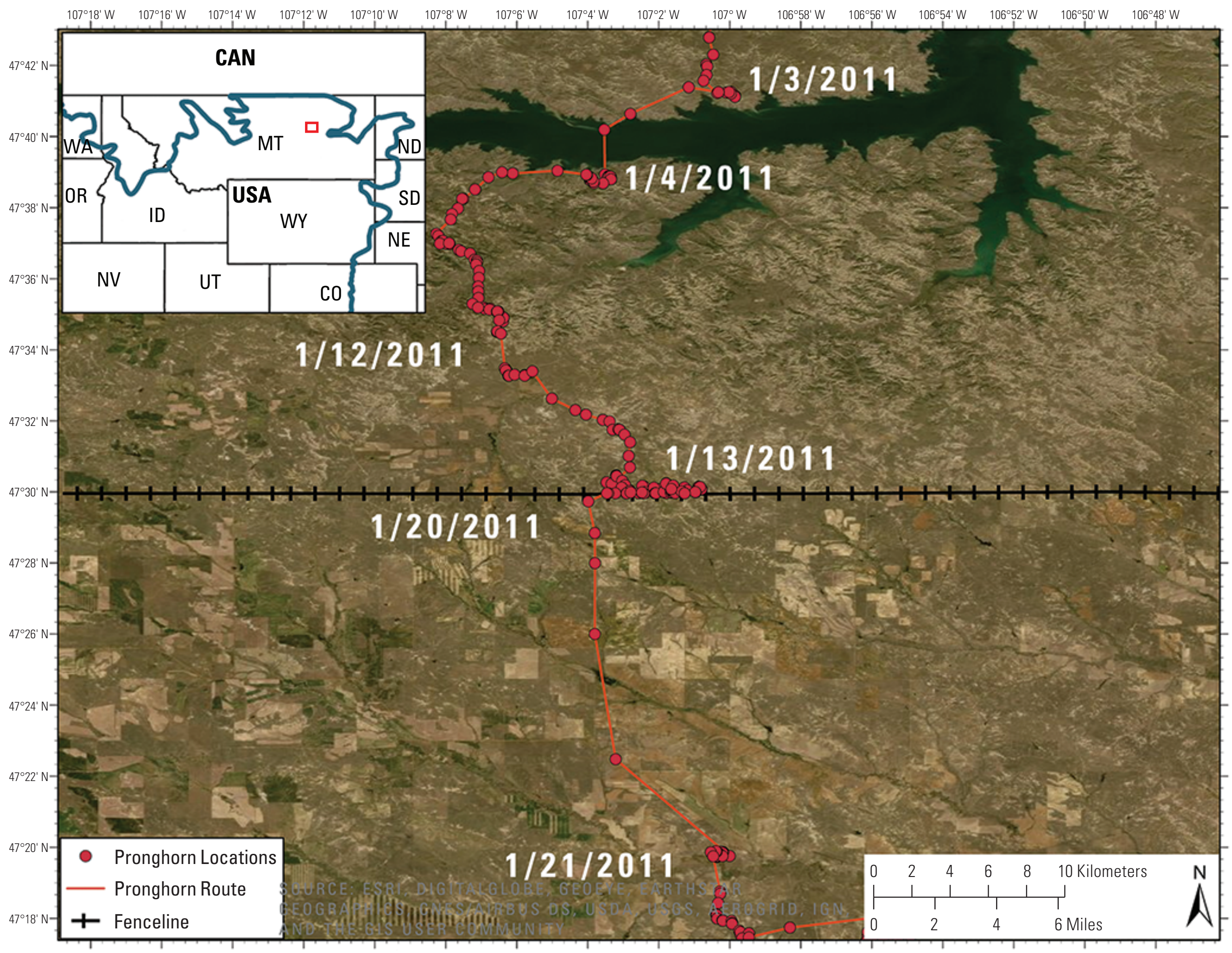

Figure F3. The effect of fencing on migrating pronghorn (*Antilocapra americana*). Red circles depict global positioning system, or GPS, locations for a female pronghorn while migrating from north to south across north-central Montana in January 2011. She interacted with this fence for over a week before successfully navigating and continuing on a facultative winter migration. Note the increased rate of travel once successfully negotiating the fence. Data from Jakes (2015).

Acknowledgments

We thank State agency experts and others for their time and insights during initial interviews related to pronghorn management, including Scott Bergen, Orin Duvuvuei, Brian Ehler, Christine Found-Jackson, Rich Harris, Andy Holland, Andy Lindbloom, Hollie Miyasaki, Amber Munig, Hall Sawyer, Cody Schroeder, Renee Seidler, Justin Shannon, Richard Shinn, Bruce Stillings, John Vore, and Mark Zornes. We thank State/provincial agency and non-governmental organization personnel for providing pertinent distribution data, including Enrique Carreon, Shawn Gray, Mike Grue, Justin Gude, Paul Jones, Luke Meduna, Pat Molini, Ken Morefield, Matt Peek, Victor Sánchez, Weston Storer, Todd Whiklo, and Don Whittaker. Specifically, we would like to thank Cody Schroeder, Nevada Department of Wildlife, for providing valuable data, insights, and contacts related to rangewide distributions and requirements. We thank Justin Welty, U.S. Geological Survey, for his support in creating chapter figures and tables and Paul Jones, Alberta Conservation Association, for his review of previous versions of this chapter. Finally, we thank Tom Remington for overall direction, review, and edits of this chapter.

Chapter G. Mule Deer

By Cody A. Schroeder[1] and Matthew J. Kauffman[2]

Executive Summary

Mule deer (*Odocoileus hemionus*) are among the most important ungulate species in western North America, conferring considerable social and economic benefits to many individuals and State and local economies throughout the West. Mule deer have a broad distribution throughout western North America, occupying distinct ecoregions from the southern coastal Alaska islands in their northern range to their southern extent in Baja, Mexico. They occur across a broad diversity of climatic regimes and vegetation associations, although some of the highest densities can be found in sagebrush-associated plant communities in the Intermountain West and Colorado Plateau Shrubland and Forest ecoregions. The greatest degree of overlap with the sagebrush biome (approximately 75 percent) occurs in the Intermountain West ecoregion, and the Colorado Plateau Shrubland, Great Plains, and Southwest Desert ecoregions also overlap the sagebrush biome.

Mule deer occupy the majority of their historical range and are not currently threatened with extirpation. However, there is widespread concern for this species because of periodic and sometimes dramatic fluctuations in population sizes from historical highs. This concern has led to concerted effort by State and Federal agencies to coordinate efforts to conserve and improve crucial habitats for this species. Mule deer inhabit a variety of shrub communities, but sagebrush is an important part of the diet for many populations, especially during the winter. Mule deer forage on a variety of different sagebrush taxa and show high selection for certain species, subspecies, and even individual plants that may be related to levels of plant defensive compounds and their interactions with nutrient levels.

Movements and the use of seasonal habitats throughout their home range is a significant factor in the ecology and population dynamics of mule deer. Many populations rely on the ability to migrate through sagebrush foothills, from their winter ranges in sagebrush basins to summer ranges in higher elevation forests. Migration corridors serve as key habitats (in terms of access to high-quality forage) for these herds. Key threats to mule deer populations include loss and degradation of sagebrush habitats from invasive species, altered fire regimes, or anthropogenic disturbances. Because of the importance of seasonal migration, mule deer populations are also impacted by barriers that impede movement through migration corridors such as highways, oil and gas development, mining operations, and residential developments.

Management considerations for mule deer include a variety of habitat treatments and prescriptions designed to return vegetation communities to early successional stages. In many cases, protection of intact sagebrush and other shrub communities may be of the highest importance to maintaining healthy mule deer populations. A variety of habitat treatments such as mechanical, hand-thinning, herbicides, and prescribed-fire treatments (where possible) will help to ensure that high-quality forage is available for mule deer.

Introduction

Mule deer (*Odocoileus hemionus*) once included up to seven recognized subspecies (Wallmo, 1981). Recent genetic work across the entire range of their distribution has suggested that many of these subspecies may not be valid, but the two black-tailed deer subspecies, *Odocoileus hemionus sitkensis* and *O.h. columbianus,* are differentiated (Latch and others, 2014). Collectively, mule deer are distributed throughout western North America from the coastal islands of Alaska down the West Coast to southern Baja Mexico, and from the northern border of the Mexican State of Zacatecas up through the Great Plains to the Canadian provinces of Alberta, British Columbia, and Saskatchewan and the southern Yukon Territory (fig. G1).

Within this wide latitudinal and geographic range, mule deer occupy areas with diverse climatic regimes and vegetation associations. Some of the largest concentrations of mule deer occur in the sagebrush (*Artemisia* spp.) biome (fig. G1). The Western Association of Fish and Wildlife Agencies' (WAFWA), Mule Deer Working Group (MDWG), divided the rangewide distribution of mule deer into seven distinct ecoregions (fig. G2; deVos and others, 2003): Intermountain West, Colorado Plateau Shrubland and Forest, Great Plains, Southwest Deserts, California Woodland Chaparral, Northern Forest, and Coastal Rainforest. The greatest degree of overlap with the sagebrush biome (approximately 75 percent) is in the Intermountain West ecoregion, and the Colorado Plateau Shrubland and Forest, Great Plains, and Southwest Deserts ecoregions also overlap the sagebrush biome.

[1]Nevada Department of Wildlife.

[2]U.S. Geological Survey, Wyoming Cooperative Fish and Wildlife Research Unit, University of Wyoming.

Habitat Selection and Dependency on Sagebrush

Mule deer are primarily browsers, with most of their diet consisting of leaves and twigs of woody shrubs. Deer digestive tracts differ from cattle (*Bos taurus*) and elk (*Cervus canadensis*) in that they have a smaller rumen in relation to their body size (Barboza and others, 2009). Because of this, deer must be more selective in their forage selection. Instead of eating large quantities of low-quality forage, such as grass, mule deer typically select the most nutritious plants and parts of plants (Hofmann, 1989; Barboza and Bowyer, 2001).

Sagebrush is an important part of mule deer diets, especially during winter months (Welch and Wagstaff, 1992; Kucera, 1997; Pierce and others, 2004; Smith and others, 2015). In the Bridger Mountains of Montana, sagebrush was the most common forage in mule deer diets during December, January, and February (Wilkins, 1957). Mule deer exhibit distinct but locally variable preferences for different types of sagebrush at species, subspecies, accessions, and even individual-plant levels (Sheehy and Winward, 1981; Welch and others, 1981, 1983; Welch and McArthur, 1986; Wambolt, 1996). This high degree of selectivity may be related to levels of plant defense compounds and the interaction of these plant compounds with nutrient levels (Scholl and others, 1977; Welch and others, 1983; Behan and Welch, 1985; Personius and others, 1987; Bray and others, 1991).

Movements and Home Ranges

Mule deer are highly mobile ungulates that use sagebrush habitats extensively. Many populations rely on the ability to migrate through sagebrush foothills, from their winter ranges in sagebrush basins to summer ranges in higher elevation

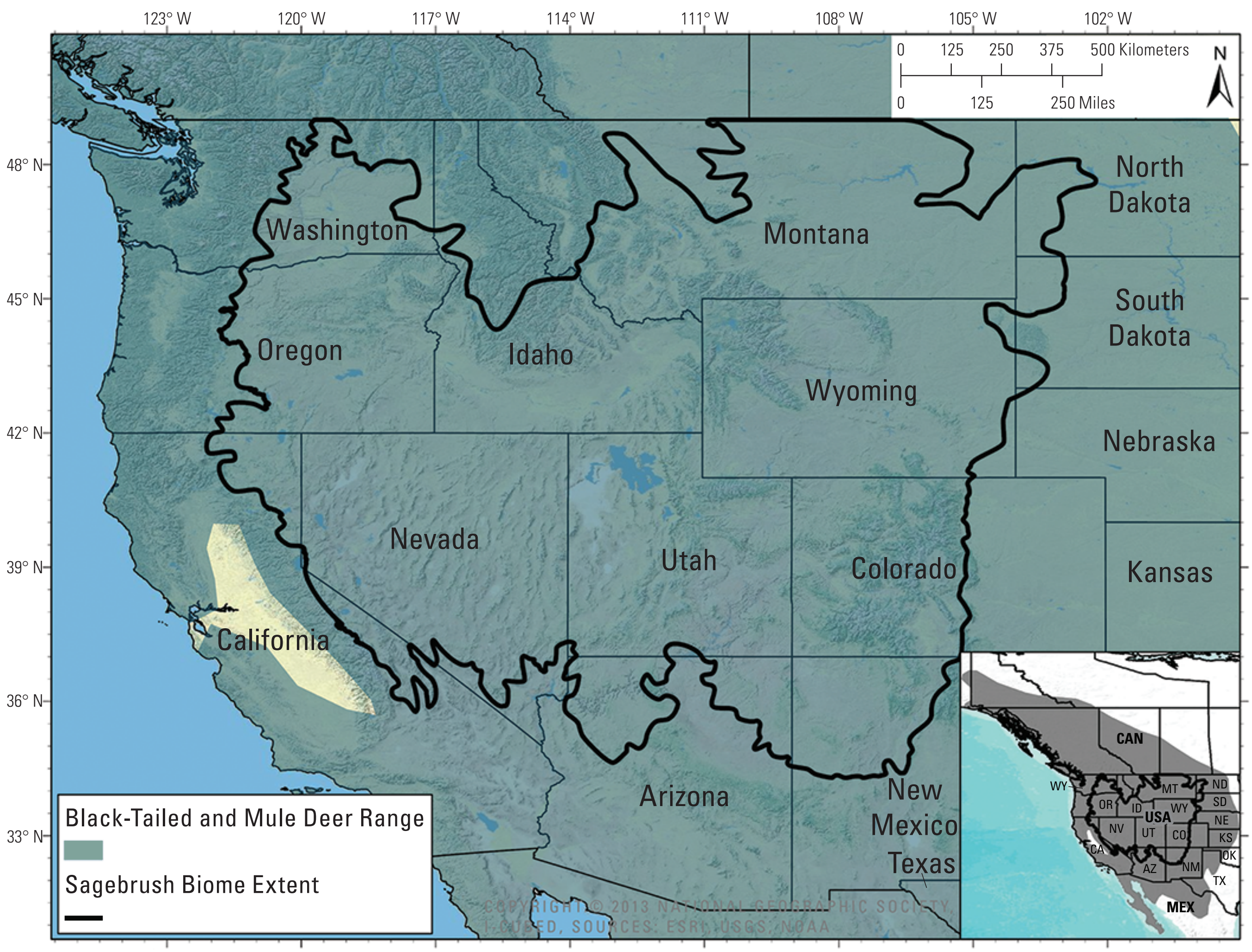

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Figure G1. Current distribution of black-tailed (*Odocoileus hemionus sitkensis*) and mule deer (*Odocoileus hemionus*) in relation to the sagebrush (*Artemisia* spp.) biome. Data from the Mule Deer Working Group (2020).

forests. Traversing these sagebrush habitats allows individual mule deer to access high-quality forage in spring and avoid deep snow in winter (Sawyer and Kauffman, 2011).

Migration corridors serve as key habitats (in terms of access to high-quality forage) for these herds (Merkle and others, 2016; Aikens and others, 2017). Recognition of migration corridors and stopovers as important habitats for mule deer is leading land managers to prioritize these areas and apply treatments to improve habitats. Because mule deer show high fidelity to migration routes and stopovers (Sawyer and Kauffman, 2011; Sawyer and others, 2014), habitat treatments can be targeted to known routes to maximize benefits to deer populations. However, treatments outside of existing routes may be less effective for mule deer because routes must be learned (Jesmer and others, 2018), and significant deviations by migrating mule deer to access higher quality forage beyond established routes is inefficient or unlikely (Blum and others, 2015). Mule deer may spend several weeks migrating from winter to summer range, but their use of specific locations along the route is limited to a few days that occur near the time of maximum vegetative growth (Jachowski and others, 2018).

Despite extensive research on deer ecology and vegetation responses to habitat treatments, understanding the tradeoffs of alternative treatments is inhibited by the lack of research on how deer use treated habitats and associated demographic responses (Beck and others, 2012), particularly along migration routes. Treatment effectiveness likely depends on forage availability, seasonal use, environmental conditions, livestock interactions, spatial extent, and more. For example, treatments that remove sagebrush may improve habitat where herbaceous vegetation is a limiting factor, but excessive removal can be detrimental to deer (Wambolt, 1998) that depend on sagebrush for forage and cover (Welch and McArthur, 1986; Anderson and others, 2012). Treatments that reduce sagebrush have been shown to have little effect when

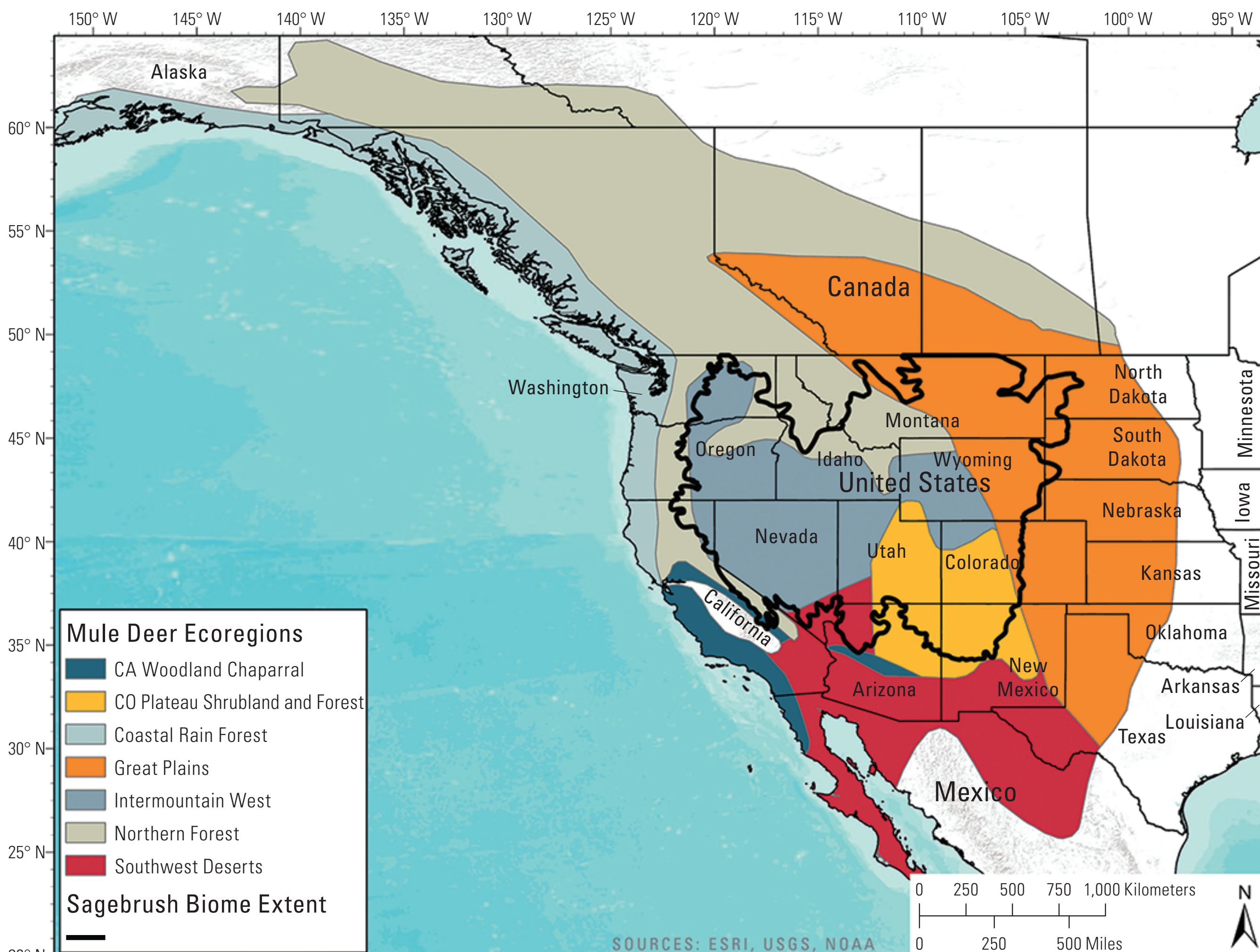

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Figure G2. Seven black-tailed (*Odocoileus hemionus sitkensis*) and mule deer (*Odocoileus hemionus*) ecoregions in North America. Data obtained from the Mule Deer Working Group (2019). CA, California; CO, Colorado.

viewed on satellite-based metrics of vegetative phenology (Johnston and others, 2018) that have been found to explain direction and timing of spring migrations by mule deer.

Native ungulate migrations are an important part of the cultural, hunting, and conservation heritage of the American West, which in turn has led to increased awareness, funding opportunities, and conservation efforts. However, given increasing levels of energy development and recreation on public lands, sprawling housing development on private lands, and increasing traffic volumes on our roadways, the long-term persistence of these migration corridors is uncertain (Kauffman and others, 2018). Development on sagebrush-dominated landscapes can affect the speed at which animals migrate, force animals to detour from established routes, or in some cases, impede migration altogether (Sawyer and others, 2009b; Sawyer and others, 2013). The associated fitness costs of such behavioral alterations have yet to be quantified.

Population Trends and Conservation Status

Mule deer still occupy most of their historical range and are not currently threatened with extirpation. However, there is widespread concern for this species because of periodic and sometimes dramatic fluctuations in population sizes (Forrester and Wittmer, 2013) and declines from historical high numbers. Mule deer are among the most economically and socially important wild mammals in western North America (Mule Deer Working Group, 2004). A recent survey by the U.S. Department of the Interior (DOI), U.S. Fish and Wildlife Service (FWS), and U.S. Department of Commerce indicated that more than 103 million Americans (about 40 percent) participated in fishing, hunting, or other wildlife-associated recreation such as photography (U.S. Fish and Wildlife Service, U.S. Department of Commerce, and U.S. Census Bureau, 2016). Approximately 9.2 million people identified as big game hunters and, among big game species, deer (white-tailed deer [*O. virginianus*] and mule deer) were the most popular species to hunt. Deer hunters spent 115 million days in the field in 2016. Big game hunters in general spent $14.9 billion on trip-related and hunting equipment expenditures (U.S. Fish and Wildlife Service, U.S. Department of Commerce, and U.S. Census Bureau, 2016). Although, this includes hunters for a variety of species, mule deer have traditionally been one of the most important big game animals in the West (Mule Deer Working Group, 2004).

Concern over declining populations relative to the high social value of mule deer led the WAFWA Mule Deer Working Group to develop a “North American Mule Deer Conservation Plan” in 2004 (Mule Deer Working Group, 2004). More recently, most western State wildlife agencies—including California, Colorado, Idaho, Nevada, Oregon, South Dakota, Utah, Washington, and Wyoming—have convened working groups and developed mule deer management plans in an attempt to reverse declines.

In February 2018, then Secretary of the Interior Ryan Zinke signed Secretarial Order (SO) 3362, “Improving Habitat Quality in Western Big-Game Winter Range and Migration Corridors.” This SO directed bureaus within DOI to work with western States to enhance and improve the quality of big-game winter range and migration corridor habitat on Federal lands. FWS provided significant funding in 2018 to States to help identify and conserve their highest priority corridors and winter ranges, and in May 2019, Secretary of the Interior David Bernhardt announced the award of $2.1 million in grants to State and local partners in Colorado, Montana, Nevada, Utah, Washington, and Wyoming for habitat conservation activities in migration corridors and winter range for elk, mule deer, and pronghorn (*Antilocapra americana*). State action plans developed in response to this grant program can be accessed at https://www.nfwf.org/programs/rocky-mountain-rangelands/improving-habitat-quality-western-big-game-winter-range-and-migration-corridors/state-action-plans. Through a public-private partnership among DOI, the National Fish and Wildlife Foundation (NFWF), and ConocoPhillips, the grants are expected to leverage more than $8.6 million in matching contributions, generating a total conservation impact of more than $10.7 million.

Mule Deer Response to Habitat Changes

Mule deer populations have fluctuated significantly over time as sagebrush and associated mountain shrub communities have been altered by a variety of natural and anthropogenic influences. Historical population highs were likely influenced by anthropogenic influences that may not be sustainable, including (1) succession of rangelands from dominance by grasses to dominance by woody plants that constitute superior mule deer habitat (Leopold, 1950; Julander, 1962; Longhurst and others, 1976); (2) conversion of forests to shrublands by wildfire and logging that generally resulted in improved deer habitat, particularly availability of browse (Lyon, 1969); (3) conservation and predator control that dramatically reduced deer mortality (Leopold and others, 1947; Rasmussen and Gaufin, 1949); and (4) reduction in numbers of livestock on the open range increased the amount of forage available to mule deer (Rasmussen and Gaufin, 1949). Reduced deer populations in much of their range today reflect the cumulative effect of human land use, livestock land use, and fire suppression. These factors have led to a wide range of conditions in sagebrush-grasslands and mixed mountain shrub communities, which continue to become less productive for mule deer (Anderson, 1958).

If primary productivity and vigor of sagebrush grasslands and mixed mountain shrub communities decline, the availability of food or browse for mule deer consumption also declines (Tollefson and others, 2010). Competition between free-ranging and domestic ungulates for the remaining vegetative occurs as mule deer switch to lower quality but more abundant food sources (Barboza and others, 2009). Specific effects on mule deer herds are difficult to discern because many other factors may affect vegetative production and deer populations. Annual variation in vegetative production and winter severity can substantially change mule deer population levels between years. Consequently, long-term wildlife population trends will reflect the changes in vegetative community structure and age (Bergman and others, 2014a).

Threats

Key management challenges for mule deer habitat (including forage and cover) include loss and degradation of sagebrush and other browse species as a result of invasive plant infestations (chap. K, this volume) and fire (chap. J, this volume), conversion of native vegetation to residential developments (Johnson and others, 2017; chap. P, this volume), oil and gas development (Wyckoff and others, 2018; chap. O, this volume), conifer expansion (chap. M, this volume), and in some areas, cumulative habitat degradation from improper grazing (chap. P, this volume). Chronic wasting disease (CWD) is of increasing concern as an impediment to the long-term viability of mule deer populations throughout the West (Western Association of Fish and Wildlife Agencies, 2017). At sufficiently high prevalence (for example, greater than 5 percent among adult females), CWD can measurably lower overall survival among prime reproductive cohorts and contribute to depressed population growth rates (Miller, M.W., and others, 2008; Dulberger and others, 2010; DeVivo and others, 2017; Colorado Parks and Wildlife, 2018). This disease is well-established within mule and white-tailed deer (and elk) populations in parts of Colorado, Montana, Nebraska, New Mexico, South and North Dakota, Utah, and Wyoming, as well as Alberta and Saskatchewan, but is not yet established in the Great Basin or along the West Coast.

Management Considerations

Management for Early Successional Stages to Provide Forage

Deer populations do not consistently respond demographically to each habitat alteration, and interactions occur along soil and climatological gradients (Dasmann and Dasmann, 1963; Chambers and others, 2017a). However, mule deer tend to favor early successional habitat stages for foraging. Numerous tools are available for converting vegetative associations to an earlier successional stage, including fire, grazing, or mechanically or chemically induced changes. Removing climax vegetation and providing early successional communities favors mule deer in forest and chaparral habitats by increasing the amount and quality of forage (Brown, 1961; Dasmann and Dasmann, 1963; Taylor and Johnson, 1976; Krueger, 1981; Thill and others, 1990; Kucera and Mayer, 1999). However, early succession forage plants may be higher in digestion-inhibiting secondary plant compounds (Happe and others, 1990). The benefits of fire on herbaceous plants are generally short-term, about 6–11 months, but the beneficial effects on browse species can be longer lasting (Carlson and others, 1993). Fire can improve winter forage, as young forbs, grasses, and shrubs have elevated concentrations of protein and *in vitro* digestible organic matter in winter diets of mule deer (Hobbs and Spowart, 1984). However, large, intense wildfires can eliminate or reduce shrubs from winter ranges, decreasing their value to deer and even eliminating deer use.

Manage for a Diversity of Key Plants, Including Forbs

Treatments designed to increase desired forage components are beneficial to mule deer. Except for moisture, which is generally higher in more preferred species, nutritional components may not show any consistent relationship to preference in deer diets (Radwan and Crouch, 1974). White-tailed deer feed more on grass and forbs on excellent-condition range, whereas they feed more on browse on poor-condition range (Bryant and others, 1981). Forage quantity is generally not a problem during winter, but forage with adequate digestible energy and crude protein may be limited (Bartmann, 1983). Digestible energy seems to be more limiting than protein to mule deer health and reproduction (Bryant and others, 1980). Providing a diversity of forage composition across a landscape provides the greatest opportunity for mule deer to meet their year-round nutritional requirements.

Acknowledgments

We would like to acknowledge Jim Heffelfinger, Orrin Duvuvuei, and members of the Mule Deer Working Group for assistance in in drafting and reviewing this chapter. We also acknowledge Justin Welty, U.S. Geological Survey, for assistance in producing and editing maps of mule deer and sagebrush distribution.

Chapter H. Sagebrush-Dependent Small Mammals

By Patricia A. Deibert[1] and Dawn M. Davis[1]

Executive Summary

Small mammals provide a diverse presence within the sagebrush (*Artemisia* spp.) biome, often serving key ecological functions such as seed dispersal, soil aeration (burrowing mammals), and prey. While the sagebrush biome hosts a wide array of small mammals, only a few are considered sagebrush obligates. Those species, as identified by sagebrush and mammal experts, are described in this chapter, along with known habitat requirements and threats. However, most species lack sufficient life history data, which is needed to adequately understand their habitat needs, and therefore the impacts of sagebrush management on species persistence. Despite the prevalence of small mammals across the sagebrush biome, the array of species discussed here have largely been understudied, and no special management activities have been developed or implemented for many of these species. For most species, there is insufficient information on vital rates, distribution, and habitat use. Most management actions are surmised from actions for other small mammals and include protecting habitats from loss and fragmentation. Although management to conserve intact sagebrush landscapes is presumed to benefit these species, additional research is needed to inform conservation efforts.

Introduction

Small mammals provide a diverse presence within the sagebrush (*Artemisia* spp.) biome, and provide key ecological functions, including seed dispersal, soil aeration, and as a food resource. While the sagebrush biome hosts a wide array of small mammals, the species discussed below were selected by sagebrush and mammal experts as either having a unique association with sagebrush or having their entire range completely contained within the sagebrush ecosystem. Therefore, management activities within sagebrush have the potential to influence these species. This chapter summarizes key information for the dark kangaroo mouse (*Microdipodops megacephalus*), Columbia Plateau and Great Basin pocket mice (*Perognathus parvus* and *P. mollipilosus*, respectively), Preble's and Merriam's shrews (*Sorex preblei* and *S. merriami*, respectively), Ord's kangaroo rat (*Dipodomys ordii*), sagebrush vole (*Lemmiscus curtatus*), Southern Idaho and Wyoming ground squirrels (*Urocitellus endemicus* and *U. elegans*, respectively), Wyoming pocket gopher (*Thomomys clusius*), white-tailed prairie dog (*Cynomys leucurus*), and black-tailed jackrabbit (*Lepus californicus*). Many more small-mammal species occur within the sagebrush ecosystem or associated habitats (for example, mesic areas). The limited list described below is not intended to dismiss these other species, but rather to focus on the few that are clearly obligates or require sagebrush during at least one phase of their life cycle (for example, winter forage). State wildlife agencies have identified the other species associated with sagebrush that may be of conservation concern (sometimes associated with the condition of the sagebrush habitat) through their State wildlife action plans.

Dark Kangaroo Mouse

Taxonomy and Distribution

The dark kangaroo mouse is a small bipedal mouse with conspicuous white facial markings (Arkive, 2016), geographically variable pelage color (Hafner and Upham, 2011), and external fur-lined cheek pouches to carry food. Their tail is widest in the middle where fat is stored for use during hibernation (Utah Wildlife Action Plan Joint Team, 2015). Dark kangaroo mice typically move by hopping on their hind legs, using their tail for balance. This species is found in California, Nevada, Oregon, Utah, and extreme southwestern Idaho (Dobkin and Sauder, 2004), and current distribution is thought to mimic historical distribution (fig. H1, Dobkin and Sauder, 2004; Utah Wildlife Action Plan Joint Team, 2015). Local populations of dark kangaroo mice are patchy and genetic analyses suggest they are distinct "islands" (Hafner and Upham, 2011). Loss of any individual "island" may have larger population impacts if it results in loss of connectivity or genetic information.

There is little information regarding population trends, but populations are suspected to be decreasing (International Union for Conservation of Nature, 2016). The species is considered rare based on the frequency of captures during trapping (Hafner and Upham, 2011). Populations in the northern part of their distribution appear to be declining faster than those in the southern range, potentially because of conifer expansion (Nevada Wildlife Action Plan Team, 2013), poor habitat quality, and smaller, more isolated populations (Hafner and Upham, 2011). In Utah, wildlife managers are concerned that the northern populations are small and fragmented, and some may be locally extinct (Utah Wildlife Action Plan Joint Team, 2015). Factors that influence abundance and distribution are likely similar to those affecting other rodents in this family (*Heteromyidae*), including destruction and degradation of native habitats (Dobkin and Sauder, 2004).

[1]U.S. Fish and Wildlife Service.

Habitat Selection and Sagebrush Association

Dark kangaroo mice occur in areas of big sagebrush (*A. tridentata*) with gravelly or fine-textured soils (Dobkin and Sauder, 2004; Hafner and Upham, 2011), typically at elevations below pinyon (*Pinus* spp.)-juniper (*Juniperus* spp.) occurrence (Nevada Wildlife Action Plan Team, 2013). Other associated vegetation includes rabbitbrush (*Chrysothamnus* spp., *Ericameria* spp., *Lorandersonia* spp.), greasewood (*Sarcobatus vermiculatus*), shadscale (*Atriplex confertifolia*), and horsebrush (*Tetradymia* spp.; Dobkin and Sauder, 2004; Nevada Wildlife Action Plan Team, 2013). Habitat characteristics that affect presence and abundance of dark kangaroo mice are poorly understood (Dobkin and Sauder, 2004), but Ghiselin (1970) hypothesized that soil characteristics are a primary factor in species occurrence and rangewide distribution. Dark kangaroo mice are habitat specialists of open and sandy habitats (Hafner and Upham, 2011). Because of the overlap in species distribution with sagebrush, dark kangaroo mice are classified as sagebrush dependent.

Threats

Habitat loss and fragmentation resulting from wildfire, invasive grass establishment, and conifer encroachment have negatively affected the dark kangaroo mouse through loss of population connectivity (Hafner and Upham, 2011). Populations in Idaho are at extreme risk owing to restricted distribution in that State and the potential for wildfire based on the presence of invasive annual grasses (Idaho Department of Fish and Game, 2017). Other threats include unsustainable livestock grazing (Dobkin and Sauder, 2004; Nevada Wildlife Action Plan Team, 2013), habitat loss caused by agricultural conversion (Dobkin and Sauder, 2004), and climate change (Nevada Wildlife Action Plan Team, 2013). Given the localized nature and small size of most populations of dark kangaroo mice, single catastrophic events (such as wildfire) may result in their local extirpation.

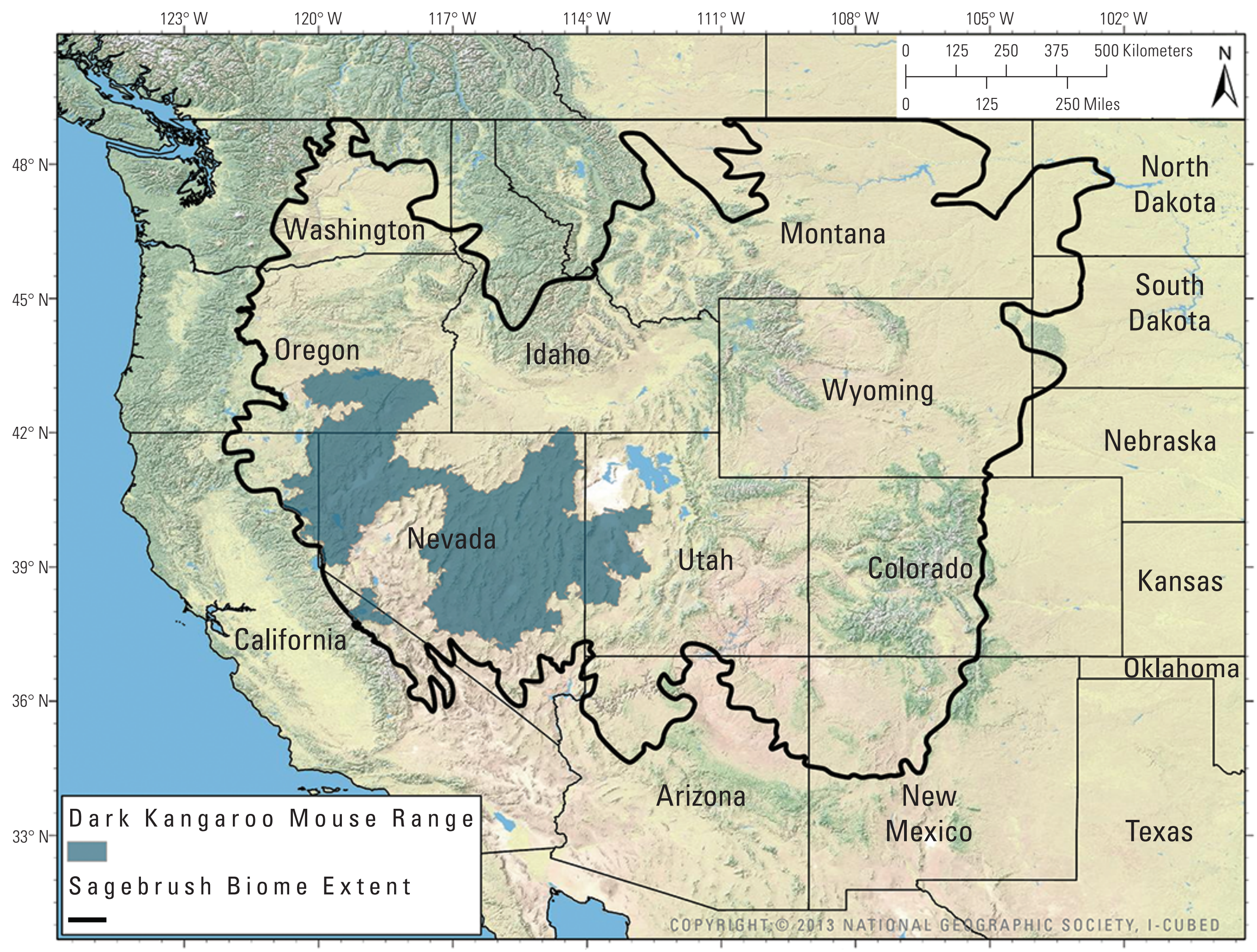

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Figure H1. Range of the dark kangaroo mouse (*Microdipodops megacephalus*) in the sagebrush (*Artemisia* spp.) biome. Data were developed and created by the U.S. Geological Survey (2013).

Management Considerations

Management to reduce invasive annual grasses and the risk of wildfire are important to retaining local populations (Nevada Wildlife Action Plan Team, 2013; Idaho Department of Fish and Game, 2017). Resolving data gaps on the species status, stability of isolated populations, and tolerance thresholds for invasive grasses and pinyon-juniper will be key to developing management strategies (Nevada Wildlife Action Plan Team, 2013; Idaho Department of Fish and Game, 2017).

Great Basin and Columbia Plateau Pocket Mouse

Taxonomy and Distribution

Great Basin pocket mice and Columbia Plateau pocket mice are large (205 millimeters [mm]; 8 inches [in.] in length; Montana Field Guide, 2016a), buff-colored mice with elongated hind legs, fur-lined external cheek pouches, and a bicolored tail that is longer than the body (Verts and Kirkland, 1988; Buskirk, 2016; Montana Field Guide, 2016a). In 2014, these mice were split into two species along previously recognized northern (Columbia Plateau) and southern (Great Basin) clades based on nuclear and mitochondrial DNA analyses (Bradley and others, 2014; Riddle and others, 2014). However, the two species are morphologically similar and difficult to identify without genetic analyses (Riddle and others, 2014; Buskirk, 2016).

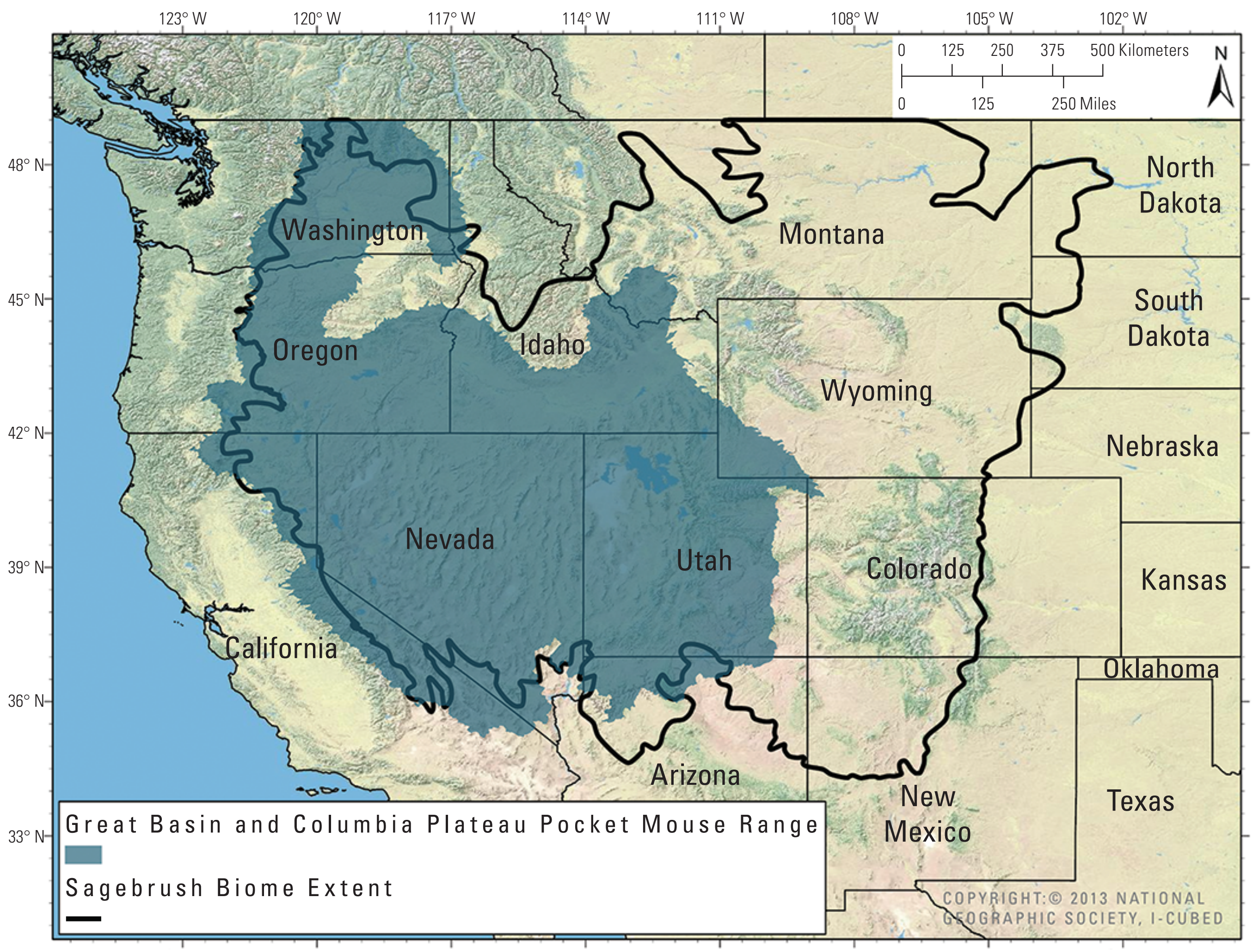

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Figure H2. Range of the Great Basin pocket mouse (*Perognathus mollipilosus*) and Columbia Plateau pocket mouse (*P. parvus*) in the sagebrush (*Artemisia* spp.) biome. Data were developed and created by the U.S. Geological Survey (2013a).

Great Basin and Columbia Plateau pocket mice are found in the Intermountain West, documented in Arizona, California, Idaho, Montana, Nevada, Oregon, Utah, Washington, and Wyoming (Dobkin and Sauder, 2004). Columbia Plateau pocket mice range from south-central British Columbia, Washington, Oregon and southwestern Idaho. Great Basin pocket mice occur in Arizona, California, Nevada, Oregon, Utah, and Wyoming (fig. H2; Riddle and others, 2014). Data from Wyoming suggests the range in that State is more expansive than currently recognized in the literature (Wyoming Field Guide, 2020).

The species are relatively common, but population trends are unknown (Dobkin and Sauder, 2004; Buskirk, 2016). They are the most abundant small mammal captured in the Great Basin, and population densities may reach 80 or more per hectare (ha) in years of high precipitation (Montana Field Guide, 2016a). Home ranges are rapidly filled when the initial occupant is removed, but it is unclear if the replacement is caused by immigration or by expansion of adjacent home ranges (Verts and Kirkland, 1988). Population abundance, as estimated by capture per unit effort while trapping, appears to fluctuate with precipitation, suggesting a relationship between food abundance and number of mice (Verts and Kirkland, 1988).

Habitat Selection and Sagebrush Dependency

Great Basin and Columbia Plateau pocket mice are most consistently found in big sagebrush and native grass areas (Verts and Kirkland, 1988) but have occasionally been trapped in other arid plant communities such as shadscale, greasewood, rabbitbrush, winterfat (*Krascheninnikovia lanata*), and spiny hopsage (*Grayia spinosa*; Dobkin and Sauder, 2004). These mice are associated with sandy, deep soils that allow burrow excavation (Dobkin and Sauder, 2004). Abundance of these species increases with increased shrub cover and soil sand content, although it has not been recorded in greater than (>) 40 percent shrub cover in Montana (Verts and Kirkland, 1988). Increasing ground (nonshrub) vegetative cover appears to positively influence species abundance (Parmenter and MacMahon, 1983; Verts and Kirkland, 1988; Montana Field Guide, 2016a), and it has been captured throughout its range in areas with an understory of cheatgrass (*Bromus tectorum*; Dobkin and Sauder, 2004). The loss of shrubs did not appear to influence the microclimate of this species as most foraging occurs at night (Parmenter and MacMahon, 1983). However, shrubs provide the necessary microclimate for herbaceous understory growth and therefore indirectly create food resources for the pocket mouse (Parmenter and MacMahon, 1983). In an opportunistic study in Idaho, researchers observed no pocket mouse population response to fire, presumably because of the quick regeneration of herbaceous plants (Hedlund and Rickard, 1981). However, wildfire has been documented to negatively affect the species (Montana Field Guide, 2016a). These pocket mice are considered sagebrush near-obligates.

Threats

The lack of natural history information limits the identification of threats to pocket mouse species (Buskirk, 2016). Studies regarding the impacts of domestic livestock grazing present conflicting results (Rosenstock, 1996; Dobkin and Sauder, 2004, and references therein). The lack of consistent results is likely associated with the intensity and differential timing of livestock grazing between study areas. Studies on the effect of wildfire are also inconclusive (Hedlund and Rickard, 1981; Montana Field Guide, 2016a), which is likely related to the intensity and timing of fire and the amount of vegetation removed by fire. Drought may also have negative influences on the abundance of pocket mice, females will produce no young if there are insufficient food resources (Verts and Kirkland, 1988).

Management Considerations

The retention of shrub overstories is identified as being important to sustaining the species' food supply (Parmenter and MacMahon, 1983).

Merriam's Shrew

Taxonomy and Distribution

The Merriam's shrew is a small to medium size shrew (Armstrong and Jones, 1971; Buskirk, 2016) easily identified by its pale coloration (Freeman and others, 1993) and distinctly bicolored tail (Buskirk, 2016; Montana Field Guide, 2016b). Merriam's shrew occurs in Arizona, British Columbia (Canada), California, Colorado, Idaho, Montana, Nebraska, Nevada, New Mexico, North Dakota, Oregon, South Dakota, Utah, Washington, and Wyoming (Shaughnessy and Woodman, 2015; International Union for Conservation of Nature, 2016; fig. H3). There are no estimates of population densities or numbers, and the species likely occurs in low abundance as several hundred trap nights are often required to capture one specimen in areas where the species is known to occur (Johnson and Clanton, 1954). Population trends in Washington were estimated from the reduction of sagebrush-steppe habitats, not from actual trapping results (Washington Department of Fish and Wildlife, 2015). Merriam's shrews may be more extensively distributed than known as the immense trapping effort necessary to detect its presence causes studies to be rarely conducted (Azerrad, 2004). The species is believed to be widely distributed across its range (International Union for Conservation of Nature, 2016).

Habitat Selection and Sagebrush Dependency

Merriam's shrews are most commonly recorded in sagebrush-bunchgrass communities (Johnson and Clanton, 1954; Nevada Wildlife Action Plan Team, 2013; Dyke and others, 2015; Buskirk, 2016; International Union for Conservation of Nature, 2016). However, the species occurs in a wide variety of habitats, including grasslands, pinyon-juniper, mountain-mahogany (*Cercocarpus* spp.), mixed woodlands, subalpine meadows, moist but not saturated sites, and nonnative grasses (for example, timothy hay [*Phleum pratense*]; Montana Field Guide, 2016b). Merriam's shrews prefer drier habitats than other shrews (Johnson and Clanton, 1954; Armstrong and Jones, 1971; George, 1990; Azerrad, 2004; Montana Field Guide, 2016b). The association with sagebrush may not indicate that Merriam's shrew is a habitat obligate but rather may reflect similar abiotic habitat conditions that favor both species (Shaughnessy and Woodman, 2015). Shrub cover at capture sites ranged from 5 to 71 percent, including a site with 30 percent juniper cover (Montana Field Guide, 2016b).

Threats

No threats have been documented, but the species is thought to be vulnerable to habitat conversion resulting from wildfires, invasive annual grasses, agricultural activities, decline and fragmentation of sagebrush habitats (Washington Department of Fish and Wildlife, 2015), and loss of prey caused by insecticides (Azerrad, 2004). Impacts of grazing on the shrew are unknown but are suggested based on studies on congeneric species (Dobkin and Sauder, 2004). Research on effects of grazing is consistently highlighted as a research need.

Management Considerations

Most management actions are surmised from actions taken for other small mammals and include protecting habitats from loss and fragmentation (Montana Field Guide, 2016b), vegetation manipulation, and reduction or elimination of the use of insecticides (Azerrad, 2004).

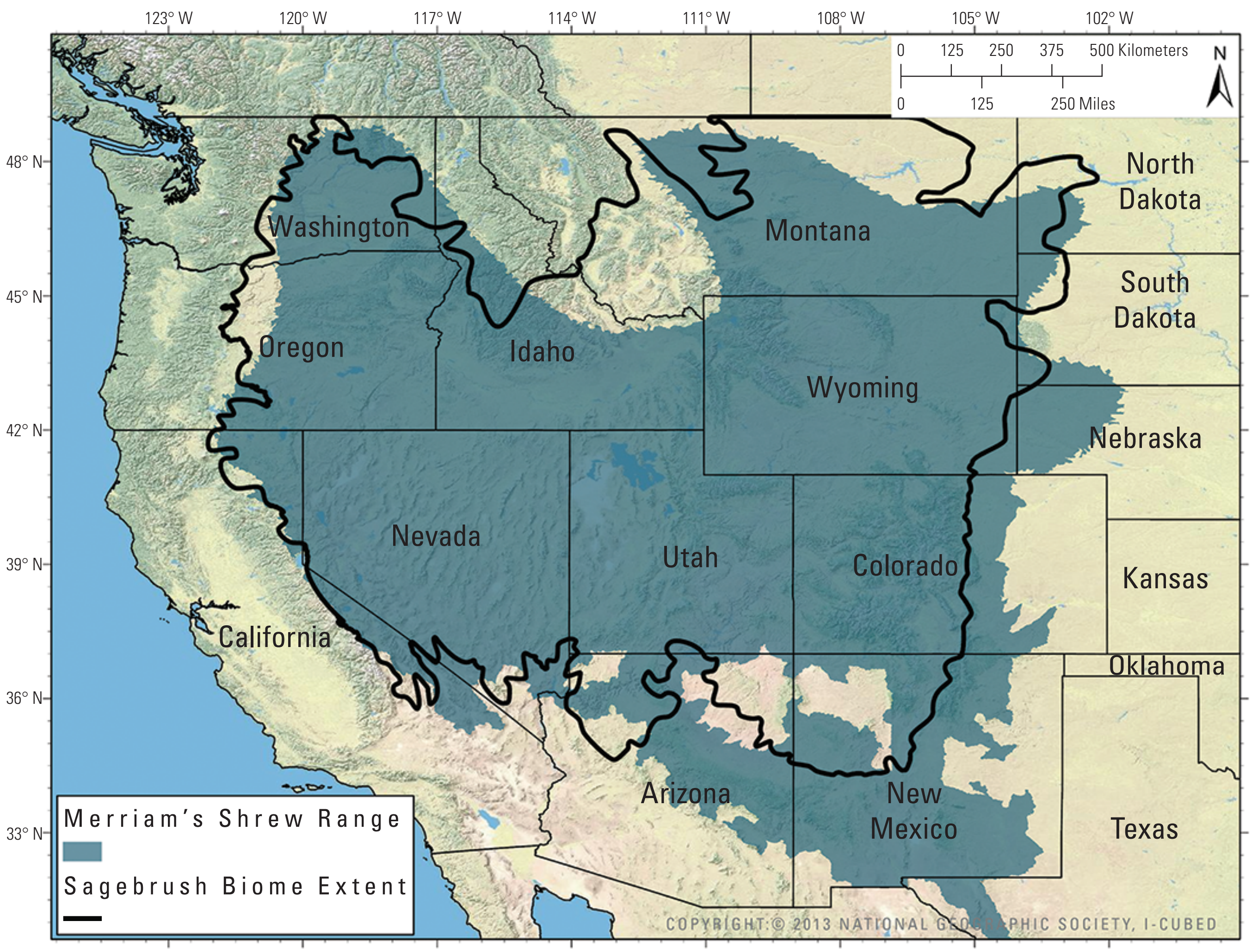

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Figure H3. Range of the Merriam's shrew (*Sorex merriami*) in the sagebrush (*Artemisia* spp.) biome. Data were developed and created by the U.S. Geological Survey (2013a).

Preble's Shrew

Taxonomy and Distribution

The Preble's shrew is one of the smallest and rarest (based on capture rates) North American mammals (Cornely and others, 1992; Buskirk, 2016; Montana Field Guide, 2016c). This species occurs in British Columbia, Colorado, Idaho, Montana, Nevada, New Mexico, Oregon, Washington, and Wyoming (Hoffmann and Fisher, 1978; Tomasi and Hoffmann, 1984; Williams, 1984; Stewardship Centre for British Columbia, 2016; International Union for Conservation of Nature, 2016; fig. H4). Populations appear to be disjunct, but this determination may simply reflect incomplete sampling (Cornely and others, 1992). Preble's shrews are often described as uncommon or rare (Kirkland and Findley, 1996; Buskirk, 2016), but this may reflect a lack of adequate sampling (Washington Department of Fish and Wildlife, 2015; International Union for Conservation of Nature, 2016). Neither population numbers nor area of occupied habitat are known (NatureServe, 2019). However, it is designated as "least concern" by the International Union for Conservation of Nature (2016) because the species is considered widespread with no evidence of declining populations.

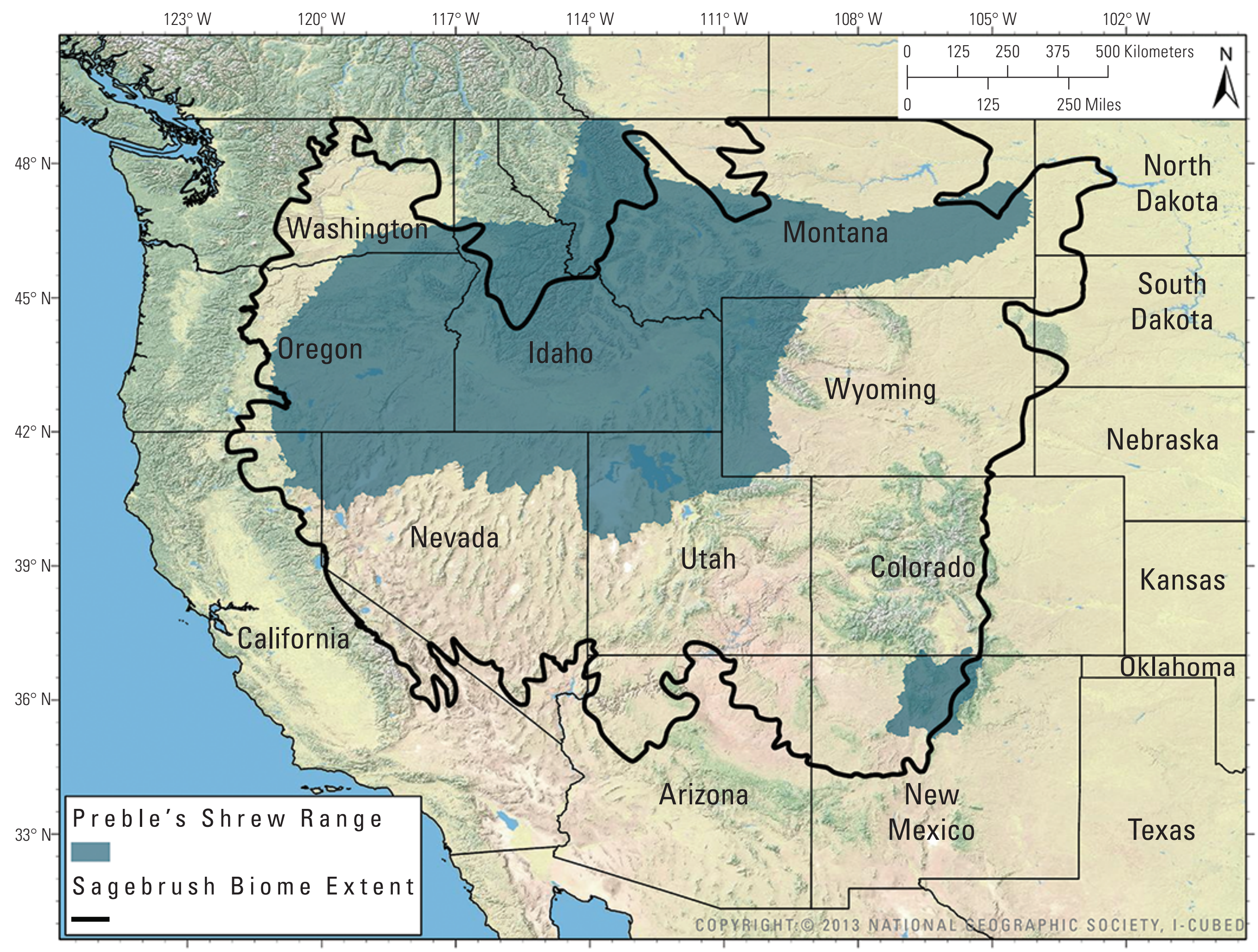

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Figure H4. Range of Preble's shrew (*Sorex preblei*) in the sagebrush (*Artemisia* spp.) biome. Data were developed and created by the U.S. Geological Survey (2013a).

Habitat Selection and Sagebrush Dependency

Preble's shrews are associated with arid shrub-steppe habitats (Kirkland and others, 1997; Demboski and Cook, 2003; Nevada Wildlife Action Plan Team, 2013; Washington Department of Fish and Wildlife, 2015). While often trapped in sagebrush areas, they have also been trapped in areas dominated by bunchgrasses, alkaline shrublands, salt desert shrublands (Tomasi and Hoffmann, 1984; Williams, 1984; Kirkland and others, 1997; Hendricks and Roedel, 2002; Demboski and Cook, 2003; Buskirk, 2016; Montana Field Guide, 2016c), ephemeral and perennial streams dominated by shrubs, willow (*Salix* spp.) fringed creeks and marshes (Nevada Wildlife Action Plan Team, 2013), and in dense lodgepole pine (*P. contorta*) forests in the Blue Mountains of Washington (Washington Department of Fish and Wildlife, 2015). Habitat characteristics that influence presence and abundance are unknown (Dobkin and Sauder, 2004), but Preble's shrews are considered a sagebrush near-obligate species.

Threats

Although the International Union for Conservation of Nature identifies no known threats to Preble's shrew, (International Union for Conservation of Nature; 2016) habitat loss to agricultural and urban development have been reported as threats to this species (British Columbia Conservation Data Centre, 2009). Some authors have suggested that activities that increase soil compaction, reduce the litter layer, and alter microhabitats (such as improper grazing, wildfire, mechanical treatments, application of herbicides, and establishment of exotic grasses) have the potential to adversely impact prey, and therefore Preble's shrews (Hendricks and Roedel, 2002; Dobkin and Sauder, 2004; Montana Field Guide, 2016c).

Management Considerations

There is insufficient information for this species on vital rates, distribution, and habitat use, which limits management recommendations (Colorado Parks and Wildlife, 2015; NatureServe, 2019; Montana Field Guide, 2016c). Suggested management actions are to minimize habitat alteration, preclude the establishment of invasive annual grasses, and to maintain a diversity of size and cover classes of sagebrush (Hendricks and Roedel, 2002; Dobkin and Sauder, 2004; Montana Field Guide, 2016c).

Ord's Kangaroo Rat

Taxonomy and Distribution

The Ord's kangaroo rat measures approximately 267 mm (10.5 in.), with the tail composing half or more of the total body length (Montana Field Guide, 2016d) and has external fur-lined cheek pouches. It primarily moves using all four feet (Buskirk, 2016) but also hops on its hind feet, with the tail acting as a rudder (Garrison and Best, 1990; Montana Field Guide, 2016d). The small forelegs are also used for manipulating food items (Sjoberg and others, 1984) and sifting sand to look for seeds (Clark and Stromberg, 1987). The species has 34 recognized subspecies (Garrison and Best, 1990). Ord's kangaroo rats are found in Alberta and Saskatchewan, Canada; Sonoran, Chihuahua, Coahuila, Nuevo Leon, Tamaulipas, Durango, Zacatecas, San Luis Potosi, Hidalgo, Guanajuato, and Queretaro, Mexico; and in Arizona, California, Colorado, Idaho, Kansas, Montana, Nebraska, Nevada, New Mexico, North Dakota, Oklahoma, Oregon, South Dakota, Texas, Utah, Washington, and Wyoming, United States (fig. H5; Garrison and Best 1990; Gitzen and others, 2001; Buskirk 2016). Little information regarding population trends is available, but the International Union for Conservation of Nature (2016) has concluded that the species is very abundant, and trends are stable.

Habitat Selection and Sagebrush Dependency

The Ord's kangaroo rat occurs in areas with sandy or fine-textured soils below the lower elevational limit of conifers (Buskirk 2016). Habitat associations vary across the species' range and include big sagebrush, pinyon-juniper, four-wing saltbush (*Atriplex canescens*), greasewood, and yucca (*Yucca* spp.) sagebrush-shortgrass mixtures (Garrison and Best, 1990; Buskirk, 2016). Soil type and not vegetation appears to be the primary factor in habitat selection (Garrison and Best, 1990), but the species is sagebrush dependent.

Threats

Given its abundance and wide distribution, this species is thought to be secure (Buskirk, 2016; International Union for Conservation of Nature, 2016). Improper grazing, particularly when coupled with drought, has been implicated in both negatively and positively affecting the species (Sjoberg and others, 1984). In Mexico, Ord's kangaroo rat is more common in areas of low human habitation (2016; International Union for Conservation of Nature, 2016).

Sagebrush Vole

Taxonomy and Distribution

Sagebrush voles are small, short-tailed voles typically weighing less than 28 grams (g; 1 ounce [oz]; Carroll and Genoways, 1980; Boyle and Reeder, 2005; Buskirk, 2016; Montana Field Guide, 2016e). There are six recognized subspecies (Boyle and Reeder, 2005; Colorado Parks and Wildlife, 2015; Buskirk, 2016). The range of the sagebrush vole overlaps the distribution of sagebrush ecosystems and the species is found in Alberta and Saskatchewan, Canada, and in California, Colorado, Idaho, Montana, North Dakota, Nevada, Oregon, South Dakota, Utah, Washington, and Wyoming in the United States (fig. H6; Birney and Lampe, 1972; Carroll and Genoways, 1980; Dobkin and Sauder, 2004; International Union for Conservation of Nature, 2016).

Sagebrush vole population trends are poorly known (Dobkin and Sauder, 2004; Colorado Parks and Wildlife, 2015), but the species is generally considered abundant (Nevada Wildlife Action Plan Team, 2013). However, local numbers may cycle, increasing in response to favorable weather and associated food availability and declining in response to extremely hot periods, drought, or disease (Boyle and Reeder, 2005). Abundances are typically higher in shrub-steppe areas with native bunchgrass understories (Dobkin and Sauder, 2004).

Habitat Selection and Sagebrush Dependency

Because of their close association with sagebrush and dependence on sagebrush plants for winter forage (Boyle and Reeder, 2005), this species is considered a sagebrush obligate (Nevada Wildlife Action Plan Team, 2013; California Department of Fish and Wildlife, 2015; Colorado Parks and Wildlife, 2015;

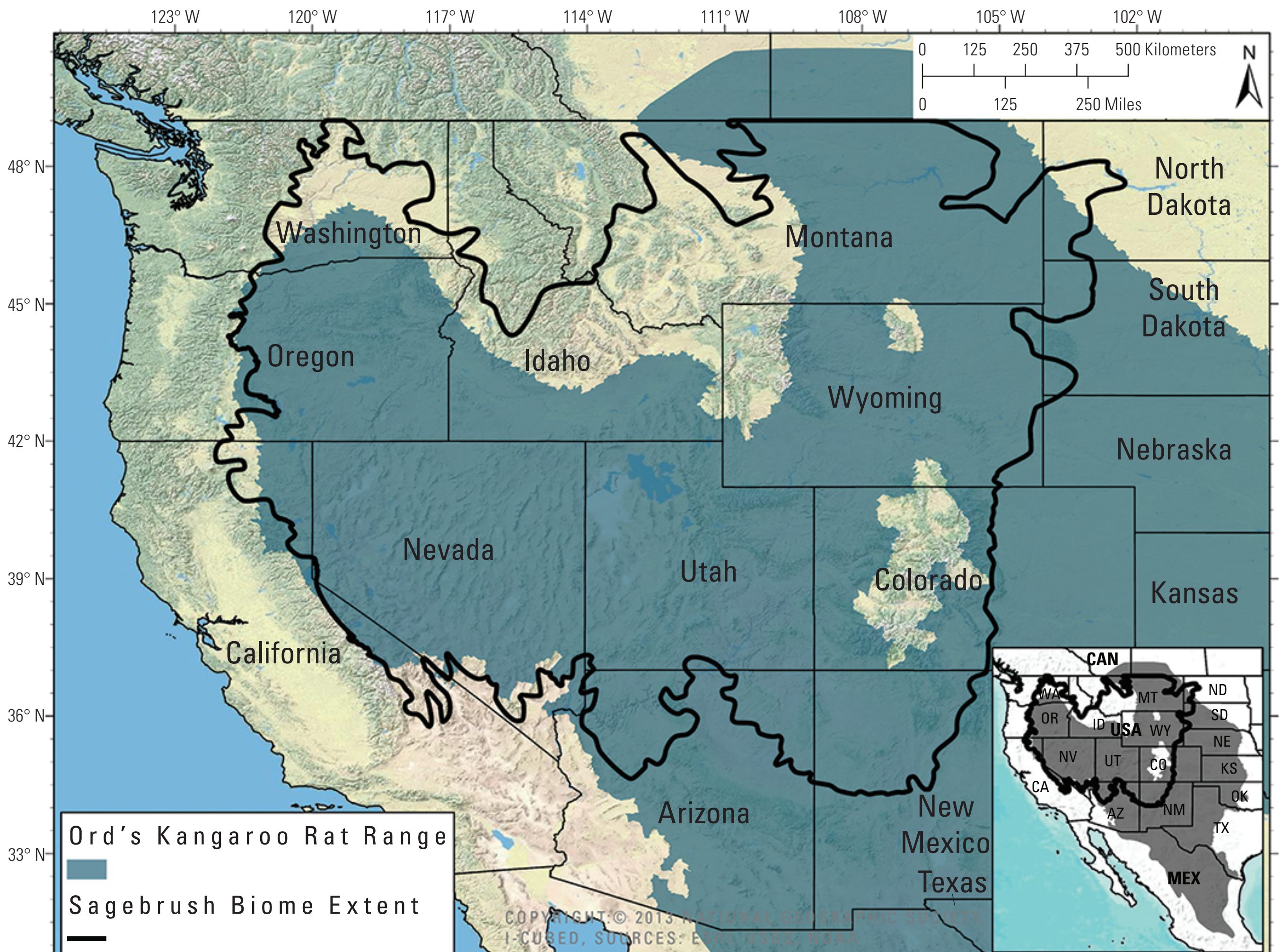

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Figure H5. Range of Ord's kangaroo rat (*Dipodomys ordii*) in the sagebrush (*Artemisia* spp.) biome. Data were developed and created by the U.S. Geological Survey (2013a), within the U.S. boundary, and by The International Union for Conservation of Nature (2016), outside the U.S. boundary.

Dyke and others, 2015). They typically occur in arid areas with well-drained soils where sagebrush provides the dominant shrub cover, although rabbitbrush may also be present (Carroll and Genoways, 1980; Dobkin and Sauder, 2004). Sagebrush voles use a wide range of habitat structures (for example, shrub densities, heights), but little information is available regarding how these variances influence the presence and abundance of the species (Dobkin and Sauder, 2004; Boyle and Reeder, 2005).

Threats

Activities that reduce or degrade sagebrush cover, including agricultural conversion, frequent fire, pinyon-juniper incursion, energy development, presence of invasive annual grasses, and range improvement projects can result in population declines (Dobkin and Sauder, 2004; Boyle and Reeder, 2005; Nevada Wildlife Action Plan Team, 2013; Colorado Parks and Wildlife, 2015; Buskirk, 2016). Limited information suggests that livestock grazing may negatively affect populations of sagebrush voles through soil compaction and competition for forage (Dobkin and Sauder, 2004; Boyle and Reeder, 2005). Climate change, as it affects sagebrush habitats, may also affect sagebrush voles (Nevada Wildlife Action Plan Team, 2013).

Management Considerations

Management activities to conserve intact sagebrush landscapes are presumed to benefit sagebrush voles (Dobkin and Sauder, 2004; Boyle and Reeder, 2005; Nevada Wildlife Action Plan Team, 2013; Dyke and others, 2015). Additional research is needed to understand impacts of habitat degradation on sagebrush vole abundance, distribution, and persistence. Data are also lacking regarding population cycling (Boyle and Reeder, 2005).

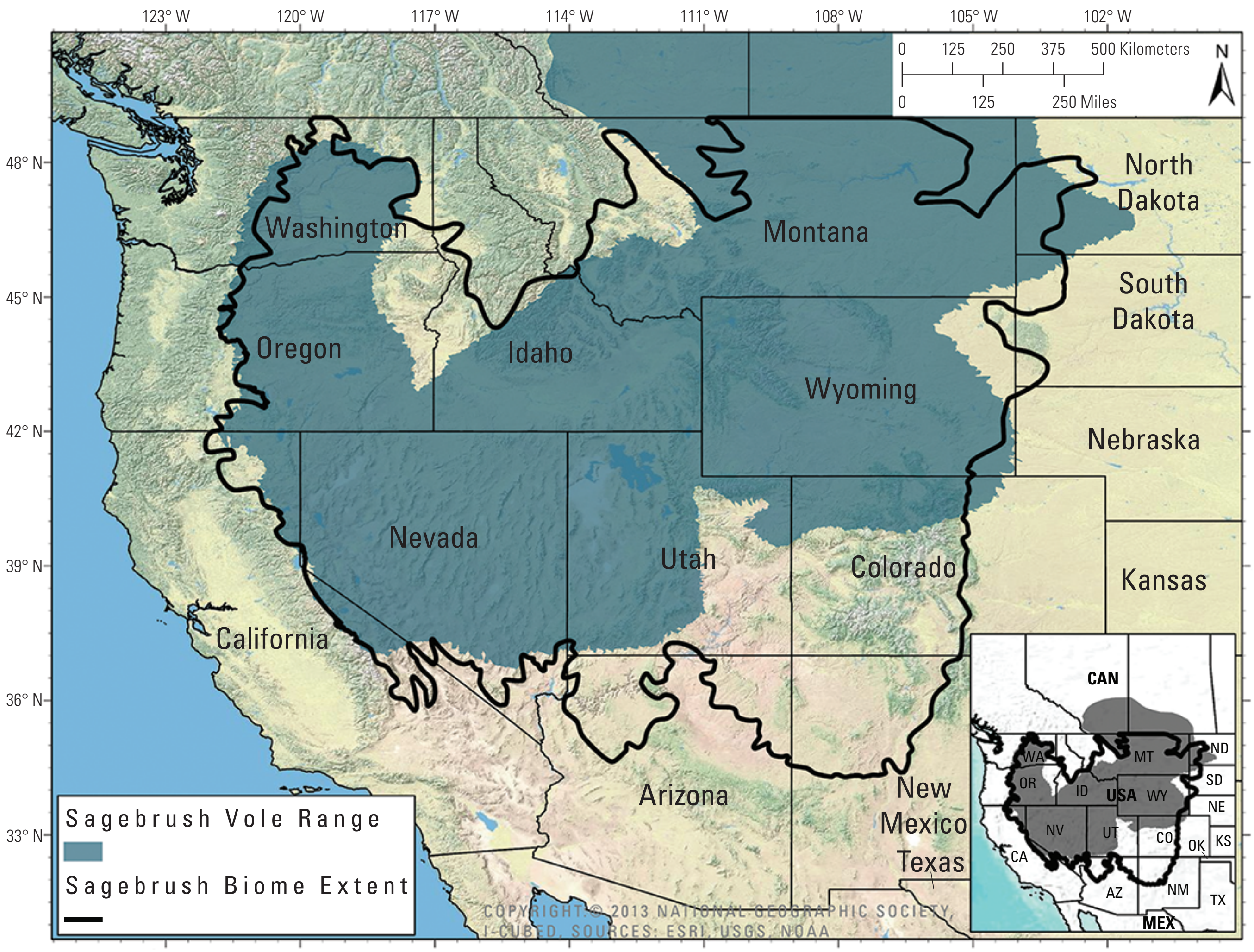

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Figure H6. Range of the sagebrush vole (*Lemmiscus curatus*) in the sagebrush (*Artemisia* spp.) biome. Data were developed and created by the U.S. Geological Survey (2013a), within the U.S. boundary, and by The International Union for Conservation of Nature (2016), outside the United States boundary.

Southern Idaho Ground Squirrel

Taxonomy and Distribution

Southern Idaho ground squirrels are a medium size ground squirrel with pelage coloration associated with the soil color throughout the species' range (Yensen and Sherman, 1997). This species was previously a subspecies of the Idaho ground squirrel (*U. brunneus*), along with the Northern Idaho ground squirrel (*U. b. brunneus*; Yensen, 1991). However, the two species were recently separated based on morphological and genetic analyses (Hoisington-Lopez and others, 2012; NatureServe, 2019).

The Southern Idaho ground squirrel is endemic to four counties in southwest Idaho, with a total known range of approximately 290,693 ha (718,318 acres; fig. H7; Lohr and others, 2013; U.S. Department of the Interior, 2015b) at elevations between 671 and 1,097 meters (m; 2,200 and 3,600 feet [ft]; State of Idaho, 2016). The northern part of the species' historical range is no longer occupied (U.S. Department of the Interior, 2015b; Lohr and Haak, 2009). The species is geographically contained by rivers to the south and west and by lava beds on the northeast (Yensen, 1991).

Population studies of the Southern Idaho ground squirrels in 1985 estimated 40,000 individuals (U.S. Department of the Interior, 2015b, and references therein; State of Idaho, 2016). Numbers declined in the late 1990s, and population estimates in 2001 were approximately 2,000 to 4,500 individuals, an estimated decline of 90 percent (Lohr and others, 2013; State of Idaho, 2016). Current local population distribution and abundance is unknown. Habitat loss, fragmentation, and degradation have caused the remaining populations of Southern Idaho ground squirrels to become discontinuously distributed (Yensen, 1991; Garner and others, 2005). Population size estimation is complicated by uneven sampling efforts (State of Idaho, 2016).

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Figure H7. Range of the Southern Idaho ground squirrel (*Urocitellus endemicus*) in the sagebrush (*Artemisia* spp.) biome. Entire range of the species is delineated in the red box, which is expanded in the inset. Data were developed and created by the U.S. Geological Survey (2013a).

Habitat Selection and Sagebrush Dependency

The Southern Idaho ground squirrel primarily occurs in lower elevation sandy soils (Yensen and Sherman, 2003), dominated by big sagebrush and antelope bitterbrush (*Purshia tridentata*) with a native forb understory (U.S. Department of the Interior, 2015b; Hoisington-Lopez and others, 2012). Squirrels have also been observed using agricultural fields, fence lines, and haystacks (U.S. Department of the Interior, 2005b and references therein). Southern Idaho ground squirrels were observed on a golf course and a nearby cemetery during surveys conducted by the Idaho Department of Game and Fish (U.S. Department of the Interior, 2005b). These diverse usages indicate that the species seems to be adaptable to altered landscapes and nonnative annual vegetation; however, those habitats do not provide sufficient food resources to allow the squirrels to survive hibernation (U.S. Department of the Interior, 2005b and references therein). These altered landscapes may serve as population sinks (U.S. Department of the Interior, 2005b and references therein). Southern Idaho ground squirrels are considered sagebrush dependent.

Threats

The primary threats to the Southern Idaho ground squirrel are the loss and fragmentation of habitat caused by agricultural activities and habitat degradation from the invasion of exotic annual grasses, loss of shrubs, and the resulting changes in the wildfire regime (International Union for Conservation of Nature, 2016; Idaho Department of Fish and Game, 2017; NatureServe, 2019). Recreational shooting and poisoning appeared to have contributed to past population declines, but regulatory changes have reduced this threat (International Union for Conservation of Nature, 2016). Disease and predation do not appear to be a limiting factor for this species (U.S. Department of the Interior, 2005b; NatureServe, 2019), although they may have disproportionate effects in small populations.

Management Considerations

Management and restoration of habitats is key to long-term conservation (Idaho Department of Fish and Game, 2017). Additional research is needed to inform conservation efforts, including better understanding of life history and reproductive biology (International Union for Conservation of Nature, 2016). Low genetic diversity in small and peripheral Southern Idaho ground squirrel populations may require direct management to resolve, including translocations or captive breeding (Garner and others, 2005; Lohr and others, 2013).

Wyoming Ground Squirrel

Taxonomy and Distribution

Wyoming ground squirrel is a medium-sized ground squirrel, with a relatively long tail and large ears (Burnett, 1920; Buskirk, 2016; Nevada Department of Wildlife, 2016; Idaho Department of Fish and Game, 2017). Three subspecies of Wyoming ground squirrels are currently recognized, each a disjunct population (fig. H8; Buskirk, 2016; International Union for Conservation of Nature, 2016; Nevada Department of Wildlife, 2016; State of Idaho, 2016; see range description below). Subspecies *Urocitellus elegans elegans* occurs primarily in Wyoming, but also in adjacent areas of Colorado, Idaho, and Utah (Helgen and others, 2009; Buskirk, 2016; International Union for Conservation of Nature, 2016). Subspecies *U. e. aureus* is found in southwestern Montana and adjoining areas of Idaho (Helgen and others, 2009; International Union for Conservation of Nature, 2016), and *U. e. nevadensis* occurs in southwestern Idaho, north-central Nevada, and possibly southeastern Oregon (Helgen and others, 2009; International Union for Conservation of Nature, 2016). The distribution of this subspecies in Idaho is limited to one population, and the subspecies appears to be extinct in Oregon (International Union for Conservation of Nature, 2016). These subspecies are believed to be the remaining peripheral populations of a much more widely distributed species, whose core populations were lost because of habitat changes in the Pleistocene (Zegers, 1984).

The Wyoming ground squirrel is widespread and abundant in two of the three disjunct populations units (Wyoming, Montana) and can reach high densities in local areas (Zegers, 1984; Buskirk, 2016; International Union for Conservation of Nature, 2016; NatureServe, 2019). However, population trends are unknown (International Union for Conservation of Nature, 2016). In Wyoming, the subspecies (*U. e. elegans*) is abundant with little conservation risk (Buskirk, 2016). In Idaho, the population size of the subspecies (*U. e. nevadensis*) is unknown (Idaho Department of Fish and Game, 2017), but they are now extirpated from several areas where they were reportedly once abundant (State of Idaho, 2016). Only one extant population is known in Idaho (State of Idaho, 2016). No information could be located regarding the population abundance or trend for *U. e. aureus* in Montana and adjacent States.

Habitat Selection and Sagebrush Dependence

The Wyoming ground squirrel primarily occurs in dry grasslands or shrub-steppe habitats, particularly sagebrush (International Union for Conservation of Nature, 2016). They are sagebrush near-obligates, although they can also occur in subalpine talus slopes, montane meadows, reclaimed surface mines, along the edges of cultivated fields, and in railroad embankments and livestock pastures (Zegers, 1984 and references therein; Nevada Department of Wildlife, 2016; State of Idaho, 2016). The species prefers open, grassy areas over areas with dense shrub cover (Johnson and others, 1996). In some areas, local distribution may be limited by interspecific competition versus vegetative conditions (State of Idaho, 2016).

Threats

Populations of *U. e. nevadensis* in Idaho are affected by the loss and degradation of sagebrush habitats, particularly as related to invasive plants and altered wildfire regimes (Idaho Department of Fish and Game, 2017). Agricultural and residential development may also be factors affecting the distribution and density of Wyoming ground squirrels (International Union for Conservation of Nature, 2016; State of Idaho, 2016). Changes in the structure and composition of shrub-dominated habitats caused by livestock grazing may also affect Wyoming ground squirrels (State of Idaho, 2016), although research is lacking. In contrast, research on chemical sagebrush thinning in Wyoming found no significant differences in the abundance of Wyoming ground squirrels between treatments, although they were captured more frequently in heavily thinned areas (Johnson and others, 1996). Wyoming ground squirrels are often poisoned to reduce crop damage (Buskirk, 2016; International Union for Conservation of Nature, 2016; State of Idaho, 2016). Recreational shooting may also be a limiting factor in small populations (State of Idaho, 2016).

Management Considerations

Additional monitoring efforts are needed to fully characterize the distribution and status of the species, particularly for *U. e. nevadensis* (State of Idaho, 2016; Idaho Department of Fish and Game, 2017; NatureServe, 2019). Habitat protection and restoration may be necessary where populations of *U. e. nevadensis* are small or declining (State of Idaho, 2016). Enforcement limiting recreational shooting may also be helpful (State of Idaho, 2016).

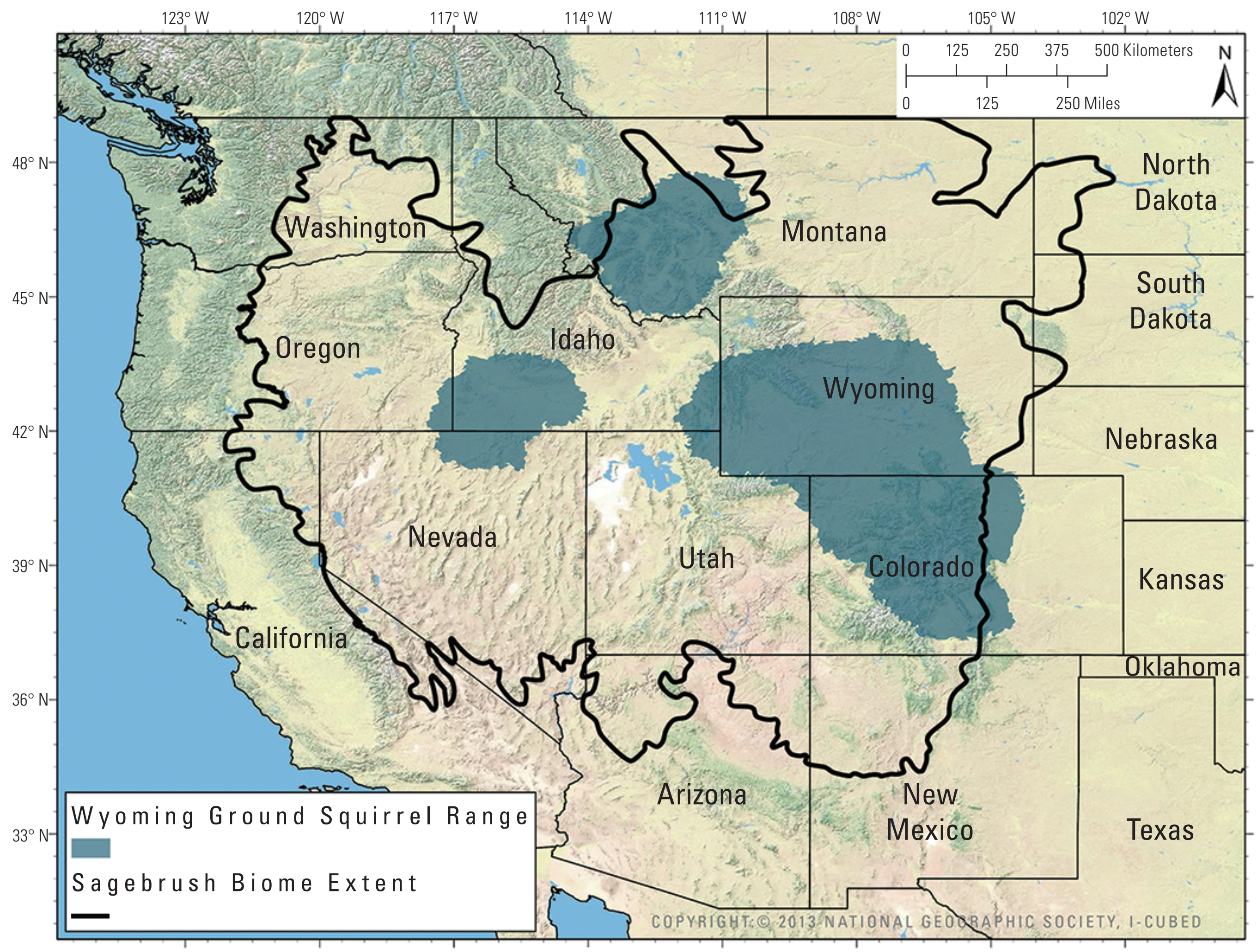

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Figure H8. Range of Wyoming ground squirrel (*Urocitellus elegans*) in the sagebrush (*Artemisia* spp.) biome. Data were developed and created by the U.S. Geological Survey (2013a).

Wyoming Pocket Gopher

Taxonomy and Distribution

The Wyoming pocket gopher is a powerful digger, strongly adapted for fossorial living with small eyes and ears, strong front limbs with long nails, and fur-lined cheek pouches (Keinath and Beauvais, 2006). It is one of four species of pocket gopher within Wyoming and can be distinguished in the field from the northern pocket gopher (*T. talpoides*) where their ranges overlap (Keinath and others, 2014). This species is endemic to two counties in south-central Wyoming (fig. H9) and its entire global range is within a small part of the sagebrush biome (Clark and Stromberg, 1987; Keinath and Beauvais, 2006).

There is no available information on the abundance of the Wyoming pocket gopher (Keinath and Beauvais, 2006; Wyoming Game and Fish Department, 2017). Extensive trapping efforts suggest the species is uncommon (Keinath and Beauvais, 2006; Keinath and others, 2014). There is insufficient information to determine population trends, but Wyoming pocket gophers may be declining based on their absence from known historical locations (Keinath and Beauvais, 2006) and minimal dispersal capabilities. Long-distance movement and dispersal capabilities of all pocket gophers are limited (Verts and Carraway, 1999).

Habitat Selection and Sagebrush Dependence

All pocket gophers require soils stable enough to hold burrow systems and herbaceous plants for food (Keinath and Beauvais, 2006). Wyoming pocket gophers seem to be most reliably trapped in small islands of low vegetation within a sagebrush matrix (Keinath and Griscom, 2009), and possibly limited to areas of Gardner's saltbush (*A. gardneri*; Keinath and others, 2014; Wyoming Game and Fish Department, 2017). Historical trapping locations include greasewood communities based on information collected from specimen

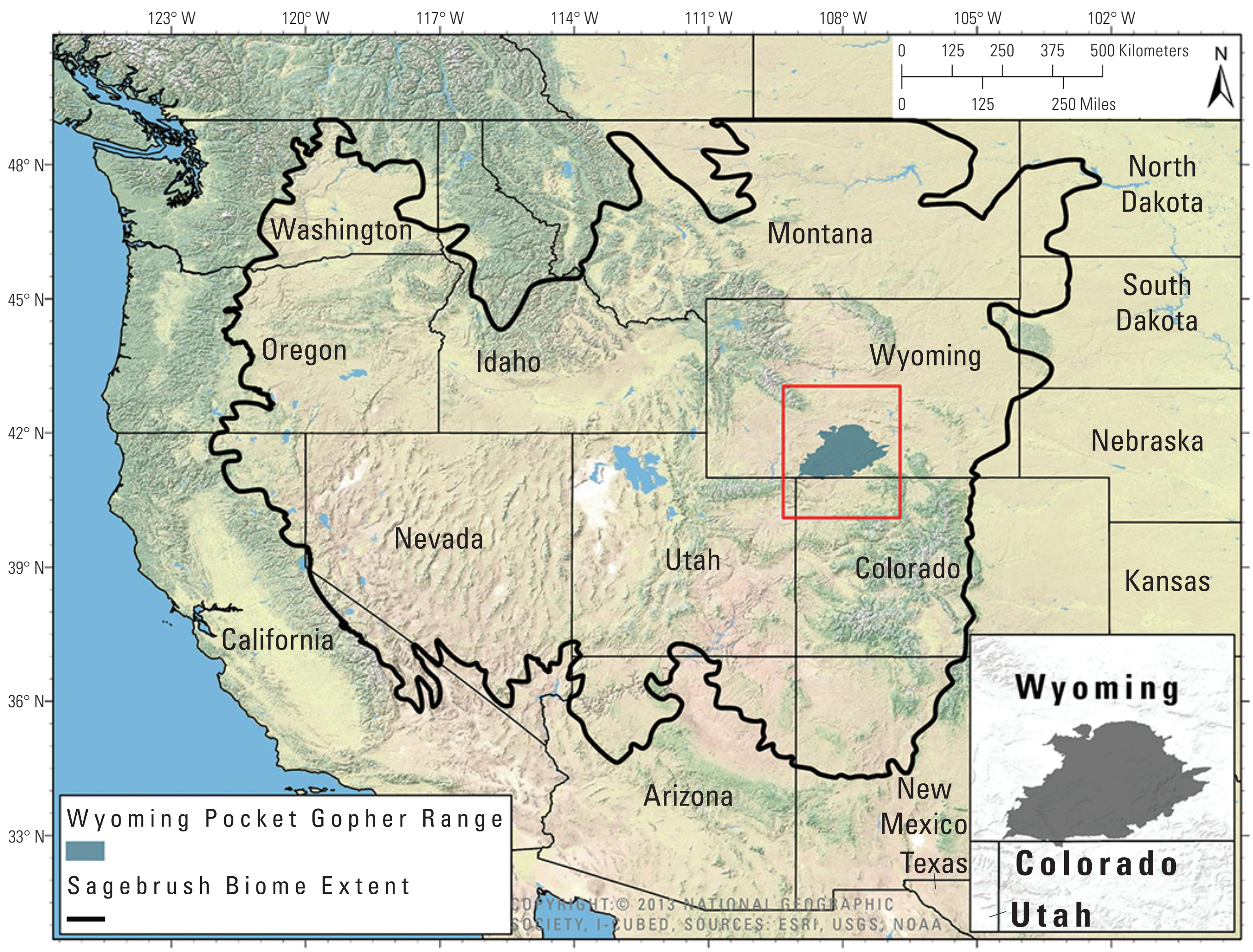

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Figure H9. Range of the Wyoming pocket gopher (*Thomomys clusius*) in the sagebrush (*Artemisia* spp.) biome. Range of the species is delineated in the red box, which is expanded in the inset. Data were developed and created by the U.S. Geological Survey (2013a).

tags (Keinath and Beauvais, 2006), but those results could not be duplicated in more recent trapping efforts (Keinath and Griscom, 2009). Because its entire range is encompassed within the sagebrush biome, the Wyoming pocket gopher is considered a sagebrush near-obligate.

Threats

Wyoming pocket gophers are assumed to be sensitive to threats facing other species of pocket gophers, including intensive livestock grazing, pest control (including direct control), habitat loss from agricultural practices, reduced forage resulting from herbicide application, and any activities that disturb or compact the soil. Energy exploration and extraction may impact Wyoming pocket gophers (Keinath and Beauvais, 2006; Wyoming Game and Fish Department, 2017).

Management Considerations

Determining the extent of this species range and specific habitat requirements is needed for effective management (Keinath and Beauvais, 2006; Keinath and others, 2014; Wyoming Game and Fish Department, 2017). Known occupied areas should be considered for protection from disturbances, such as grazing, petroleum exploration and extraction activities, and vegetation removal (Keinath and Beauvais, 2006; Keinath and others, 2014). Similar to other fossorial animals, the locations of gopher colonies likely shift over time, making conservation of potential but currently unoccupied habitat surrounding areas of occupation necessary to support long-term persistence.

White-Tailed Prairie Dog

Taxonomy and Distribution

White-tailed prairie dogs are social, burrowing ground squirrels (Keinath, 2004; Buskirk, 2016) that dig their own burrow complexes in deep, well-drained soils (Seglund and others, 2006; U.S. Department of the Interior, 2010c). They occur in Wyoming, eastern Utah, Colorado, and southern Montana (fig. H10). Most of the species range falls within Wyoming (Keinath, 2004). However, within the range, habitat suitability is limited, making the actual distribution of this species difficult to determine (Keinath, 2004).

A lack of historical population information and inconsistencies in survey methodologies limits analyses of population trends (Seglund and others, 2006; U.S. Department of the Interior, 2010c). States' monitoring results show variation in rates of colony occupancy and in population numbers in a colony (U.S. Department of the Interior, 2010c). Sampling methods may both under- and overestimate colony occupancy (Keinath, 2004) and therefore may affect analyses of population trends. A summary of population survey efforts found that white-tailed prairie dog populations are likely below historical numbers (U.S. Department of the Interior, 2010c), although there is disagreement regarding the changes in the extent of their overall historical distribution (see Keinath, 2004 and Buskirk, 2016). The species is classified by the International Union for Conservation of Nature (2016) as least concern because it is relatively widespread and still occurs throughout most of its historical range, although colony size and distribution are much reduced.

Habitat Selection and Sagebrush Dependency

White-tailed prairie dogs typically occur in higher-elevation grasslands with abundant shrub cover (Keinath, 2004) and, while they prefer areas with lower vegetation heights (Seglund and others, 2006), they may use dense vegetation within sagebrush habitats to hide from predators (U.S. Department of the Interior, 2010c). Colonies have also been documented in saltbush-dominated areas associated with fine-textured soils. Understory vegetation is typically composed of native grasses and forbs, but colonies in Colorado and Utah often have invasive annual grasses such as cheatgrass as a common understory component (Seglund and others, 2006). Wide variances in total vegetative canopy cover (from 10 to 70 percent) have been observed between and within colonies (Tileston and Lechleitner, 1966; Menkens and others, 1987). White-tailed prairie dogs are considered a sagebrush-dependent species.

Threats

The primary threat to white-tailed prairie dogs is the nonnative sylvatic plague (*Yersinia pestis;* Keinath, 2004; Seglund and others, 2006; U.S. Department of the Interior, 2010c) and plague-free populations are unknown (Biggins and Kosoy, 2001). The long-term effect of this disease on the viability of prairie dogs is unknown (Seglund and others, 2006; U.S. Department of the Interior, 2010c), but reductions in fertility rates have been suggested as a possible outcome (Keinath, 2004).

Determining historical impacts from activities such as agricultural land conversion and urbanization is difficult because of the lack of historic distribution and abundance information for prairie dogs (Seglund and others, 2006). Direct habitat loss from these activities has occurred, and continues to occur, but the extent of impacts from urbanization is unknown (U.S. Department of the Interior, 2010c). Indirect impacts from these activities, such as poisoning, increased numbers of domestic pets, increased human access to recreational activities, and increased stress from human presence may have significant effects on prairie dog occurrence and abundance adjacent to these areas (U.S. Department of the Interior, 2010c; Colorado Parks and Wildlife, 2015). Continued habitat loss and fragmentation from agricultural conversion is likely minimal simply because of the lack of arable lands for crop production (U.S. Department of the Interior, 2010c).

Habitat loss from oil and gas exploration and development does occur but is likely not a significant factor because large colonies are protected for the purpose of black-footed ferret (*Mustela nigripes*) reintroduction (Seglund and others, 2006). Impacts from energy development—habitat loss and fragmentation and noise—likely have had negative effects on white-tailed prairie dogs, including mortality (U.S.

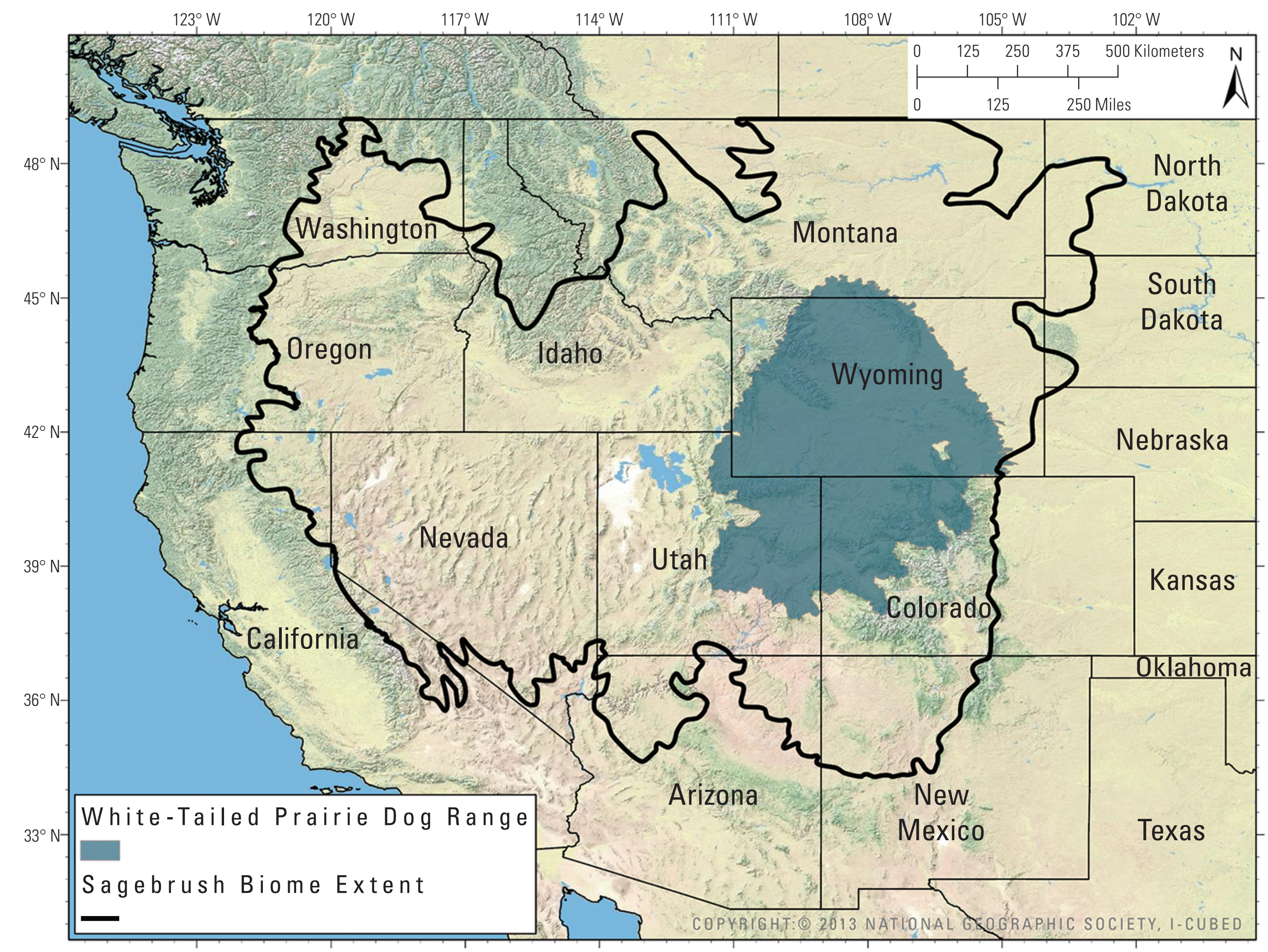

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Figure H10. Range of the white-tailed prairie dog (*Cynomys leucurus*) in the sagebrush (*Artemisia* spp.) biome. Data were developed and created by the U.S. Geological Survey (2013a).

Department of the Interior, 2010c). Development of wind energy has similar potential to impact white-tailed prairie dogs (U.S. Department of the Interior, 2010c).

Livestock grazing could potentially impact white-tailed prairie dogs through soil compaction, changes in plant species composition (particularly reductions during key foraging periods such as juvenile emergence from burrows), and introduction of nonnative annual grasses (Keinath, 2004; Seglund and others, 2006; U.S. Department of the Interior, 2010c). Prolonged grazing during periods of drought may impact prairie dogs if it results in vegetation alteration or alteration of ecosystem structure (Seglund and others, 2006).

Recreational shooting has been demonstrated to reduce fitness and alter social behavior in black-tailed prairie dogs (*C. ludovicianus*) and in some cases has led to colony extirpation (Keinath, 2004; Seglund and others, 2006; U.S. Department of the Interior, 2010c). Similar studies have not been conducted on white-tailed prairie dog colonies, but many authors have suggested the impacts would be similar (Seglund and others, 2006, and references therein). In many areas, shooting impacts have been minimized by State, land management agency, or local regulations (Seglund and others, 2006). Poisoning to reduce conflicts between agricultural users and prairie dogs is also common (Seglund and others, 2006). Although invasive annual grasses have been documented in extant colonies, early curing does not provide late-season nutrition, potentially decreasing the ability of prairie dogs to develop sufficient fat reserves to survive hibernation (Keinath, 2004; Colorado Parks and Wildlife, 2015).

Management Considerations

Although white-tailed prairie dogs may not be sagebrush obligates, they do depend on sagebrush habitats across their range. Prairie dogs also depend on emigration to re-establish colonies affected by plague, which requires retention of habitat corridors (Keinath, 2004). Therefore, conservation of large, undisturbed tracts is essential for the long-term persistence of this species (Keinath, 2004; Montana Fish, Wildlife and Parks, 2015). Shooting and poisoning activities should be considered for increased restriction until research can determine levels of these activities that do not affect long-term viability of the white-tailed prairie dog (Keinath, 2004; Montana Fish, Wildlife and Parks, 2015). Altered wildfire regimes may be beneficial to prairie dogs by reducing shrub density and stimulating growth of forage species (Colorado Parks and Wildlife, 2015). However, the incursion of nonnative annual grasses, often associated with altered wildfire regimes, may negate the beneficial impact of fire (Colorado Parks and Wildlife, 2015).

Black-Tailed Jackrabbit

Taxonomy and Distribution

Like other jackrabbits (*Lepus* spp.), the black-tailed jackrabbit has characteristic long ears and long hind legs. The distinctive tail is gray to white with a black median-dorsal stripe (Orr, 1940; Whitaker and Hamilton, 1998), and the remaining pelage is dark buff with white undersides (Corbet, 1983; Hoffmeister, 1986). The ears are black tipped on the outer surfaces and unpigmented inside. There is considerable variation in coloration among subspecies, which is believed to be indicative of corresponding changes in climatic conditions (Nelson, 1909) and substrate coloration throughout the species' range (Baker, 1960).

The black-tailed jackrabbit is the most widely distributed jackrabbit in North America, occurring throughout the Great Basin as well as much of western North America (fig. H11). Black-tailed jackrabbits are found in central Washington, extending east to Missouri, and south to Hidalgo and Queretaro, Mexico (Best, 1996). Black-tailed jackrabbit distribution is currently expanding eastward in the Great Plains at the expense of white-tailed jackrabbit (*L. townsendii*; Flux, 1983; Jones and others, 1983). Few data are available to assess the population status of the black-tailed jackrabbit across the sagebrush biome. In the northern Great Basin, populations of black-tailed jackrabbits are cyclic, reaching high densities at approximately 10-year intervals (Gross and others, 1974; Johnson and Peek, 1984; Bartel and others, 2008).

Habitat Selection and Sagebrush Dependence

The black-tailed jackrabbit is a generalist species that occupies plant communities with a mixture of shrubs, grasses, and forbs for food, and shrubs or small trees for cover (Johnson and Anderson, 1984). It prefers moderately open areas without dense understory growth and is rarely found in closed-canopy habitats. Shrubland-herbaceous mosaics are preferred over pure stands of shrubs or herbaceous vegetation. Black-tailed jackrabbits are common in sagebrush (Nydegger and Smith, 1986), creosote bush (*Larrea tridentata*; Mares and Hulse, 1977), and other desert shrublands; palouse, shortgrass, and mixed-grass prairies; desert grassland; open-canopy chaparral; oak (*Quercus* spp.; Hall and others, 1992) and pinyon-juniper woodlands (Dunn and others, 1982); and early seral and low- to mid-elevation coniferous forests (Giusti and others, 1992). It is also common in and near croplands, especially alfalfa (*Medicago sativa*) fields (Dunn and others, 1982).

Black-tailed jackrabbits require shrubs or small conifers for hiding, nesting, and thermal cover, and grassy areas for night feeding (Dunn and others, 1982; Johnson and Anderson, 1984). Small shrubs (such as winterfat or shadscale) do not provide adequate cover (Johnson and Anderson, 1984; Alipayou and others, 1993). Components of diet are variable among locations and seasons (Dunn and others, 1982); however, in the Great Basin, big sagebrush is a primary forage species and is used throughout the year (Anderson and Shumar, 1986; Fagerstone and others, 1980). The black-tailed jackrabbit is a sagebrush-dependent species.

Threats

No threats have been documented for this species. However, the quality and abundance of black-tailed jackrabbit habitat have declined within sagebrush communities across the Intermountain West and Great Basin because of invasive plant species, such as cheatgrass, and subsequently altered fire regimes (Knick and Dyer, 1997; Simes and others, 2015). Additionally, black-tailed jackrabbits have been perceived as an agricultural threat, and eradication efforts (such as bounties, rabbit drives, and poisoning) to control jackrabbit populations were common in several States throughout the 19th and early 20th centuries (Simes and others, 2015).

Management Considerations

Despite their abundance and widespread distribution, the black-tailed jackrabbit remains understudied (Smith and others, 2002; Simes and others, 2015), and no special management activities have been developed or implemented for this species.

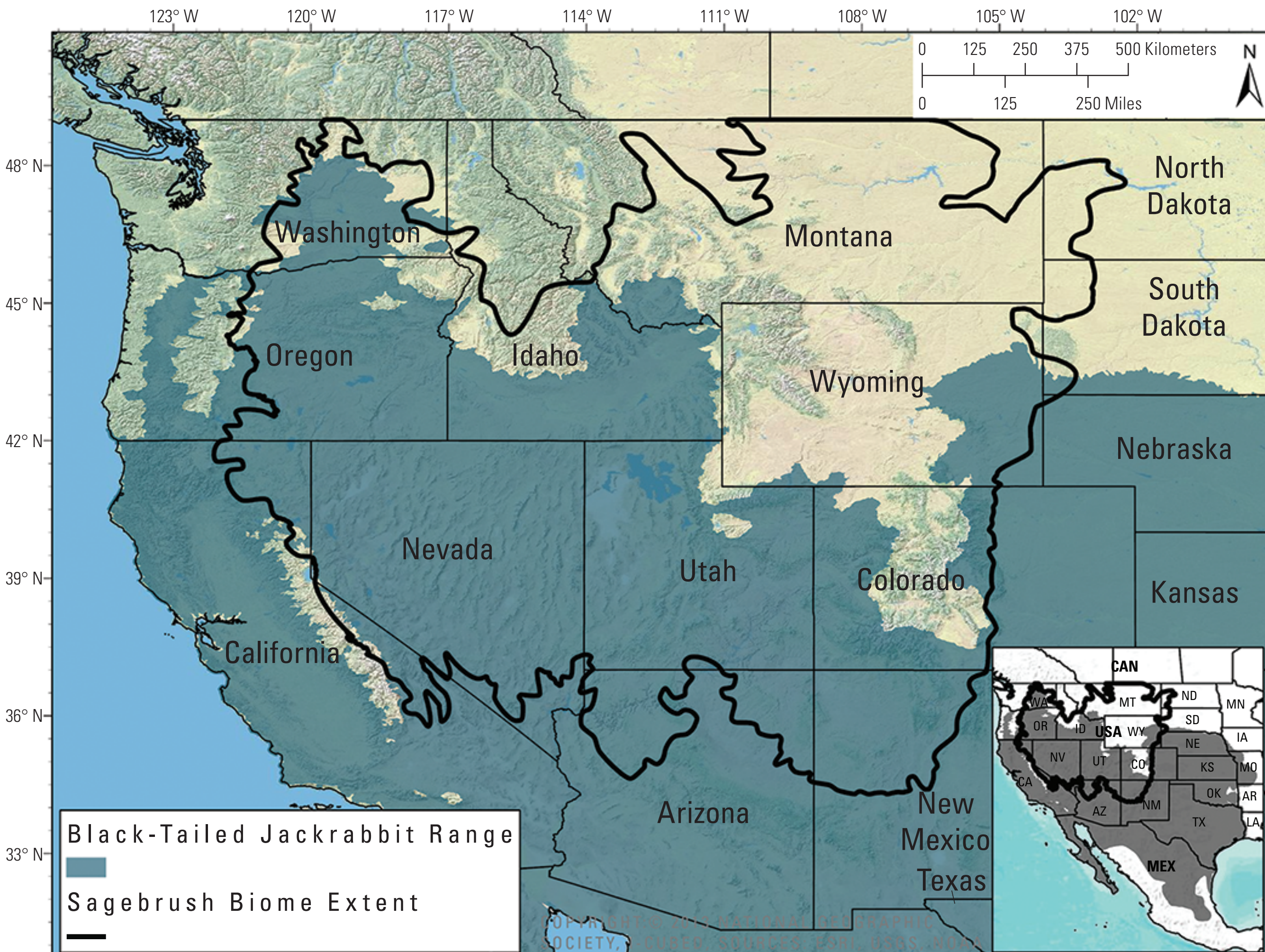

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Figure H11. Range of the black-tailed jackrabbit (*Lepus californicus*) in relation to the sagebrush (*Artemisia* spp.) biome. Data were developed and created by the U.S. Geological Survey (2013a), within the U.S. boundary, and The International Union for Conservation of Nature (2016), outside the U.S. boundary.

Chapter I. Amphibians and Reptiles in Sagebrush

By David S. Pilliod[1] and Michelle I. Jeffries[1]

Executive Summary

Amphibians and reptiles are vertebrates that are often overlooked in assessments of the importance of sagebrush (*Artemisia* spp.) ecosystems for wildlife. Given their dependence on water, few amphibians are strongly associated with sagebrush habitats, although several use these uplands for foraging, shelter, or dispersal. Of the 60 amphibian species that are predicted to occur within the sagebrush biome, the Great Basin spadefoot (*Spea intermontana*) is probably the only species that occupies enough of the biome and lives predominantly in terrestrial habitats (mostly in burrows) to be considered sagebrush associated.

Of the 116 reptiles that are predicted to occur within the sagebrush biome, about 5 lizards and 5 snakes were identified as both strongly associated with sagebrush habitats and occupied areas likely to be managed for sage-grouse (*Centrocercus* spp). However, this list could be lower or higher depending on the specific location within the biome, and there remains considerable uncertainty regarding potential threats to reptiles, as well as basic information on distribution and abundance of most reptile species.

Introduction

Amphibians and reptiles are grouped taxonomically as herpetofauna, but they are distinct vertebrates that have different life-history attributes and habitat requirements that influence their distribution and abundance within sagebrush (*Artemisia* spp.) ecosystems. In general, amphibians are limited by available moisture in any given habitat and reptiles are constrained by habitat temperature (Qian, 2010). Almost all amphibians within the sagebrush biome require surface water for reproduction (the only exception are the Plethodon salamanders). Adult amphibians are also rarely found far from water, but most species that occur in arid areas will use nearby terrestrial habitats. Thus, no amphibian is truly a sagebrush obligate, but a few are found in sagebrush habitats. These species may forage or disperse through sagebrush uplands and use burrows to escape adverse temperature and moisture conditions. For reptiles, snakes and lizards (members of the order Squamata) warrant the most attention from land managers in sagebrush ecosystems because turtles and tortoises are rarely found in or adjacent to sagebrush habitats (Pilliod and others, 2020a). Squamates are particularly diverse in the arid and semiarid regions of the western United States (fig. I1). Several species could be considered sagebrush dependent, and many species are sagebrush associated. All common and scientific names in this chapter are derived from Crother (2017).

Amphibians

A query of all amphibian species that have Gap Analysis Program (GAP) distribution maps available (U.S. Geological Survey, 2013a) was conducted to identify those occurring within the sagebrush biome. The distribution maps are “predictions of the spatial distribution of suitable environmental and land cover conditions within the United States for individual species” (U.S. Geological Survey, 2013a). Sixty amphibian species are predicted to occur within the sagebrush biome, but only 27 have greater than 10 percent of their distribution within the biome (app. I1, table I1.1). Of those, only 2 species had greater than 10 percent of their distribution within priority habitat management areas (PHMAs) for greater sage-grouse (*Centrocercus urophasianus*: the Great Basin spadefoot [*Spea intermontane*]; 95 percent of the distribution found within the biome) and Columbia spotted frog ([*Rana luteiventris*]; 75 percent of the distribution found within the biome; table I1).

The Great Basin spadefoot (fig. I2) was also identified by Rowland and others (2006) as an amphibian that shares similar habitats as sage-grouse. In Wyoming, spatial analyses suggest that the Great Basin spadefoot might benefit from an umbrella reserve created for sage-grouse (Carlisle and others, 2018b). In contrast, the Columbia spotted frog, northern leopard frog (*Lithobates pipiens*), and plains spadefoot (*Spea bombifrons*) may not benefit from greater sage-grouse habitat conservation in that State and may be at greater risk if development is redirected to areas outside of the reserves (Carlisle and others, 2018b). Several other amphibian species whose distributions overlapped with the sagebrush biome by more than 10 percent (app. I1, table I1.1) can occasionally be found in sagebrush habitats: Inyo Mountains salamander (*Batrachoseps campi*), long-toed salamander (*Ambystoma macrodactylum*), barred tiger salamander (*Ambystoma mavortium*), Great Plains toad (*Anaxyrus cognatus*), Wyoming toad (*Anaxyrus baxteri*), western toad (*Anaxyrus boreas*), Woodhouse’s toad (*Anaxyrus woodhousii*), plains spadefoot (*Spea bombifrons*), boreal chorus frog (*Pseudacris maculata*), northern leopard frog, Sierran or Pacific treefrog (*Pseudacris*

[1]U.S. Geological Survey.

sierra, formerly synonymous with *Pseudacris regilla*), and canyon treefrog (*Hyla arenicolor*; Pilliod and Wind, 2008). Of those species, the barred tiger salamander, northern leopard frog, Woodhouse's toad, Great Plains toad, boreal chorus frog, and plains spadefoot are all predicted to occupy a considerable part of the PHMAs at the eastern extent of the sagebrush biome (app. I1, table I1.1). Thus, some areas of the biome may warrant different prioritization for certain species and careful coordination with State and local biologists is prudent. The remaining species are largely dependent on aquatic, hot desert, coniferous, or subalpine ecosystems and are uncommonly found in sagebrush habitats.

Reptiles

Using an analytical approach similar to that used for amphibians, 53 of 116 reptile species predicted to occur within the sagebrush biome overlap with the biome by greater than 10 percent of their distributions. Only 10 species, including 5 lizards and 5 snakes, also had 9.2 percent or more of their distribution within PHMAs (table I2; app. I1, table I1.2). Of this group, the pygmy short-horned lizard (*Phrynosoma douglasii*) has the greatest proportion of its distribution within the sagebrush biome and PHMAs, followed by the common sagebrush lizard (*Sceloporus graciosus*), greater short-horned lizard (*P. hernandesi*), and desert nightsnake (*Hypsiglena torquata*; fig. I3, table I2). These findings are

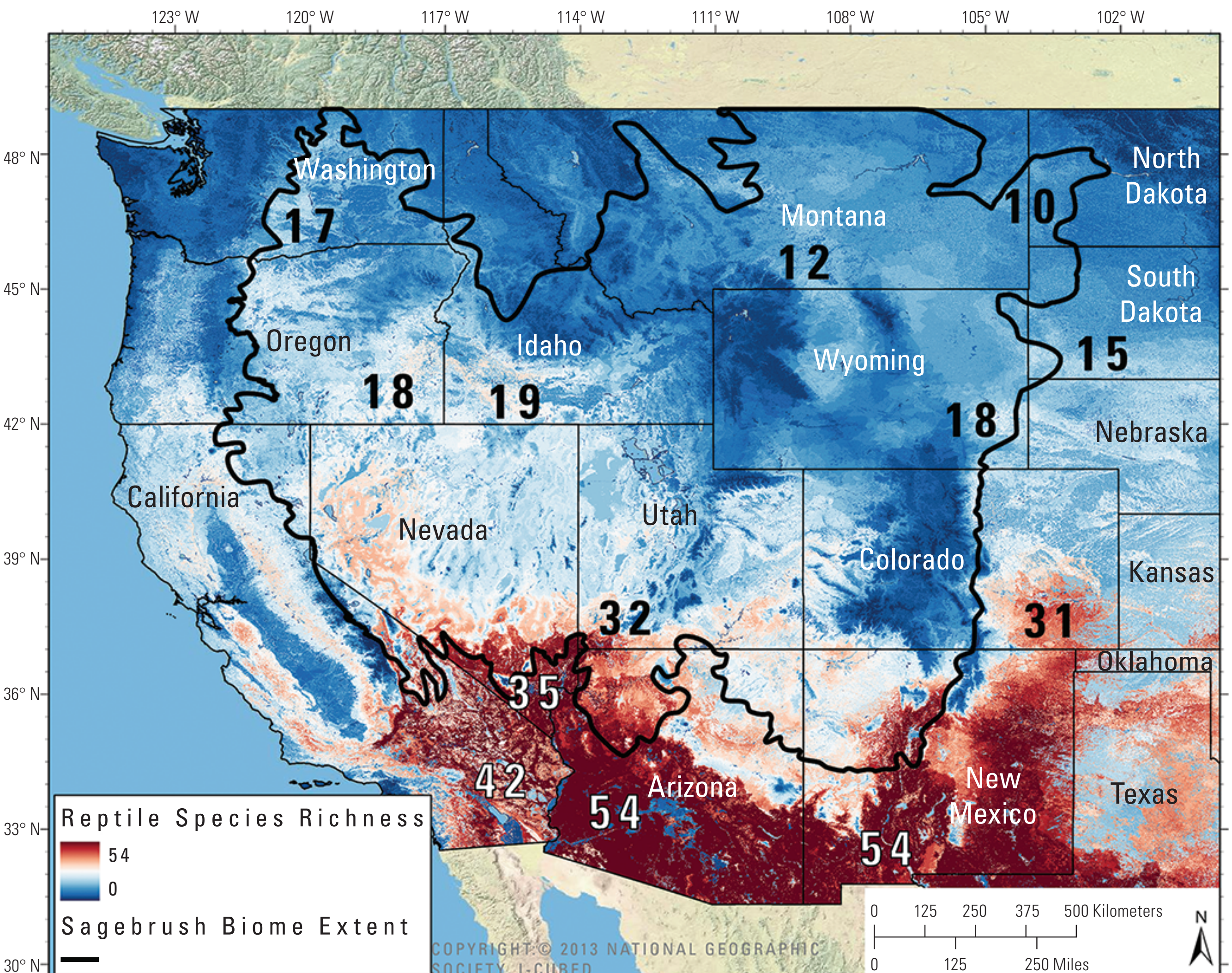

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Figure I1. Species richness of reptiles within the sagebrush (*Artemisia* spp.) biome relative to richness across the western United States. The numbers on the map indicate the maximum species richness found within each State and are placed in the general location of the maximum. Species richness was developed from predicted species distributions obtained from the U.S. Geological Survey GAP Species Program (U.S. Geological Survey, 2013a).

Figure I2. A Great Basin spadefoot (*Spea intermontana*) emerging during a thunder shower. Photograph by Alan St. John in 2008 and used with permission.

mostly consistent with previous assessments of reptile species whose distributions have considerable rangewide overlap with the distribution of sage-grouse (Rowland and others, 2006; Pilliod and others, 2020a). Interestingly, the panamint alligator lizard (*Elgaria panamintina*) ranks as the fifth most overlapping species but neither occurs in the PHMA area nor has it been highlighted in prior assessments featuring overlap with sage-grouse (Rowland and others, 2006; Carlisle and others, 2018b; Pilliod and others, 2020a). The panamint alligator lizard has a small distribution, but over 80 percent occurs within the southwestern extent of the sagebrush biome (table I2); thus, this species will only be a concern for managers in a very specific part of the biome (east-central California/Nevada border).

These regionally specific differences in species importance were also identified by Carlisle and others (2018b), who found that the greater short-horned lizard appears to be the only reptile that might benefit from the umbrella reserves created for sage-grouse in Wyoming. They also concluded that four snake species that occur in the eastern part of the biome may experience negative effects if surface development is redirected to areas outside of the PHMAs (Carlisle and others, 2018b). These species include the plains hog-nosed snake (*Heterodon nasicus*), eastern milksnake (*Lampropeltis triangulum*), smooth greensnake (*Opheodrys vernalis*), and northern rubber boa (*Charina bottae*). All of these species were identified in our overlap analysis (app. I1, table I1.2), and the rubber boa ranked seventh (table I2). Several additional lizard and snake species warrant some consideration in all or part of the biome, particularly those ranked 6–11 in table I2 (see also Pilliod and others, 2020a). Lastly, the western pond turtle (*Actinemys marmorata*) was the only turtle or tortoise with greater than 10 percent of its distribution within the biome, but this species is rarely associated with sagebrush habitats, including during nesting or overwintering.

Conservation Status

Few amphibians or reptiles have national conservation status in sagebrush ecosystems. Of those that overlap with the biome by at least 10 percent of their distribution, the Jemez Mountains salamander (*Plethodon neomexicanus*), Oregon spotted frog (*Rana pretiosa*), Sierra Nevada yellow-legged frog (*Rana sierra*), Wyoming toad (*Anaxyrus baxteri*), and the Yosemite toad (*Anaxyrus canorus*) are listed as threatened or endangered under the Endangered Species Act of 1973 (16 U.S.C. 1531 et seq.; app. I1). An additional 15 amphibian and 19 reptile species have high conservation priority in at least one western State (app. I1). This list of species lengthens when considering the taxonomic complexity of the species with high conservation priority and evaluating subspecies requirements. Therefore, State Wildlife Action Plans may need to be consulted for current or local information. Habitat requirements for these overlapping species of highest concern should be evaluated. For example, the northern tree lizard (*Urosaurus ornatus wright*) is a subspecies designated under highest conservation need in Wyoming. The tree lizard is a species that depends on standing pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) trees for habitat and thus could be impacted by extensive conifer removal programs (James and M'Closkey, 2003).

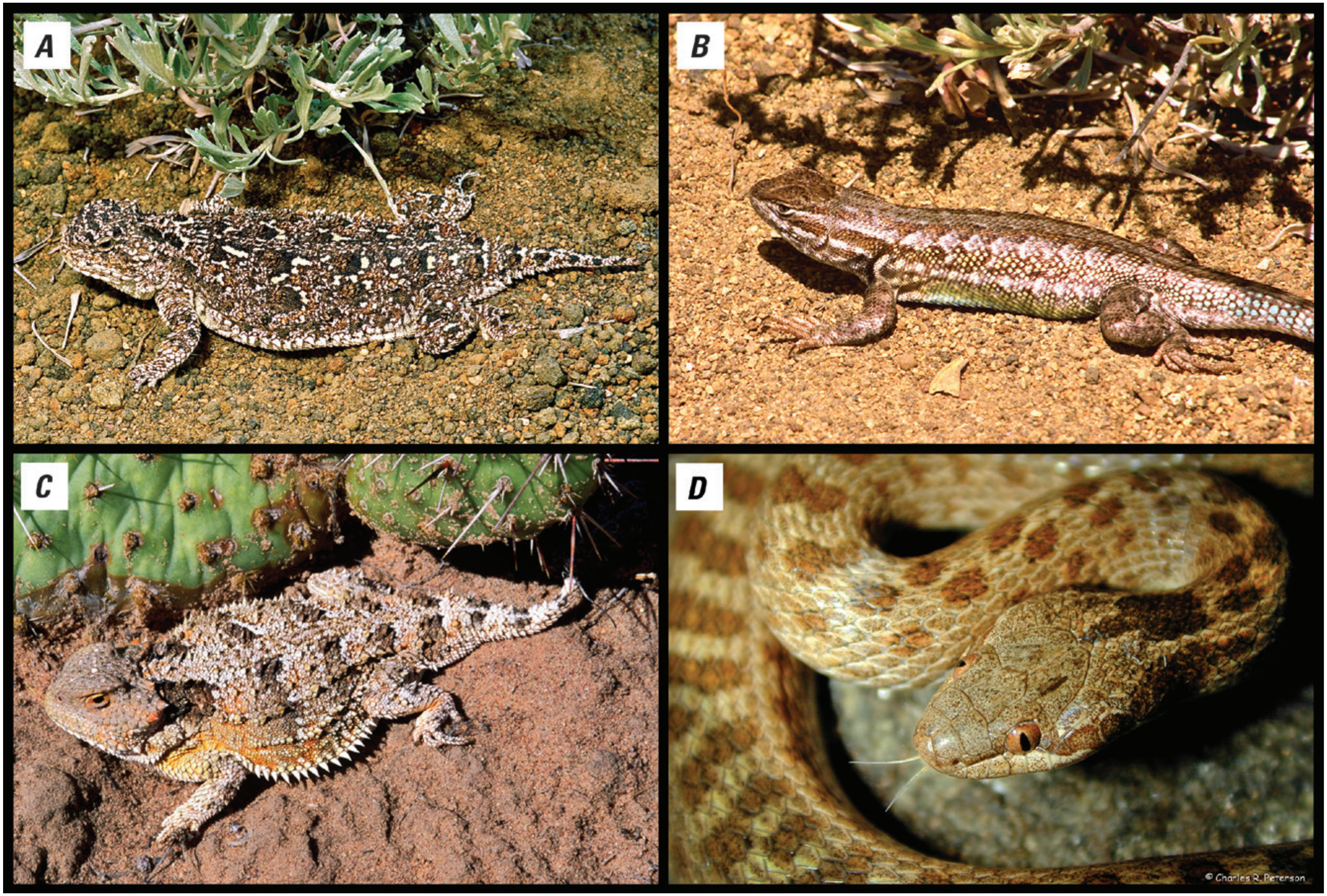

Figure I3. Four reptiles that have a high proportion of their distributions within the sagebrush (*Artemisia* spp.) biome and are commonly found in sagebrush habitats: *A*, The pygmy short-horned lizard (*Phrynosoma douglasii*) has the greatest proportion of its distribution within the biome and priority habitat management areas for greater sage-grouse (*Centrocercus urophasianus*); followed by *B*, the common sagebrush lizard (*Sceloporus graciosus*); *C*, greater short-horned lizard (*P. hernandesi*); and *D*, desert nightsnake (*Hypsiglena torquata*). Photographs *A*–*C* by Alan St. John in 2008, 2016, and 2008, respectively. Photograph D by Charles R. Peterson in 2000. All photos are used with permission.

Table I1. The top five amphibian species that have greater than 10 percent of their predicted distributions within the sagebrush (*Artemisia* spp.) biome. Two of these species also have greater than 10 percent of their predicted distributions within priority habitat management areas (PHMAs) created for the greater sage-grouse (*Centrocercus urophasianus*). Species are ranked (1 to 5) by the combined values of these two metrics.

[National and State priority conservation status is shown. National status abbreviations are as follows: E (endangered) and T (threatened). See individual State Wildlife Action Plans for additional information. Common and scientific names are derived from Crother (2017). -, no status or designation; CA, California, CO, Colorado, ID, Idaho, NV, Nevada; OR, Oregon; WY, Wyoming]

Common name	Scientific name	Rank	Amphibian distribution within the sagebrush biome (proportion)	Amphibian distribution within PHMAs (proportion)	National status	States with priority designations
Great Basin spadefoot	*Spea intermontana*	1	0.951	0.167	-	CO[2], WY[2]
Black toad	*Anaxyrus exsul*	2	1.000	0	-	CA[1]
Jemez Mountains salamander	*Plethodon neomexicanus*	3	1.000	0	T	-
Wyoming toad	*Anaxyrus baxteri*	4	1.000	0	E	WY[1]
Columbia spotted frog	*Rana luteiventris*	5	0.748	0.102	-	ID[1], NV[2], OR[1], WY[2]

[1]Highest conservation priority.

[2]Second tier of conservation priority.

Table I2. The top 11 reptile species that have greater than 10 percent of their predicted distributions within the sagebrush (*Artemisia* spp.) biome. Ten of these species also have greater than 9 percent of their predicted distributions within priority habitat management areas (PHMAs) created for greater sage-grouse (*Centrocercus urophasianus*). Species are ranked (1 to11) by the combined values of these two metrics.

[National and State priority conservation status is shown. National status abbreviations are as follows: E (endangered) and T (threatened). States are listed by their two-letter abbreviation. See individual State Wildlife Action Plans for additional information. Common and scientific names are derived from Crother (2017). -, no status or designation; OR, Oregon; ND, North Dakota; SD, South Dakota; WA, Washington; WY, Wyoming]

Common name	Scientific name	Rank	Reptile distribution within the sagebrush biome (proportion)	Reptile distribution within PHMAs (proportion)	National status	States with priority designations
Pygmy short-horned lizard	*Phrynosoma douglasii*	1	0.928	0.240	-	ND[2]
Common sagebrush lizard	*Sceloporus graciosus*	2	0.827	0.133	-	SD[2]
Greater short-horned lizard	*Phrynosoma hernandesi*	3	0.729	0.120	-	SD[2], WY[2]
Desert nightsnake	*Hypsiglena chlorophaea*	4	0.736	0.105	-	WY[2]
Panamint alligator lizard	*Elgaria panamintina*	5	0.813	0	-	-
Terrestrial gartersnake	*Thamnophis elegans*	6	0.668	0.103	-	-
Northern rubber boa	*Charina bottae*	7	0.631	0.136	-	WY[2]
Desert horned lizard	*Phrynosoma platyrhinos*	8	0.635	0.124	-	
Western fence lizard	*Sceloporus occidentalis*	9	0.637	0.111	-	
Striped whipsnake	*Coluber taeniatus*	10	0.630	0.099	-	WA[1]
Western rattlesnake	*Crotalus oreganus*	11	0.634	0.092	-	OR[1]

[1]Highest conservation priority.

[2]Second tier of conservation priority.

Threats

Threat to amphibians and reptiles in the sagebrush biome include loss and degradation of habitat as a result of invasive annual grasses (chap. K, this volume) and fire (chap. J, this volume), conversion of native vegetation to residential developments (chap. P, this volume), oil and gas development (chap. O, this volume), and habitat degradation from improper grazing (chap. P, this volume). The impact of these threats will vary both locally and regionally across the biome for this diverse group of species.

Management Considerations

At the biome-wide scale, some management actions such as juniper cutting, herbicide applications, and riparian restoration could affect herpetofauna species. Protecting surface water (that is, streams, ponds, and springs), riparian areas, and seasonally inundated meadows from degradation is probably the most important strategy for maintaining all amphibian species and breeding populations in sagebrush ecosystems. Adding a protective buffer around these areas that extends into the uplands could benefit some species that live in shallow, self-excavated burrows most of the year (especially the Great Basin spadefoot). Considering connectivity among water bodies could also benefit amphibians to avoid population isolation. However, the size of the buffer zones and the connectivity requirements needed for amphibians in sagebrush ecosystems is unknown.

Actions that open canopy and reduce invasive grasses yet leave some habitat structure for perching or basking habitat and as protection from predators could benefit several lizard and snake species (Pilliod and others, 2020a). For example, reducing dense cover of nonnative annual grasses could increase the probability of occupancy for many reptile species, as cheatgrass (*Bromus tectorum*) reduces locomotion and prey availability (Newbold, 2005; Hall and others, 2009). Most sagebrush-associated reptiles appear to avoid areas of dense grasses, including introduced Eurasian species such as crested wheatgrass (*Agropyron cristatum*; Pilliod and others, 2020a). However, empirically evaluated reptile responses to land treatments are not well studied (Pilliod and others, 2020a). Thus, a complete understanding of the effects of management actions on herpetofauna is unlikely and a certain level of uncertainty is expected.

Acknowledgments

Thanks to Deanna Olson (U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station) and Robert Arkle (U.S. Geological Survey) for their thoughtful discussions on the topic. Charles R. Peterson and Alan St. John provided photographs.

Appendix I1. Amphibians and Reptiles that Overlap with the Sagebrush Biome

Table I1.1. Amphibians that overlap with the sagebrush (*Artemisia* spp.) biome by at least 10 percent of their predicted distribution, the proportion of their distribution within priority habitat management areas (PHMAs) created for the greater sage-grouse (*Centrocercus urophasianus*), and their national and State conservation status. Species are ranked by the combined values of both overlap metrics.

[National status abbreviations: E, endangered; T, threatened, as defined by the Endangered Species Act of 1973 (16 U.S.C. 1531 et seq.). See State Wildlife Action Plans for additional information. Common and scientific names are derived from Crother (2017). -, no status or designation; <, less than; AZ, Arizona; CA, California; CO, Colorado; ID, Idaho; NV, Nevada; NM, New Mexico; ND, North Dakota; OR, Oregon; UT, Utah; WA, Washington; WY, Wyoming]

Common name	Scientific name	Rank	Amphibian distribution within the sagebrush biome (proportion)	Amphibian distribution within PHMAs (proportion)	National status	States with priority designations
Great Basin spadefoot	*Spea intermontana*	1	0.951	0.167	-	CO[2], WY[2]
Black toad	*Anaxyrus exsul*	2	1.000	0	-	CA[1]
Jemez Mountains salamander	*Plethodon neomexicanus*	3	1.000	0	T	-
Wyoming toad	*Anaxyrus baxteri*	4	1.000	0	E	WY[1]
Columbia spotted frog	*Rana luteiventris*	5	0.748	0.102	-	ID[1], NV[2], OR[1], WY[2]
Inyo Mountains salamander	*Batrachoseps campi*	6	0.686	0	-	-
Western toad	*Anaxyrus boreas*	7	0.643	0.016	-	ID[2], NM[1], OR[2], WY[1]
Boreal chorus frog	*Pseudacris maculata*	8	0.494	0.052	-	NM[2]
Sierran treefrog	*Pseudacris sierra*	9	0.504	0.042	-	-
Long-toed salamander	*Ambystoma macrodactylum*	10	0.387	0.014	-	-
Canyon treefrog	*Hyla arenicolor*	11	0.384	0.004	-	CO[2]
Woodhouse's toad	*Anaxyrus woodhousii*	12	0.307	0.041	-	ID[2]
Barred tiger salamander	*Ambystoma mavortium*	13	0.290	0.041	-	-
Baja California Treefrog	*Pseudacris hypochondriaca*	14	0.316	0	-	AZ[2]
Plains spadefoot	*Spea bombifrons*	15	0.251	0.044	-	ND[1], UT[1]
Northern leopard frog	*Lithobates pipiens*	16	0.252	0.019	-	AZ[1], CA[2], CO[1], ID[2], NV[2], NM[2], WA[1]
Sierra Nevada yellow-legged frog	*Rana sierrae*	17	0.224	0	E	CA[1]
Great Plains toad	*Anaxyrus cognatus*	18	0.193	0.026	-	NV[2], UT[1], WY[2]
Mexican spadefoot	*Spea multiplicata*	19	0.214	0.003	-	UT[1]
Rocky Mountain tailed frog	*Ascaphus montanus*	20	0.202	0.001	-	OR[2], WA[2]
Amargosa toad	*Anaxyrus nelsoni*	21	0.163	0	-	NV[2]
Yosemite toad	*Anaxyrus canorus*	22	0.156	0	T	CA[2]
Red-spotted toad	*Anaxyrus punctatus*	23	0.155	<0.001	-	-
Idaho giant salamander	*Dicamptodon aterrimus*	24	0.155	0	-	-
Oregon spotted frog	*Rana pretiosa*	25	0.153	0.001	E	CA[2], OR[1], WA[1]
Mount Lyell salamander	*Hydromantes platycephalus*	26	0.148	0	-	-
Arizona toad	*Anaxyrus microscaphus*	27	0.123	0.001	-	AZ[2], NV[2]

[1]Highest conservation priority.

[2]Second tier of conservation priority.

Table I1.2. Reptiles that overlap with the sagebrush (*Artemisia* spp.) biome by at least 10 percent of their predicted distribution, the proportion of their distribution within priority habitat management areas (PHMAs) created for greater sage-grouse (*Centrocercus urophasianus*), and their national and State conservation status. Species are ranked by the combined values of both overlap metrics.

[National status abbreviations: E (endangered) and T (threatened), as defined by the Endangered Species Act of 1973 (16 U.S.C. 1531 et seq.). See State Wildlife Action Plans for additional information. Common and scientific names are derived from Crother (2017). -, no status or designation; <, less than; AZ, Arizona; CA, California; CO, Colorado; ID, Idaho; NV, Nevada; NM, New Mexico; ND, North Dakota; OR, Oregon; SD, South Dakota; UT, Utah; WA, Washington; WY, Wyoming]

Common name	Scientific name	Rank	Reptile distribution within the sagebrush biome (proportion)	Reptile distribution within PHMAs (proportion)	National status	States with priority designations
Pygmy short-horned lizard	*Phrynosoma douglasii*	1	0.928	0.240	-	ND[2]
Common sagebrush lizard	*Sceloporus graciosus*	2	0.827	0.133	-	SD[2]
Greater short-horned lizard	*Phrynosoma hernandesi*	3	0.729	0.120	-	SD[2], WY[2]
Desert nightsnake	*Hypsiglena chlorophaea*	4	0.736	0.105	-	WY[2]
Panamint alligator lizard	*Elgaria panamintina*	5	0.813	0	-	-
Terrestrial gartersnake	*Thamnophis elegans*	6	0.668	0.103	-	-
Northern rubber boa	*Charina bottae*	7	0.631	0.136	-	WY[2]
Desert horned lizard	*Phrynosoma platyrhinos*	8	0.635	0.124	-	-
Western fence lizard	*Sceloporus occidentalis*	9	0.637	0.111	-	-
Striped whipsnake	*Coluber taeniatus*	10	0.630	0.099	-	WA[1]
Western rattlesnake	*Crotalus oreganus*	11	0.634	0.092	-	OR[1]
Plateau fence lizard	*Sceloporus tristichus*	12	0.632	0.047	-	-
Great Basin collared lizard	*Crotaphytus bicinctores*	13	0.609	0.064	-	ID[2]
Plateau striped whiptail	*Aspidoscelis velox*	14	0.630	0.009	-	-
Ornate tree lizard	*Urosaurus ornatus*	15	0.587	0.039	-	-
Common side-blotched lizard	*Uta stansburiana*	16	0.514	0.068	-	-
Pai striped whiptail	*Aspidoscelis pai*	17	0.569	0	-	AZ[2]
Long-nosed leopard lizard	*Gambelia wislizenii*	18	0.457	0.083	-	CO[2]
Western skink	*Plestiodon skiltonianus*	19	0.478	0.030	-	-
Sonoran Mountain kingsnake	*Lampropeltis pyromelana*	20	0.450	0.025	-	NV[2]
Yellow-backed spiny lizard	*Sceloporus uniformis*	21	0.470	0.001	-	-
Tiger whiptail	*Aspidoscelis tigris*	22	0.398	0.052	-	-
Prairie rattlesnake	*Crotalus viridis*	23	0.380	0.053	-	-
Gophersnake	*Pituophis catenifer*	24	0.352	0.050	-	-
Plains hog-nosed snake	*Heterodon nasicus*	25	0.312	0.057	-	MT[2], ND[1], WY[2]
Desert night lizard	*Xantusia vigilis*	26	0.364	0.001	-	UT[2]
Western patch-nosed snake	*Salvadora hexalepis*	27	0.343	0.017	-	-
Gilbert's skink	*Plestiodon gilberti*	28	0.321	0	-	NV[2]

Table I1.2. Reptiles that overlap with the sagebrush (*Artemisia* spp.) biome by at least 10 percent of their predicted distribution, the proportion of their distribution within priority habitat management areas (PHMAs) created for greater sage-grouse (*Centrocercus urophasianus*), and their national and State conservation status. Species are ranked by the combined values of both overlap metrics.—Continued

[National status abbreviations: E (endangered) and T (threatened), as defined by the Endangered Species Act of 1973 (16 U.S.C. 1531 et seq.). See State Wildlife Action Plans for additional information. Common and scientific names are derived from Crother (2017). -, no status or designation; <, less than; AZ, Arizona; CA, California; CO, Colorado; ID, Idaho; NV, Nevada; NM, New Mexico; ND, North Dakota; OR, Oregon; SD, South Dakota; UT, Utah; WA, Washington; WY, Wyoming]

Common name	Scientific name	Rank	Reptile distribution within the sagebrush biome (proportion)	Reptile distribution within PHMAs (proportion)	National status	States with priority designations
Speckled rattlesnake	*Crotalus mitchellii*	29	0.267	0.002	-	-
Desert spiny lizard	*Sceloporus magister*	30	0.250	0.001	-	CO[1]
Smooth greensnake	*Opheodrys vernalis*	31	0.217	0.025	-	MT[2], ND[1], WY[2]
North American racer	*Coluber constrictor*	32	0.205	0.031	-	-
Many-lined skink	*Plestiodon multivirgatus*	33	0.230	<0.001	-	UT[1], SD[1]
Black-necked gartersnake	*Thamnophis cyrtopsis*	34	0.203	0	-	-
Zebra-tailed lizard	*Callisaurus draconoides*	35	0.192	0.001	-	-
Plains gartersnake	*Thamnophis radix*	36	0.161	0.028	-	-
Smith's black-headed snake	*Tantilla hobartsmithi*	37	0.177	<0.001	-	CO[2]
Sidewinder	*Crotalus cerastes*	38	0.169	<0.001	-	-
Common chuckwalla	*Sauromalus ater*	39	0.167	<0.001	-	-
Eastern collared lizard	*Crotaphytus collaris*	40	0.160	0.001	-	-
Western groundsnake	*Sonora semiannulata*	41	0.151	0.007	-	-
Western lyre snake	*Trimorphodon biscutatus*	42	0.156	0.001	-	-
Arizona black rattlesnake	*Crotalus cerberus*	43	0.137	0	-	AZ[2]
Western banded gecko	*Coleonyx variegatus*	44	0.135	<0.001	-	-
Eastern milksnake	*Lampropeltis triangulum*	45	0.117	0.014	-	AZ[1], CO[2], MT[2]
Little striped whiptail	*Aspidoscelis inornata*	46	0.129	0	-	-
Long-nosed snake	*Rhinocheilus lecontei*	47	0.124	0.005	-	CO[2], ID[2]
Coachwhip	*Coluber flagellum*	48	0.123	0.001	-	-
Sierra garter snake	*Thamnophis couchii*	49	0.116	0.001	-	-
Clark's spiny lizard	*Sceloporus clarkii*	50	0.115	0	-	-
Western pond turtle	*Actinemys marmorata*	51	0.106	<0.001	-	OR[1], WA[1]
Black-tailed rattlesnake	*Crotalus molossus*	52	0.102	0	-	-
Chihuahuan spotted whiptail	*Aspidoscelis exsanguis*	53	0.100	0	-	-

[1]Highest conservation priority.

[2]Second tier of conservation priority.

Part II. Change Agents in the Sagebrush Biome—Extent, Impacts, and Efforts to Address Them

Chapter J. Altered Fire Regimes

By Michele R. Crist,[1] Rick Belger,[1] Kirk W. Davies,[2] Dawn M. Davis,[3] James R. Meldrum,[4] Douglas J. Shinneman,[4] and Kenneth E. Mayer[5]

Executive Summary

Historically, fire regimes in sagebrush (*Artemisia* spp.) ecosystems were highly variable and were influenced by the diverse climatic and topographic conditions found across the American West. However, historical fire regimes in sagebrush-dominated landscapes are not well understood in many areas, primarily owing to methodological challenges in finding or adequately quantifying evidence of past fire in shrubland communities. Uncharacteristic fire owing to the spread of fire-prone invasive annual grasses is a substantial and pervasive threat to the persistence of sagebrush ecosystems, particularly in the western portion of the sagebrush biome. Factors such as large-scale nonnative annual grass invasions, climate change, and other human activities have accelerated wildfire cycles, increased fire size and severity, and lengthened fire seasons to the point that postfire recovery and current wildfire-management practices cannot keep pace. Hotter and drier conditions, combined with human-ignited fires, have increased the length of the fire season by 134 percent for the western sagebrush biome. Fire sizes have increased substantially over the past two decades, with fires of more than 40,469 hectares (100,000 acres) becoming more common. A large majority of wildfires in the United States are caused by humans (for example, from campfires, target shooting, power lines, fireworks, debris burning, and arson). In 2018, human-caused ignitions accounted for approximately 64 percent of fires and 55 percent of acres burned on U.S. Department of the Interior, Bureau of Land Management lands covering the majority of the sagebrush biome across the West. Conversely, in some areas, activities such as past overgrazing or fire suppression, have led to less frequent fires, which also has implications for sagebrush communities.

The greatest impact of altered fire regimes on the sagebrush biome is the resulting large-scale ecotype conversion from native shrub-perennial grass communities to fire-prone, nonnative, annual plant communities. These type conversions are often permanent, and sagebrush ecosystem restoration is difficult and expensive owing to unfavorable environmental conditions for reestablishment of native plants. Moreover, after invasive annual plants become dominant, the increased fuel loads they create can lead to more frequent fires, further promoting these plants' expansion. More frequently occurring fires necessitate ever greater resources for increasing fire-suppression needs.

The scope of potential impacts to sagebrush-dependent species is epitomized by greater sage-grouse (*Centrocercus urophasianus*). About 21 percent of these birds' priority habitat management areas have burned in the Great Basin since 2000. Pygmy rabbits (*Brachylagus idahoensis*) have lost from 13 to 17 percent of their occupied habitat to fire since 2000.

Wildfires impose numerous economic costs, including costs of prevention, suppression, and postfire restoration. Many factors have resulted in the increasing costs of fire suppression over the past two decades. For fires in the sagebrush biome, these factors include increases in human-caused ignitions and the spread of fire-prone invasive plants, combined with trends toward longer fire seasons, larger fires, and more extreme fire-weather conditions. The direct and indirect cost of wildfires in the United States ranges from 71 to 348 billion dollars annually. Altered fire regimes in sagebrush landscapes also have a direct impact on local communities that can best be quantified as losses of ecosystem services. These impacts include increased costs of critical services that people rely on for health and survival, loss of recreation opportunities, loss of cultural traditions and sites, and loss of existence values of wildlife species and plant communities.

Postfire recovery in sagebrush landscapes is expensive, especially in hotter-drier areas where invasive plants are prone to dominate after fire. U.S. Department of the Interior, Bureau of Land Management, Emergency Stabilization and Burned Area Rehabilitation programs aim to prevent further degradation after fire and protect natural resources by rehabilitating landscapes unlikely to recover naturally after fire. However, funding requests for rehabilitation of burned areas after large fires often exhaust postfire recovery budgets, and available seed supplies for establishing desirable plant species are limited. Changes in these postfire recovery program budgeting and policy structures may be needed to increase flexibility, prioritize funding based on ecological need, provide for quicker responses after fire, and allow longer implementation times to support postfire recovery efforts.

Changes in Federal and State wildfire management budgeting and policy structures to increase flexibility and provide for quicker responses to fire could help improve overall fire suppression effectiveness. Collaboration and partnerships across jurisdictional boundaries, agencies, and disciplines is resulting in consistent wildfire-management approaches achieved in some areas. Applying these

[1]U.S. Department of the Interior, Bureau of Land Management.

[2]U.S. Department of Agriculture, Agricultural Research Service.

[3]U.S. Fish and Wildlife Service.

[4]U.S. Geological Survey.

[5]Western Association of Fish and Wildlife Agencies.

approaches more broadly can result in greater consistency across the western United States. Review of successful coordination strategies and agreements can enable adoption by other agencies where appropriate. Interagency reviews and lessons learned that are implemented after fire incidents can be used to improve strategies and tactics for future fires with similar conditions. Coordination efforts such as Rangeland Fire Protection Associations and other cooperative agreements between State and Federal fire management agencies have proven to be successful in providing additional capacity and resources where these resources are lacking, especially in remote areas where State and Federal resources are not able to respond quickly to a fire incident.

Introduction

Historical fire regimes and their impacts on landscape-scale abundance and distribution of sagebrush (*Artemisia* spp.) ecosystems are not fully understood, but they were likely highly variable over long timeframes and among different sagebrush communities. In recent decades, uncharacteristic fire frequency and behavior caused by the influx of invasive annual grasses (for example, cheatgrass [*Bromus tectorum*]) has become the largest threat to western sagebrush landscapes. From 2000 to 2018, wildfires have burned more than 6 million hectares (ha; 15 million acres) of shrub-dominated landscapes on Federal lands, primarily in the Great Basin (fig. J1).

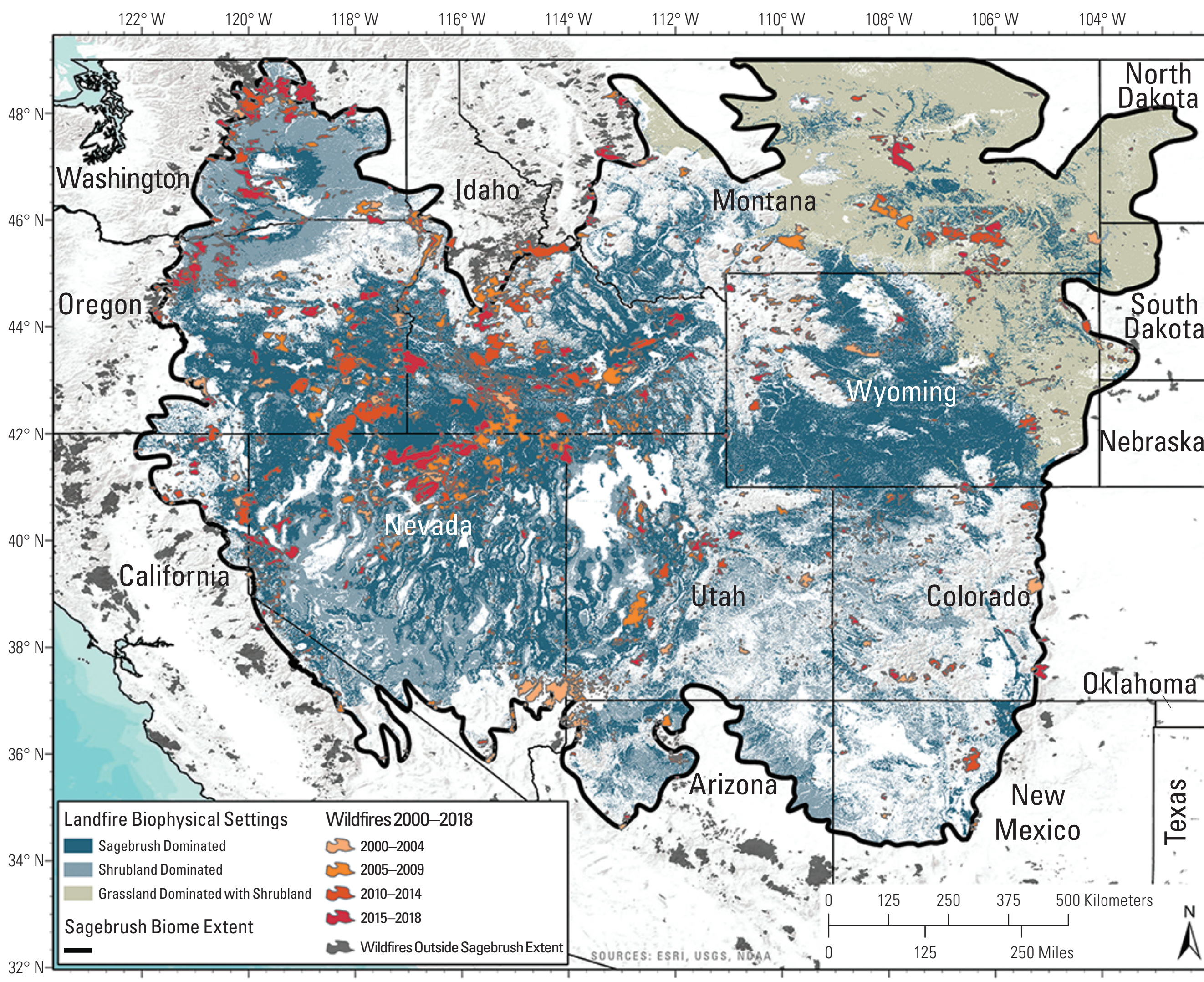

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Figure J1. Recent wildfire history for the sagebrush (*Artemisia* spp.) biome from 2000 to 2018. Fire perimeters shown in light orange to dark red depict locations where fires have burned in sagebrush-dominated communities and in grassland with sagebrush components. These communities are shown in varying shades of blue. Fire perimeters in dark gray are where fire has occurred in other vegetation types such as forested lands and other shrub communities (for example, chaparral). LANDFIRE BioPhysical Settings modified into Sagebrush Dominated, Shrubland Dominated, and Grassland Dominated with Shrubland (U.S. Geological Survey, 2014a). Wildfire information from U.S. Geological Survey (2019a).

Conversely, fire occurs less frequently than it likely did prior to settlement in mountain big sagebrush (*A. tridentata vaseyana*) communities at higher elevations where cheatgrass infestation has not occurred. In many areas, this lack of periodic fire has allowed pinyon-juniper and other conifers to expand into sagebrush communities (Miller and others, 2000; Miller and Tausch, 2001; chap. M, this volume). Addressing the causes and impacts of these altered fire regimes on sagebrush and sagebrush-associated wildlife is extremely challenging but essential to long-term conservation and retention of the multiple resources provided by this biome.

The Role of Wildfire in Sagebrush Ecosystems

Fire is an important natural disturbance in most terrestrial ecosystems that influences biological diversity, patterns of succession within natural communities, and ecological function over time and space. Determining how a particular ecosystem evolved with and responds to fire is important to understanding the historical or natural variability of those ecosystems, which in turn provides a baseline for detecting any ecological deviation or degradation caused by human influences. In sagebrush ecosystems, modern fire regimes have been influenced by numerous factors, including invasion of nonnative plant species, livestock grazing, and changing climate. However, these influences and their effects can vary greatly among different regions, landscapes, and sagebrush community types. Fire histories are often reconstructed using dendrochronology methods (for example, dating fire scars recorded in tree rings) and charcoal analysis of soils, lake sediments, and trees. Yet, finding this historical evidence of fire is relatively difficult in sagebrush-dominated landscapes, especially compared to finding historical evidence of fire in forested ecosystems. Although long-lived trees that survive fires and retain scars have been used to infer past fire frequency in sagebrush ecosystems, these inferences are limited to localized areas along forested ecotones. Mean fire return intervals are estimated to have ranged from a few decades in colder-moister sagebrush ecosystems near forest and woodland ecotones (Miller and Heyerdahl, 2008) to hundreds of years in hotter-drier sagebrush ecosystems (Bukowski and Baker, 2013).

Historical fire regimes in the sagebrush biome likely varied in large part because of the influence of climatic gradients on fuel loads and ignition rates. Sagebrush communities in the eastern part of the sagebrush biome are generally less fuel limited but also less prone to fire ignition depending on the timing of summer or monsoonal precipitation, whereas in the western part of the sagebrush biome, most precipitation occurs in the winter, and summers are dry. Differing precipitation patterns also occur along elevational and latitudinal gradients, with generally hotter-drier conditions in the south and at lower elevations, and cooler-wetter conditions in the north and at higher elevations (Brooks and others, 2015). In response, fuel loads and fire activity vary considerably by geography and with seasonal precipitation and ignition patterns, with more fire generally occurring where summers are drier than winters, lightning more regularly occurs in or nearby sagebrush ecosystems, and fuels are more continuous and less limiting.

As Euro-Americans settled the West, Native American land use practices, such as burning—which is thought to have been relatively common in some higher elevation sagebrush ecosystems (Griffin, 2002; Stewart, 2002; McAdoo and others, 2013)—were replaced with new land use practices, such as widespread livestock grazing, mining, and road building. Land management practices, such as fire suppression and sagebrush removal for grazing purposes, were also introduced. After the introduction of extensive livestock grazing in the late 1800s, fine fuels were substantially reduced across many sagebrush landscapes and fires likely became less frequent and burned with less intensity (Miller and others, 2011) until subsequent spread of invasive annual grasses that provide contiguous, fine-fuel loadings. These fire-prone, nonnative grasses currently dominate millions of acres of the sagebrush biome (Romme and others, 2009; Morris and Rowe, 2014; Brooks and others, 2015). All of these changes contributed to altered fuel characteristics and ignition patterns within sagebrush landscapes and have significantly altered sagebrush fire regimes over vast areas. Fire-driven conversion from native sagebrush communities to nonnative plant communities is considered a primary threat to sagebrush ecosystems and associated wildlife species, particularly in the western half of the sagebrush biome (U.S. Department of the Interior, 2015a).

The invasion of nonnative annual grasses and forbs, most notably cheatgrass, is the most influential factor in altering fire regimes across much of the western part of the sagebrush range (Knick and Rotenberry, 1997; Brooks and others, 2015). Cheatgrass can fill the interspaces between native perennials and facilitate fire spread where it would not otherwise occur, especially in arid regions where native plant productivity is low (Whisenant, 1990). Nonnative annual grasses also senesce and dry out earlier than most native vegetation, potentially elongating the wildfire season (Keane and others, 2008; Davies and Nafus, 2013). Nonnative annual grasses are of particular concern for more arid sagebrush shrublands, dominated by Wyoming big sagebrush (*A. tridentata wyomingensis*) and basin big sagebrush (*A. t. tridentata*; Brooks and others, 2016). These sagebrush communities are not adapted to frequent fires and often have a minimal perennial grass component resulting in low resilience to fire (that is, slow recovery) and a low resistance to cheatgrass invasion (Chambers and others, 2014a, b; Brooks and others, 2016; Chambers and others, 2017b). These conditions can result in greatly reduced fire-free intervals that encourage cheatgrass establishment while preventing reestablishment of the native sagebrush community. This dynamic leads to a self-perpetuating grass-fire cycle (D'Antonio and Vitousek, 1992) that favors the dominance and spread of invasive annual grasses, which in turn facilitates more frequent fire (Brooks and others, 2004; Brooks, 2008).

Mountain big sagebrush communities have been historically characterized as having shorter fire-return intervals compared to other sagebrush communities (those characterized by relatively infrequent fire). These montane sagebrush communities have moderate to high resilience following fire and thus recover more quickly than other sagebrush communities and are more resistant to cheatgrass invasion (Chambers and others, 2014a, b). Although climate variability and other dynamics play a role, past management activities are thought to have decreased competition and increased fire return intervals to the point where conifer species (especially juniper [*Juniperus* spp.] and pinyon [*Pinus* spp.]) in many areas can establish and eventually outcompete sagebrush, which also leads to a reduction of perennial grasses and forbs (Miller and others, 2000; Miller and Tausch, 2001; chap. M, this volume).

Recent Fire Trends and Patterns

It is challenging to quantify and summarize changes in fire regimes across the sagebrush biome over time for several reasons. First, recently published studies that analyzed broad scale, contemporary fire trends and patterns in the western United States generally, or in the sagebrush biome specifically, varied in spatial and temporal extents examined and methodologies used (Miller and others, 2011; Baker, 2013; Bukowski and Baker, 2013; Dennison and others, 2014; Brooks and others, 2015). Second, most of this research used perimeter data from large fires (greater than 405 ha; 1,000 acres). These data are mainly available only for fires that burned since the early 1980s. Third, with only about 30–35 years of accurately mapped large fires, it is difficult to meaningfully quantify useful attributes of fire regimes, such as mean fire return interval. This is especially true for parts of the sagebrush biome where fire is still relatively infrequent or where there is substantial interannual variability in fire occurrence.

Despite these challenges, some key fire trends have emerged. For instance, the proportion of cheatgrass-dominated areas that burned in recent decades is likely two to four times higher compared to areas dominated by other vegetation types in the Great Basin (Balch and others, 2013). Other commonalities among disparate fire regime studies have also emerged, and key fire regime trends analyzed by several of these studies are summarized across three organizational levels: (1) the broader sagebrush biome; (2) among ecoregions or floristic provinces (fig. J2); and (3) for dominant sagebrush taxa. Fire regime attributes discussed here include trends in fire area (that is, area burned), fire intervals (that is, fire rotation and mean fire return intervals), fire size, fire season length, and fire recurrence (reburns).

Fire Area

Across all fire-history studies relevant to sagebrush ecosystems, most have generally concluded that fire area (that is, area burned) over the past approximately 30 years has increased in some regions. However, there is mixed agreement regarding landscape trends in area burned owing to different spatial and temporal extents, ecosystem delineations, statistical approaches, and datasets used. Thus, it is important to note that direct comparisons must be considered carefully. For instance, one study found no significant trends in total area burned over a 25-year period (1984–2008) across the sagebrush biome (Baker, 2013). However, using different methods and a slightly longer period of record (1984–2013), Brooks and others (2015) found a potentially significant upward trend in total area burned across the sagebrush biome.

In addition, detection of ecoregional trends in area burned has also varied among studies. Miller and others (2011) found a weak but significant upward trend in area burned in four of five floristic provinces (Northern Great Basin, Southern Great Basin, Silver Sagebrush, and Wyoming Basin; fig. J2). Baker (2013) found significant trends in only two of seven provinces (Colorado Plateau and Columbia Basin) by using different methods, although three others (Silver Sagebrush, Snake River Plains, and Southern Great Basin) were nearly significant. In another study (Brooks and others, 2015), there was strong evidence of increased fire area in the Wyoming Basin, Snake River Plain, Columbia Basin, and Great Plains (comparable to the Silver Sagebrush Province) but not in the Northern Great Basin, Southern Great Basin, and Colorado Plateau. These discrepancies are not surprising given methodological difference, and also because of the substantial limitations of statistical trend detection for a short record of time relative to high interannual variability in area burned over the long term.

Despite differences in study methods and findings, there are important key points of agreement to highlight in area-burned trends and patterns. First, most studies documented general upward trends in annual fire area across the sagebrush biome, even if an increase was not detected as a significant trend in all studies or across all floristic provinces. There is also agreement in ecoregional trends of increasing area burned, especially for the Columbia Basin and somewhat for the Silver Sagebrush (Great Plains) floristic provinces. Second, these studies document that a disproportionately larger area has burned in the western region of the sagebrush biome than in the eastern region. Twenty-one percent of the total available area burned in the western half of the sagebrush biome during 1984–2013 (17 percent when considering repeatedly burned area only once), representing 82 percent of the total burned area within greater sage-grouse (*Centrocercus urophasianus*) range (Brooks and others, 2015). In contrast, only 5 percent of the total available area burned during that period in the eastern half of the sagebrush biome. Third, much of the total area burned over time occurred during relatively infrequent years with large and extensive fires, but consistency in the temporal patterns of large fires within bioclimatic regions suggests the strong influence of interannual climate variability on area burned (Littell and others, 2009; Balch and others, 2013; Pilliod and others, 2017a). Wet years enhance fine fuels, and this leads to increases in the amount of area burned when followed by dry years in sagebrush ecosystems, especially in areas occupied by nonnative annuals, such as cheatgrass (Balch and others, 2013; Pilliod and others, 2017a).

Fire Intervals

The time between fires, or fire interval (often quantified as either a mean fire return interval or fire rotation to characterize the fire regime for a given point or landscape area over time), has great importance for the sustainability of sagebrush ecosystems. This is particularly important if average intervals are too short for sagebrush plants to regenerate and provide adequate habitat conditions for sagebrush-dependent wildlife. Most sagebrush taxa are slow to recover after fire because of limited seed dispersal, low frequency of resprouting, and poor seed viability (Young and Evans, 1989; Miller and others, 2011). Several studies have documented that sagebrush recovery to near preburn cover after fire can take from several decades to more than a century (for example, Welch and Criddle, 2003; Lesica and others, 2007; Shinneman and McIlroy, 2016). Sagebrush landscapes were characterized by large patches of both dense and scattered sagebrush, as well as large, grass-dominated areas based on historical General Land Office Survey data from the late 1800s to the early 1900s (Bukowski and Baker, 2013). Prior to Euro-American settlement, small fires likely occurred more often, and large fires were more infrequent within sagebrush stands. This resulted in dynamic sagebrush landscapes with a fine-scaled

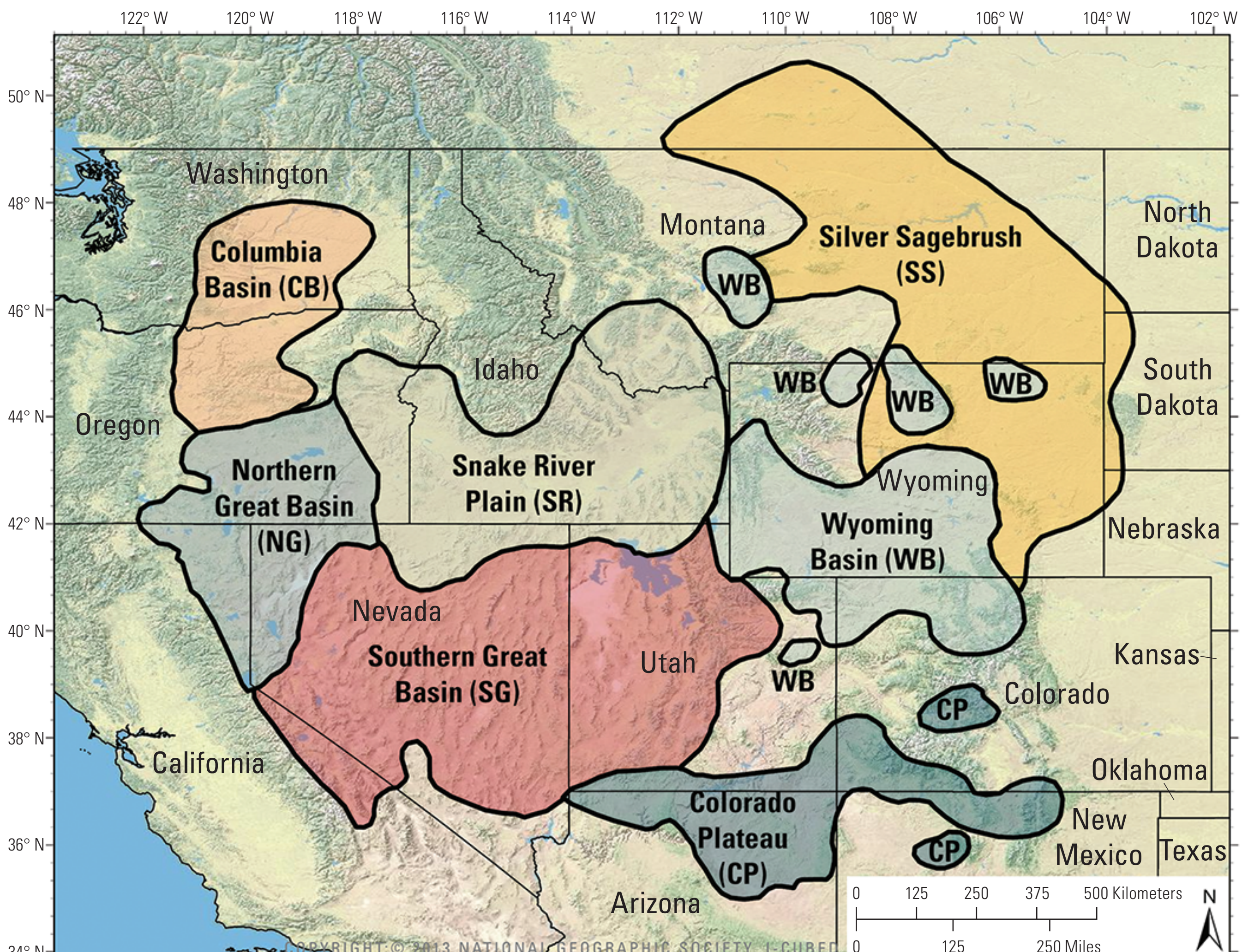

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Figure J2. Seven floristic provinces used in recently published studies that analyzed fire patterns and trends in the sagebrush (*Artemisia* spp.) biome, particularly by focusing on ecosystem types or biophysical settings capable of supporting sagebrush as dominant species.

small patch mosaic that alternated between periods of ecosystem recovery and more extensive maturity (Bukowski and Baker, 2013).

A key issue is whether modern fire intervals for sagebrush communities are different from historical intervals and whether differences between the two suggest fire regimes have departed from their historical ranges of variability thus limiting or prohibiting sagebrush recovery after fire. Modern fire intervals among floristic regions and sagebrush community types have been more accurately assessed using contemporary fire perimeter data. Contemporary fire intervals are likely shorter than historical intervals in many but not all sagebrush ecosystem types and regions (Baker, 2013; Brooks and others, 2015). Modern fire intervals for some big sagebrush (*A. tridentata*) communities in the western part of the sagebrush biome represented a substantial reduction compared to historical fire intervals based on land-survey data, particularly for Wyoming big sagebrush, with historical fire rotations that likely exceeded 200 years in most regions (Bukowski and Baker, 2013). In addition, contemporary rotations for some xeric low sagebrush (*A. arbuscula*) and black sagebrush (*A. nova*) communities are also generally substantially shorter than historical rotations, which were estimated to have exceeded 1,000 years (Baker, 2013; Bukowski and Baker, 2013).

In contrast, for much of the eastern half of the sagebrush biome and many mountain big sagebrush communities, studies suggest that modern fire intervals are often either similar or even longer than historical intervals (Baker, 2013; Bukowski and Baker, 2013; Brooks and others, 2015). Contemporary fire intervals are roughly 500–1,000 years for big sagebrush communities in the eastern sagebrush range (Brooks and others, 2015). However, differences between current and historical intervals for these sagebrush communities, as well as for little and black sagebrush communities, are difficult to assess given the relative lack of historical data or currently available reliable estimates.

Fire Size

Historical fire sizes in sagebrush ecosystems are poorly understood. Some researchers have suggested that infrequent large fires were part of historical sagebrush fire regimes (Baker, 2011; Bukowski and Baker, 2013), whereas others suggest that the sizes of sagebrush fires during recent decades may be unprecedented (Keane and others, 2008). Yet, both historical and contemporary fire-size distributions suggest that burn patterns on sagebrush landscapes fluctuated temporally, as episodes of large fires were followed by interludes with smaller fires (Bukowski and Baker, 2013). Fire size is strongly influenced by topography, fuel continuity, fire-weather conditions, and climate. Relatively gentle terrain supporting sagebrush is strongly correlated with fire spread (Baker, 2009), and larger fires in sagebrush are often characterized by a southwest to northeast orientation owing to broad-scale atmospheric conditions that drive wind patterns (Baker, 2013).

Recent trends suggest that fire sizes are increasing across much of the sagebrush biome based on contemporary fire-history data. Fire size increased with time throughout most ecoregions of the western United States between 1984 and 2011 (Dennison and others, 2014). Specific to the sagebrush biome, Baker (2013) compared the top fire years in sagebrush vegetation types in the western United States based on the total area burned over two consecutive 12-year periods (1985–1996 and 1997–2008) and suggested that fire sizes may be increasing. In general, larger fires occurred in the Northern Great Basin, Snake River Plain, and Southern Great Basin compared to other regions, and notable and significant upward shifts in annual fire-size distributions occurred throughout the western region (but not the eastern region) of the sagebrush biome over a recent 30-year period (1984–2015; Brooks and others, 2015). Most (39 of 50) of the largest fires that occurred in the Great Basin during 1980–2008 were associated with cheatgrass, suggesting a significant conversion to a grass-fire cycle in that region (Balch and others, 2013). Increases in fire sizes on the Snake River Plain have also been attributed to extensive cheatgrass invasion, combined with higher numbers of human-set fires and high winds across flatter terrain that generally promote larger fires (Knapp, 1998).

Fire Season

Fire season is a variable that is rarely analyzed in the scientific literature, partly because it has no standardized definition. Fire seasons typically are reported in broad terms, such as “summer-early fall,” but peak fire season is often reported as June–September throughout much of the western United States (Littell and others, 2009). According to the U.S. Department of Agriculture (USDA), Forest Service, wildfire seasons have expanded by 78 days since 1970 and, as a result, fire season is now referred to as “fire year.” At national and regional scales, studies suggest that the increasing prevalence of human-ignited wildfires and climate change are contributing to longer fire seasons and to increased duration of fire-weather conditions (Abatzoglou and Williams, 2016; Balch and others, 2017; Syphard and others, 2017). The median discovery date for human-started fires was more than 2 months earlier than lightning-started fires nationwide, and the most common day for human-starts was July 4 (Balch and others, 2017). Human fire ignitions also had a stronger influence on lengthening fire seasons than climate change (Syphard and others, 2017). For the North American desert region (which includes the Great Basin and much of the sagebrush biome), human-ignited fires expanded the wildfire season length by 230 percent (Balch and others, 2017).

In the western United States, Dennison and others (2014) did not find significant trends in large fire start dates between 1984 and 2011 across large ecoregions. However, a trend toward longer fire seasons and earlier large fire start dates in some ecosystems has been documented. This has occurred particularly in mid-elevation montane forests that were correlated with earlier spring snowmelt (Westerling and others, 2006). Specific to the sagebrush biome, significant increases in fire-season lengths were observed over a recent 30-year period for the Southern Great Basin, Wyoming Basin, and Great Plains (that is, the Silver Sagebrush) floristic provinces (Brooks and others, 2015). Increasing fire-season length in the Southern Great Basin may be of particular concern considering the relatively low resilience of sagebrush types to fire in that area. Fires starting earlier in the season are likely occurring with nonnative annual grass invasions as they dry out about a month earlier than most native herbaceous vegetation.

Fire Recurrence—Reburns

As changing fuels, ignition rates, and climate conditions promote greater annual and cumulative area burned and shorter fire intervals, the probability of specific parts of the landscape burning repeatedly also increases. As fire recurrence over a given time period increases, conditions become more suitable for the persistence of annual plants, such as cheatgrass, and less suitable for the persistence of woody perennials, such as sagebrush, resulting in a high probability of transitioning to a grass-fire cycle (D'Antonio and Vitousek, 1992). An influential study documented this dynamic in big sagebrush on the Snake River Plain (Whisenant, 1990), in which mean fire return intervals declined from an estimated 60 to 110 years historically to as short as 5 years or less during the 1960s through the 1980s. Consequently, many areas burned repeatedly and transitioned to cheatgrass-dominated systems with decreased native plant abundance and diversity (Whisenant, 1990).

Of the 1.4 million ha of recurrent fire area reported for the sagebrush biome during 1984–2015, roughly two-thirds of that area occurred in the Snake River Plain, constituting approximately 25 percent of that region's total fire area and approximately 8 percent of its big sagebrush area (Brooks and others, 2015). Most of that recurrent fire area burned twice (71 percent) resulting in an average fire return interval of 15 years for those areas, and the remainder (29 percent) burned three or more times for an average fire return interval of 7.5 years or less. However, the region with the highest percentage (34 percent) of its fire area classified as recurrent was the Columbia Basin, potentially indicating an even greater risk of conversion to a grass-fire cycle than the Snake River Plain. These two provinces have some of the highest proportions of landscapes with low resilience to fire and low resistance to cheatgrass invasions, especially in Wyoming big sagebrush and other low productivity sagebrush communities (Chambers and others, 2014a, b, 2017b). Low resistance makes these landscapes particularly vulnerable to ecosystem type conversion via the grass-fire cycle.

Human-Caused Wildfires

Human-caused ignitions account for thousands of wildfires each year across the western United States and well over half of all wildfires annually. Approximately 90 percent of wildland fires in the United States are caused by humans and, on average, humans ignite 61,375 wildfires per year (National Interagency Fire Center, 2019a). While a majority of these ignitions occur in the Southeast and California, human-caused fires across the sagebrush biome have increased substantially over the past two decades. For example, in 2018, human ignitions occurring on U.S. Department of the Interior (DOI), Bureau of Land Management (BLM) lands for 13 western States were responsible for 64 percent of all wildfires and 55 percent of acres burned (National Interagency Fire Center, 2019a).

There are many causes of human ignitions. Some human-caused fires are from campfires left unattended, target shooting, powerlines, fireworks, the burning of debris, and intentional acts of arson. Also, heat and sparks from vehicles and equipment can cause wildfires. Data from the DOI Wildland Fire Management Information (WFMI) system from 1997 to 2016 identifies the most common human causes of fires that burn on sagebrush habitats owned by the BLM. Although each region has its own unique set of wildfire causes, two common causes for human-caused fires are powerline failures in areas with improper clearance and roadside ignitions along highways and major roads bordered by hot, dry, fine fuels. Many of these fires occurred near wildland-urban interfaces and required a substantial fire suppression response. Such fires take firefighting resources away from fires occurring in sagebrush and other high-value resource areas, especially when multiple fire starts occur during high-wind or lightning events. Areas most at risk from human-caused fires are sagebrush communities with low resilience to fire and resistance to invasive annual grasses that are located near the wildland-urban interface.

Human-caused fires tend to ignite easily, spread quickly, and are difficult to control, especially in areas where continuous fuels from invasive annual grasses are present. Once areas are burned, options to protect and rehabilitate these sagebrush communities are limited, often resulting in dominance of invasive annual grasses postfire. This in turn often results in more human-caused fire ignitions. This invasive annual grass-fire cycle could be disrupted with a targeted fire-prevention program that is focused on the causes of human ignitions in sagebrush communities. These preventative actions can be more effective when tailored and delivered to local communities surrounding BLM districts and Forest Service lands.

While not all human-caused wildfires can be prevented, many can and are being prevented through enhancing the public's understanding of fire risk and encouraging the public to follow precautions while conducting activities that may start a fire (Butry and Prestemon, 2019). Recent social-science studies conducted over the past several years have focused on the public's perception of wildfire risk and the public's motivation to take action, especially at the community or individual level (McCaffrey and others, 2012; Hamilton and others, 2018; Meldrum and others, 2019). While general awareness campaigns are effective to help the public understand their risk from wildland fire, awareness does not necessarily lead to action.

Impact of Altered Fire Regimes on Sagebrush Communities and Postfire Recovery

Sagebrush community recovery from fire is highly variable because of vast differences between lower (characterized by Wyoming big sagebrush) and higher (characterized by mountain big sagebrush) elevation sagebrush communities, prefire community composition, site differences, and prefire and postfire weather (Maier and others, 2001; Ziegenhagen and Miller, 2009; Nelson and others, 2014). Furthermore, some areas may never recover from fire because climate change may render these environments less suitable for sagebrush (Bradley, 2010; Schlaepfer and others, 2015).

Lower elevation sagebrush communities are hotter and drier than higher elevation communities, and recovery from a fire is expected to be exceedingly slow in comparison (Winward and Tisdale, 1977; West and others, 1978; Winward, 1980). What information is available is relatively short term compared to how long it may take for sagebrush recovery in these hotter-drier sites. After 23 years, sagebrush recovery was only 2 percent in Wyoming big sagebrush communities in Montana (Lesica and others, 2007). Thus, when full recovery will occur is generally unknown and likely to vary by a suite of factors. For example, recruitment (germination and survival of seedlings) of sagebrush at lower elevations is greater with above-average cool season precipitation (Maier and others, 2001).

Lower elevation sagebrush communities also have a greater risk of postfire nonnative annual grass invasion and dominance than higher elevation big sagebrush communities (Chambers and others, 2014a). This risk is significantly greater if native perennial grasses have been reduced (Chambers and others, 2007). Thus, prefire composition of sagebrush communities is an important factor for determining postfire recovery. If native perennial grasses and forbs dominate the community prior to fire, they are likely to dominate the community after fire (Bunting, 1985; Rhodes and others, 2010; Bates and others, 2013). If native perennial grass and forb cover were low and nonnative annual grasses already existed in the community prior to the fire, nonnative annual grasses are likely to dominate the postfire community (Young and Evans, 1978; Hosten and West, 1994; Chambers and others, 2007).

Fuel loading (the amount of fuel available to burn) can also influence fire severity and postfire recovery in sagebrush communities. In Wyoming big sagebrush communities in Oregon, the accumulation of fine fuels on native perennial bunchgrasses increased fire-induced mortality of perennial grasses and led to a substantial postfire nonnative annual grass invasion (Davies and others, 2009, 2016a). Nonnative annual grass dominance of lower elevation sagebrush communities likely indicates a permanent shift in the plant community without additional inputs (D’Antonio and Meyerson, 2002; Bagchi and others, 2013). Substantial nonnative annual grass invasion prevents sagebrush re-establishment because it increases fire frequency to the point that sagebrush cannot reach maturity (that is, produce seed) before the next fire occurs. As a result, the sagebrush seedbank is depleted (D’Antonio and Vitousek, 1992; Rossiter and others, 2003). Nonnative annual grass competition for soil moisture can also prevent sagebrush establishment (Booth and others, 2003).

Increased fire frequency favors nonnative annual grasses and is detrimental to native floras that are not adapted to frequent fire (D’Antonio and Vitousek, 1992). This creates a positive feedback cycle between fire and continued nonnative annual grass dominance (grass-fire cycle) of the community (D’Antonio and Vitousek, 1992; Rossiter and others, 2003). Thus, the effects of increased fire frequency cannot be separated from the effects of exotic annual grass invasion. Nonnative annual grass invasion exponentially decreases plant community biodiversity and native perennial species abundance (Davies and Svejcar, 2008; Davies, 2011). Nonnative annual grasses use soil water earlier than native plants (Melgoza and others, 1990), resulting in vegetation drying out as much as a month earlier than it would have if nonnative grasses were not present (Davies and Nafus, 2013). This allows earlier season wildfires to occur (Davies and Nafus, 2013) at a time when native bunchgrasses are more susceptible to fire (Wright and Klemmedson, 1965; Britton and others, 1990; Davies and Bates, 2008). Frequent fire in lower elevation sagebrush communities results in a threshold being crossed to an annual grass-dominated state that has proven to be exceedingly difficult and expensive to reverse at a meaningful scale for conservation and land management (Davies and others, 2011; Miller and others, 2011). Therefore, there is a substantial risk that lower elevation sagebrush communities will not recover from fire.

Higher elevation, more mesic sagebrush communities (typically mountain big sagebrush) with moderate to high resilience to fire and resistance to annual invasive grasses are experiencing increased conifer encroachment owing to a generally decreased fire frequency. This is attributed to historical improper livestock grazing that reduced grass and forb fine fuels needed to carry fire through these communities (Miller and others, 2011). In some areas, most notably in the Great Basin, this has led to juniper and pinyon expansion from historically fire-safe sites to more productive sagebrush communities (Miller and Wigand, 1994; Gruell, 1999; Miller and Rose, 1999; Miller and Tausch, 2001; Miller and others, 2005; Romme and others, 2009), and tree density has increased in historically open savannah-like stands (Nichol, 1937; Johnson and Miller, 2008). Increasing conifer cover in sagebrush communities eliminates sagebrush and can significantly decrease the herbaceous understory (Blackburn and Tueller, 1970; Miller and others, 2000; Bates and others, 2005; Suring and others, 2005; Chambers and others, 2007).

Juniper and pinyon expansion can increase the risk of postfire annual grass invasion of these communities. The reduction in native perennial grasses and shrubs that coincides with conifer expansion increases the risk of nonnative annual grass invasion when the conifers are removed (Bates and others, 2013; Bates and others, 2017; Davies and others, 2019). Furthermore, once a conifer woodland has developed, the potential for a more severe fire is elevated because of increased fuel loads (Tausch, 1999; Miller, R.F., and others, 2008; Stebleton and Bunting, 2009). Higher severity fire in conifer woodlands where annual grasses are present increases the probability of nonnative annual grass dominance postfire (Bates and others, 2013). In higher elevation sagebrush communities, native perennial vegetation may be able to re-establish and subsequently limit nonnative annual grasses over time (Condon and others, 2011; Bagchi and others, 2013).

Sagebrush recovery after fire at higher elevations, in the absence of substantial nonnative annual grass invasion, is estimated to take from 15 to more than 100 years (Baker, 2006; Lesica and others, 2007; Ziegenhagen and Miller, 2009; Nelson and others, 2014). Recovery is more rapid with greater precipitation in the cool season following the fire (Nelson and others, 2014). Though with a limited sample pool of areas in recovery (n=9), sagebrush recruitment in higher elevations was greatest in years with below average spring precipitation (Maier and others, 2001). Most recovery estimates were derived from areas dominated by sagebrush prior to burning. The rate of recovery may be slower in areas where sagebrush has largely been excluded by conifer expansion, as seedbanks in these communities are likely limited (Bates and others, 2005; Davies and others, 2014a). In communities dominated by the expansion of conifers, postfire recovery rate of sagebrush decreases with increasing conifer dominance (Bates and others, 2013). Sagebrush postfire recovery is highly variable, but at cooler and wetter sites, it is likely to be more consistent and rapid, especially if conifer expansion has not appreciably reduced sagebrush prior to burning.

There is limited information on successional stages for sagebrush communities postwildfire, especially for shrub, forb, and grass species within sagebrush communities. Recovery of other shrubs in sagebrush communities is variable but most recover more rapidly than sagebrush. Other shrubs often recover rapidly after fire because of their sprouting ability. For example, if green or rubber rabbitbrush (*Ericameria teretifolia* and *E. nauseosa*, respectively) were part of the prefire community, they often increase in abundance and cover after fire (Beck and others, 2009; Davies and others, 2009). As sagebrush redominates the plant community, rabbitbrush is eventually outcompeted and reduced (Young and Evans, 1974). Other resprouters may also increase after fire. For example, prickly phlox (*Linanthus pungens*) increases after fire in sagebrush communities (Young and Evans, 1974). Antelope bitterbrush (*Purshia tridentata*) can sprout after fire or experience significant mortality depending on fire intensity, season, and other site-specific factors (Clark and others, 1982). The response of herbaceous understory vegetation to fire varies with differences in species composition, preburn site condition, fire intensity, and prefire and postfire patterns of precipitation.

Increased incidences of large and more complete fires (no patches of unburned areas within fire perimeter; Adams, 2013) potentially pose an additional challenge to timely natural recovery of sagebrush communities. Sagebrush seeds only disperse a few meters from the parent plant (Young and Evans, 1989), and a sagebrush seed source may be many kilometers away from the interior of large wildfires. Therefore, if sagebrush does not establish from seed in the first year or two postfire, the sagebrush seed bank will be depleted (Young and Evans, 1989; Wijayratne and Pyke, 2009), and sagebrush will have to disperse from the exterior of these large fires. Sagebrush establishment from the seedbank after wildfire seems moderate to exceedingly unlikely following the environmental gradient from cool and wet to hot and dry sagebrush communities (Baker, 2006; Lesica and others, 2007; Ziegenhagen and Miller, 2009; Nelson and others, 2014). How long it takes the sagebrush seedbank to disperse and establish into the interiors of these large fires is unknown. However, it is likely to significantly lengthen the time for sagebrush recovery.

Impacts of Altered Fire Regimes on Wildlife

Altered fire regimes have many implications for sagebrush wildlife species because the resulting landscape mosaic of burned and unburned areas affects wildlife habitat availability and connectivity. Owing to the delay in sagebrush recovery in some regions, large and frequent fires that lead to extensive loss of sagebrush cover will likely have negative effects on wildlife populations over longer periods of time (Longland and Bateman, 2002; Coates and others, 2015). In addition, remaining unburned areas may be too small to support the habitat requirements of some sagebrush-dependent wildlife species. At the same time, the lack of fire in other sagebrush-dominated regions has resulted in conifer expansion, which also limits habitat availability. Modern fire regimes with uncharacteristic fire intensity, size, and frequency, resulting in either too much or too little fire, pose a threat to many wildlife species (for example, sagebrush sparrow [*Artemisiospiza nevadensis*], sage thrasher [*Oreoscoptes montanus*], Brewer's sparrow [*Spizella breweri*], pygmy rabbit [*Brachylagus idahoensis*], and sage-grouse) that are dependent on sagebrush for their survival.

While there are a number of studies available on the impact of wildfire on many sagebrush-obligate species, these studies are often limited in scope and results likely vary over different spatial and temporal scales. Some studies have identified direct relationships between sagebrush obligates and wildfire, but findings are often limited in their scope to local sites (Connelly and others, 2000b), movements and habitat associations (Fischer and others, 1996, 1997; Nelle and others, 2000; Rhodes and others, 2010), relatively short timeframes (less than 10 years; Blomberg and others, 2012), habitat suitability (Davis and Crawford, 2015), and simulations (Pedersen and others, 2003). However, a few studies have examined the long-term effects of wildfire on sagebrush-obligate species across large spatial scales. For example, Coates and others

(2015) demonstrated adverse long-term effects of wildfire on greater sage-grouse population growth rates across the Great Basin and highlighted the potential threat of uncharacteristic high-frequency fire regimes.

Many studies suggest that large-scale changes in low-elevation sagebrush habitat associated with fire have had a negative influence on sagebrush-obligate species. Although the majority of studies addressing the effects of fire on sagebrush bird communities have been short-term (less than 5 years; Knick and others, 2005), most studies have found negative effects of fire on population trends and abundance for sagebrush-obligate avifauna, including the sagebrush sparrow (Welch, 2002; Reinkensmeyer and others, 2007; Earnst and others, 2009; Holmes and Robinson, 2013), Brewer's sparrow (Castrale, 1982; Bock and Bock, 1987; Knick and Rotenberry, 1999; Noson and others, 2006; Holmes, 2007), sage thrasher (McIntyre, 2002; Welch, 2002; Noson and others, 2006; Holmes, 2007), and gray flycatcher (*Empidonax wrightii*; Welch, 2002; Holmes and Robinson, 2013). Large-scale sagebrush removal resulting from fire can result in significant declines in sagebrush-obligate bird species (Magee and others, 2011). Brewer's sparrows and sage thrashers need large, unfragmented sagebrush areas (Kerley and Anderson, 1995; Knick and Rotenberry, 1995) that are dependent on infrequent fire regimes. The long-term effects of fire on a majority of these species is relatively unknown; however, Holmes and Robinson (2013) found that the impact of fire on bird abundance in mountain big sagebrush communities persisted for at least two decades.

Wildfires can impact pygmy rabbits directly through mortality and indirectly through habitat modification by depletion of concealment cover and food resources, fragmentation of sagebrush habitat, and facilitating the invasion of nonnative plants (U.S. Department of the Interior, 2010b; Hayes, 2018). Recolonization of burned areas by pygmy rabbits likely depends on fire intensity and size of area burned. Pygmy rabbit populations might persist following fires if the fire is small and creates a mosaic of habitat, if the surrounding habitat is maintained, and if enough individuals survive the fire to reestablish the population (U.S. Department of the Interior, 2010b).

More than 181,000 ha (448,000 acres), or 6.4 percent of areas pygmy rabbits are known to have occupied since 2000 (minimum occupied areas [MOAs]; Smith and others, 2019), have burned (fig. J3). During that same period, 1.1 million ha (2.8 million acres), or 7.8 percent of areas modeled as highly suitable habitat for pygmy rabbits (Smith and others, 2019), have burned (fig. J3). The MOAs likely underestimate the true extent of pygmy rabbits, since areas that were not searched are not included and modeled habitat likely overestimates occupied habitat. Regardless, this range from 6.4 to 7.8 percent of pygmy rabbit habitat burned likely brackets the true range of impact. Almost all of this impact occurred within the Great Basin; since 2000, 12.7 percent of MOAs burned within the Great Basin versus 0.3 percent of MOAs outside the Great Basin. Similarly, 17 percent of highly suitable acres for pygmy rabbits burned within the Great Basin versus 1.4 percent of highly suitable acres outside the Great Basin during this period.

While the specific effects of wildfire on other sagebrush wildlife species (for example, big game species, other small mammals, and sagebrush-dependent amphibians and reptiles) remains unclear, the response of sagebrush-obligates to other forms of habitat disturbance may give some indication of how these species will respond to wildfire-caused habitat disturbance. Widespread sagebrush removal treatments (including fire, mechanical, or chemical treatments) that reduce shrub dominance or reduce fine fuels in sagebrush communities can result in significant declines in sagebrush-obligate bird species (Magee and others, 2011) and may be detrimental to pygmy rabbits owing to their reliance on sagebrush (Wilson and others, 2011; Woods and others, 2013). Additionally, many native small mammals may be at risk of extirpation owing to fragmentation of sagebrush habitats (Hanser and Huntly, 2006). For example, habitat loss and fragmentation resulting from wildfire, invasion of introduced annual grasses and weed species (especially cheatgrass and prickly Russian thistle [*Salsola tragus*]) and conifer encroachment have negatively impacted the dark kangaroo mouse (*Microdipodops megacephalus*) through loss of connected populations (Hafner and Upham, 2011). Many populations of dark kangaroo mouse in the northern part of the Great Basin are either locally extinct or facing threats because of loss of habitat (Hafner and Upham, 2011). Populations in Idaho are considered at extreme risk owing to their restricted distribution in that State and the potential for wildfire based on the presence of invasive annual grasses (Idaho Department of Fish and Game, 2017).

Decreased prevalence of fire in higher-elevation sagebrush (owing to its higher resilience and resistance) also may pose a threat to sagebrush wildlife species in parts of their range, especially in mountain big sagebrush communities, which are more productive and include more perennial grasses and perennial forbs than hotter-drier, low elevation regions (Davies and Bates, 2010). Reduction of fuels caused by livestock grazing and fire-suppression activities is thought to have increased fire-return intervals to the point where pinyon-juniper communities can establish and eventually outcompete sagebrush. This will lead to a reduction of perennial grasses and forbs (Miller and Tausch, 2001), affecting sagebrush habitats for certain sagebrush-dependent species (Miller and Rose, 1999; Miller and Heyerdahl, 2008). For example, sage-grouse avoid areas with trees (Casazza and others, 2011). Thus, pinyon-juniper expansion can contribute to reduced sage-grouse population persistence (Baruch-Mordo and others, 2013). Conifer expansion resulting in sagebrush habitat loss also has negative consequences for high-elevation, mountain big sagebrush bird communities and may negatively impact other sagebrush obligates, including Brewer's sparrow, sage thrasher (Noson and others, 2006), and pygmy rabbit (Woods and others, 2013; chap. M, this volume).

Fires also influence invertebrate food sources (Schroeder and others, 1999) across all sagebrush communities. Ants (*Hymenoptera*), grasshoppers (*Orthoptera*), and beetles (*Coleoptera*) are an essential component of certain wildlife diets (for example, Johnson and Boyce, 1991). The abundance of arthropods did not decline following wildfire in mountain big sagebrush communities (Davis and Crawford, 2015), and Pyle and Crawford (1996) reported no apparent effect to beetles from prescribed burning. In contrast, some arthropod orders increased, and others decreased following prescribed fire in mountain big sagebrush communities (Davies and others, 2014a). The abundance of insects was significantly lower 2 to 3 years following fire in a Wyoming big sagebrush–threetip sagebrush (*A. tripartita*) community (Fischer and others, 1996). The abundance of beetles and ants was significantly greater 1 year after a burn in mountain big sagebrush communities and returned to preburn levels by years 3 to 5 (Nelle and others, 2000). The effect of fire on insect populations likely varies because of a host of environmental factors. Because few studies have been performed and the results of those available vary, the specific magnitude and duration of the effects of fire on insect communities is still uncertain.

Wildfire is considered the largest threat across the western part of the sage-grouse range, particularly in the Great Basin (Brooks and others, 2015; Coates and others, 2015). Fire occurring within the range of sage-grouse can cause direct loss of habitat, resulting in negative effects to breeding, feeding, and sheltering opportunities for the species (Call and Maser, 1985). In addition to the direct habitat loss, fire can also create a functional barrier to sage-grouse movements and dispersal that compounds the influence wildfire can have on populations

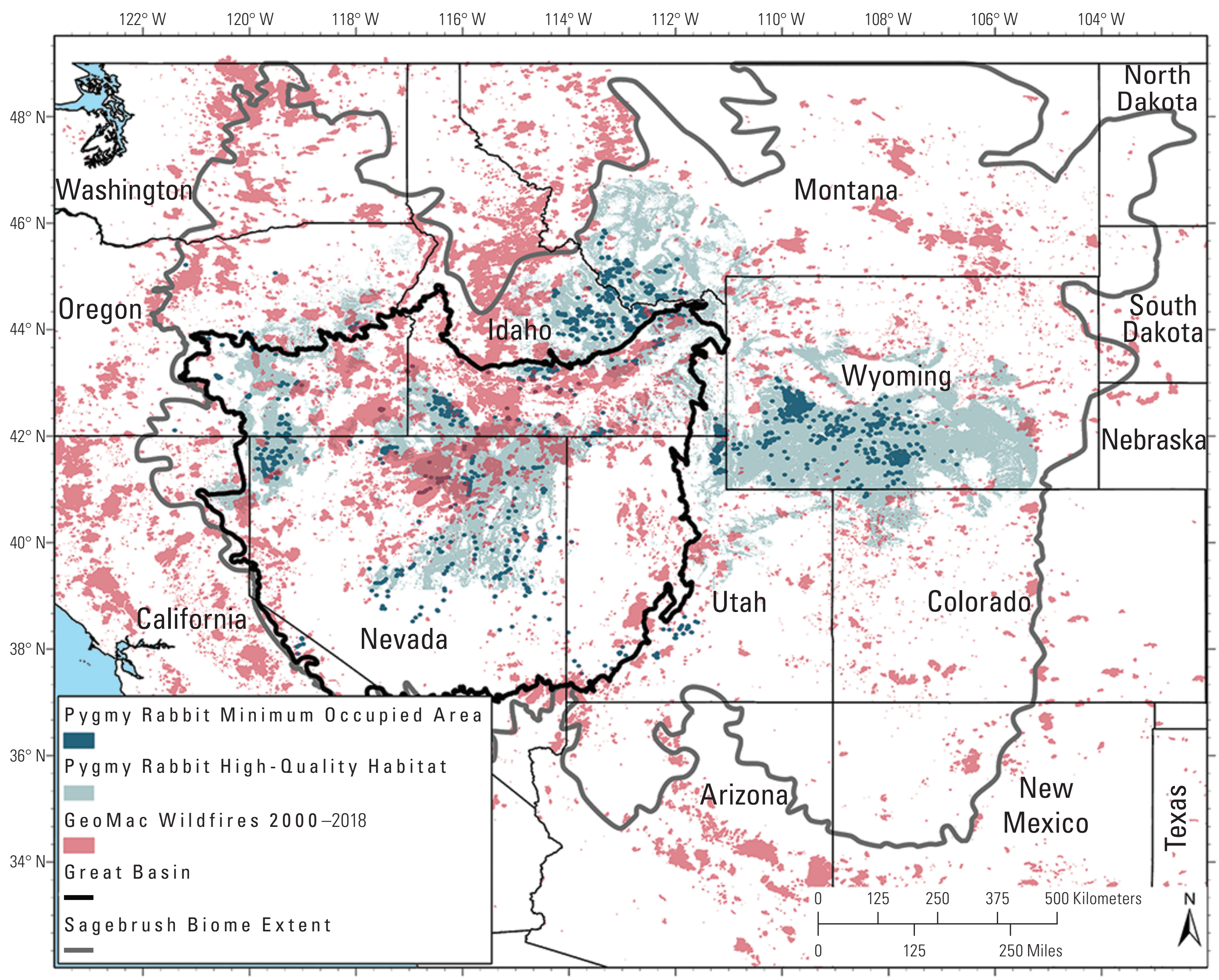

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Figure J3. Wildfires in and around the sagebrush (*Artemisia* spp.) biome from 2000 to 2018 and pygmy rabbit (*Brachylagus idahoensis*) minimum occupied areas and high-quality habitat (Smith and others, 2019). Wildfire information from U.S. Geological Survey (2019a).

and population dynamics (Fischer and others, 1997). In some cases, fire can isolate sage-grouse populations, thereby increasing their risk of extirpation (Knick and Hanser, 2011; Wisdom and others, 2011). While the direct loss of habitat from fire has been shown to be a significant factor associated with population persistence, the indirect effect posed by loss of connectivity among sage-grouse populations may greatly expand the influence of the threat of habitat loss beyond the physical fire perimeter.

Fire is one of the primary factors linked to population declines of sage-grouse because of long-term loss of sagebrush and conversion to invasive annual grasses (Beck and others, 2009; Johnson and others, 2011; Knick and Hanser, 2011). The extent and abundance of sagebrush habitats, the proximity to burned habitat, and the degree of connectivity among sage-grouse populations all affect sage-grouse persistence (Johnson and others, 2011; Knick and Hanser, 2011; Wisdom and others, 2011). Fire has been found to have negative effects on spring lek counts, recruitment rates, rates of population change (Connelly and others, 2000b; Blomberg and others, 2012; Coates and others, 2015), and sage-grouse survival (Lockyer and others, 2015), resulting in lek extirpation (Johnson and others, 2011; Knick and Hanser, 2011). For example, small increases in the amount of burned habitat surrounding a lek had a large influence on the probability of lek abandonment (Knick and Hanser, 2011). Abandonment of leks owing to fire has been documented (Hulet, 1983; Connelly and others, 2000a). Additionally, fire had a negative effect on subsequent trends in lek counts in the Snake River Plain and Southern Great Basin (Johnson and others, 2011). In southeastern Idaho, sage-grouse populations were generally declining across the entire study area, but declines were more severe in postfire years (Connelly and others, 2000b). Further, a recent study in Oregon (Foster and others, 2019) found that sage-grouse continued to use fire-affected habitat in the years immediately following wildfire, which appeared to have had an acute fitness cost. Sage-grouse experienced lower nest and adult female survival during the first 2 years postfire (Foster and others, 2019), which has likely contributed to the observed declines in sage-grouse population trends following wildfire across the Great Basin (Coates and others, 2015).

Throughout the breeding season, herbaceous understory vegetation plays a critical role as a source of forage and cover for sage-grouse females and chicks. The response of herbaceous understory vegetation to fire varies with differences in species composition, preburn site condition, fire intensity, and prefire and postfire patterns of precipitation. Although fire has been shown to promote recovery and extend the period of active growth in forbs known to be important in the diet of sage-grouse (Wrobleski and Kauffmann, 2003; Beck and others, 2009; Davis and Crawford, 2015), any short-term flush of understory perennial grasses and forbs within burned sites is essentially lost only a few years postfire (Cook and others, 1994; Fischer and others, 1996; Nelle and others, 2000; Paysen and others, 2000; Wambolt and others, 2001). Thus, any short-term benefits gained by releasing understory vegetation from competition with a shrub overstory are negated by the loss of overstory structure essential to sage-grouse life-history needs. For example, prescribed fires in mountain big sagebrush at Hart Mountain National Antelope Refuge in Oregon caused a short-term increase in certain forbs, but reduced sagebrush cover, making habitat less suitable for greater sage-grouse nesting (Rowland and Wisdom, 2002).

Small fires may maintain a suitable habitat mosaic by reducing shrub encroachment and encouraging understory growth. However, without nearby sagebrush cover, the utility of these sites is questionable (Woodward, 2006). Disturbances, such as fire, that remove sagebrush extent and limit sage-grouse habitat availability (cover and forage) appear to strongly influence the probability of local sage-grouse population persistence (Hess and Beck, 2012).

The few studies that have suggested fire may be beneficial for sage-grouse were primarily performed in mesic areas used for brood-rearing (Klebenow, 1970; Gates, 1983; Sime 1991; Pyle and Crawford, 1996; Connelly and others, 2000a, b). In mesic habitats, small fires may maintain a suitable habitat mosaic by reducing shrub encroachment and encouraging understory growth. However, unless sagebrush cover is available nearby, the utility of these sites for sage-grouse is debatable (Woodward, 2006).

The frequency, size, and severity of fires is increasing, and fire has cumulatively removed a significant and growing amount of sage-grouse habitat in the last 10 years (table J1), particularly in the Great Basin. Since 2000, more than 20,000 square kilometers (km^2; 5 million acres), or 20.6 percent, of greater sage-grouse priority habitat management areas (PHMAs) within the Great Basin have burned (fig. J4). In 2018 alone, 3.6 percent of PHMAs within the Great Basin were lost to fire. Outside the Great Basin, only 2.1 percent of PHMAs have burned since 2000 (fig. J4). A fire-threats assessment indicates that threats of too much fire are higher in four western greater sage-grouse management zones (MZs; derived by Stiver and others, 2006) than in three eastern MZs (Brooks and others, 2015), raising concern in the western region of sage-grouse range. Among the four western MZs, the Snake River Plain and the Columbia Basin ranked somewhat higher than the Southern Great Basin and Northern Great Basin in terms of loss of sage-grouse habitat owing to fire. Overall, these findings corroborate models that projected approximately one-half of the current population of sage-grouse will remain in the Great Basin by the mid-2040s if current fire trends continue unabated (Coates and others, 2015).

Collectively, these findings illustrate how sage-grouse habitat and population persistence may be compromised as sagebrush ecosystems become more impacted by fire, and increasingly invaded by annual invasive grasses, at least in the western part of the species' range. Increased management of invasive plant infestations and wildfire suppression could reduce the rate of decline depending upon the success rate of the management approach; however, Coates and others (2015) did not consider the impact of postwildfire restoration projects, which could further reduce the rate of population decline. The projected future impact of fire on sage-grouse population trends likely also depends upon climatic conditions (Coates and others, 2015), which is difficult to forecast with certainty 30 years into the future.

Wildlife Migratory Corridors

The mapping of migration routes, genetic connectivity, and movement pathways has been done for a few sagebrush wildlife species (for example, Knick and others, 2013, 2014b; Copeland and others, 2014; Crist and others, 2017; Cross and others, 2018). However, we do not have a firm understanding of the ways that wildfire affects movements by sagebrush-obligate wildlife as they meet their life-history needs. Large-scale habitat changes because of wildfire and invasive plant species likely degrade habitat and have the potential to alter or disrupt important movement pathways, including causing losses to established migration routes to winter ranges and stopover habitat(s). Large-scale habitat changes likely serve as a limiting factor to sagebrush-obligate populations.

Conservation of terrestrial migrants (for example, elk [*Cervus canadensis*], mule deer [*Odocoileus hemionus*], and pronghorn [*Antilocapra americana*]) presents a unique challenge across the sagebrush biome because connectivity of the entire migration route must be maintained since barriers to movement anywhere within the migration corridor could render it unviable (Copeland and others, 2014). Although several human threats to migrating ungulates are known (for example, energy and residential development, roadway mortality, and fencing; Harrington and Conover, 2006; Grovenburg and others, 2008; Sorensen and others, 2008; Sawyer and others, 2012, 2013; Lendrum and others, 2013), little information on the impacts of wildfire on migration routes and seasonal movements of migratory ungulates is available.

Table J1. Number of hectares (acres) of greater sage-grouse (*Centrocercus urophasianus*) range burned by wildfires by State, 2012–2018.

[CA, California; CO, Colorado; ID, Idaho; MT, Montana; NV, Nevada; ND, North Dakota; OR, Oregon; SD, South Dakota; UT, Utah; WA, Washington; WY, Wyoming; --, no data]

State	2012	2013	2014	2015	2016	2017	2018	Total[1]	Percent[2]
CA	104,597 (258,459)	0	--	6,546 (16,175)	2,082 (5,145)	35,835 (88,548)	9,292 (22,961)	158 (390)	29.4
CO	1,926 (4,759)	499 (1,233)	8,165 (20,176)	1,359 (3,358)	1,301 (3,215)	11,242 (27,779)	16,682 (41,221)	41 (101)	2.6
ID	212,435 (524,927)	92,333 (228,155)	15,189 (37,532)	105,595 (260,925)	42,431 (104,847)	101,755 (251,437)	195,045 (481,956)	765 (1,890)	12.6
MT	111,364 (275,180)	643 (1,589)	4,949 (12,229)	4,947 (12,224)	5,793 (14,315)	144,851 (357,937)	15,078 (37,258)	288 (712)	2.1
NV	179,462 (443,451)	11,492 (28,397)	7,749 (19,148)	4,951 (12,234)	87,037 (215,068)	391,462 (967,303)	416,950 (1,030,283)	1,099 (2,716)	13.2
ND	0	0	--	0	0	0	0	0	0
OR	411,817 (1,017,600)	49,183 (121,534)	186,780 (461,543)	73,256 (181,016)	45,228 (111,758)	42,454 (104,904)	7,911 (19,548)	817 (2,019)	13.6
SD	5 (12)	0	--	674 (1,665)	0	0	0	1 (2)	0.1
UT	15,374 (37,989)	7,774 (19,210)	2,953 (7,297)	153 (378)	13,463 (33,267)	37,755 (93,295)	57,864 (142,982)	135 (334)	4.6
WA	--	--	2,635 (6,511)	21,841 (53,969)	33,787 (83,488)	45,817 (113,214)	65,894 (162,824)	170 (420)	--
WY	56,962 (140,753)	971 (2,399)	241 (596)	8,408 (20,776)	22,319 (55,150)	28,089 (69,408)	38,542 (95,237)	156 (385)	0.9
Total	**1,093,942 (2,703,131)**	**162,895 (402,514)**	**228,661 (565,021)**	**227,731 (562,723)**	**253,442 (626,255)**	**839,262 (2,073,816)**	**823,260 (2,034,275)**	**3,629 (8,967)**	**6.3**

[1]In thousands.

[2]Percent of greater sage-grouse priority habitat management areas plus general habitat management area within each State.

Impacts of Prescribed Fire on Wildlife

Prescribed fire is beneficial when used to reduce fire risk around housing developments, to restore fire in sagebrush communities that may have missed fire cycles, and to reduce conifer expansion. Although there remains the potential for future use of prescribed fire (or other methods of sagebrush treatment), additional studies are needed to elucidate the potential long-term benefits or negative impacts of prescribed burning on sagebrush ecosystems. Decisions on habitat manipulations should not be made without considering impacts to sagebrush-obligate species.

Land managers use prescribed fire to obtain desired management objectives for domestic livestock and a variety of wildlife species. While the efficacy of prescribed fire in sagebrush habitats to enhance sagebrush-obligate populations is poorly understood (Peterson, 1970; Swenson and others, 1987; Connelly and others, 2000b; Nelle and others, 2000), as with wildfire, an immediate and potentially long-term result is the loss of sagebrush habitats (Beck and others, 2009). For example, small prescribed fires directly decrease habitat for sagebrush wildlife species, such as Brewer's sparrows and sage thrashers (Castrale, 1982; Kerley and Anderson, 1995).

There is limited evidence linking prescribed fire with immediate benefits to sage-grouse, particularly in Wyoming big sagebrush habitats (Fischer and others, 1996; Wambolt and others, 2001; Beck and others, 2009). For example, prescribed burns did not improve brood-rearing habitat in Wyoming

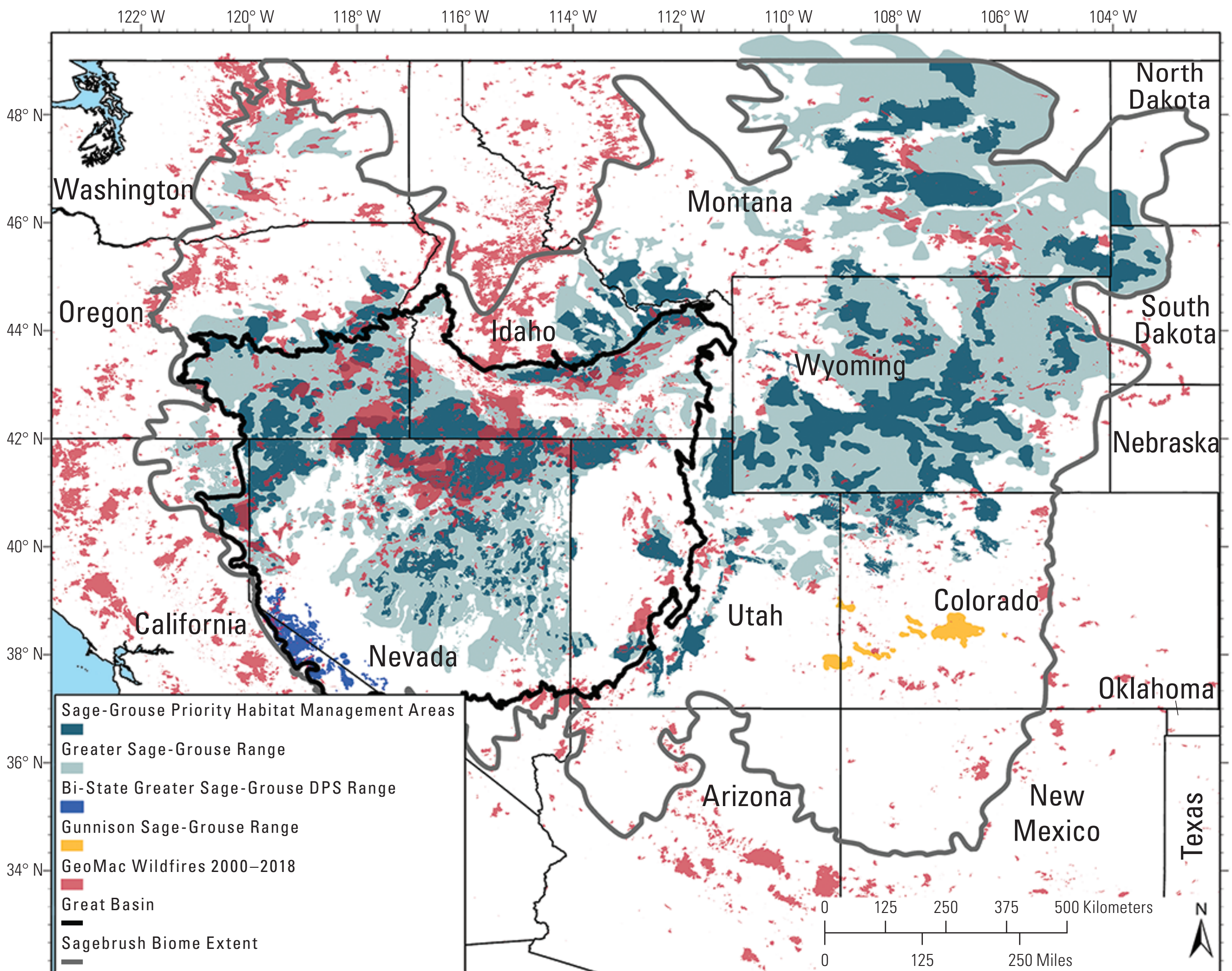

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Figure J4. Wildfires that burned from 2000 to 2018 within sage-grouse (*Centrocercus* spp.) range and greater sage-grouse (*C. urophasianus*) priority habitat management areas. Wildfire information from U.S. Geological Survey (2019a). Greater sage-grouse data obtained from U.S. Department of the Interior (2014). The Gunnison sage-grouse (*C. minimus*) range used data from Schroeder and others (2004). Priority habitat management area data obtained from the Bureau of Land Management (2019b). DPS, Distinct Population Segment.

big sagebrush, as forbs did not increase and populations of beneficial ant species (*Formicidae* spp.) declined (Fischer and others, 1996; Connelly and others, 2000b). Hence, fires in these locations appear to negatively affect brood-rearing habitat rather than improve it (Connelly and Braun, 1997; Knick and others, 2005). However, while results immediately following prescribed burns may not be beneficial for sage-grouse and other sagebrush obligates, these habitats will likely recover over time after prescribed fire. Their recovery will likely improve habitat conditions for sagebrush obligates over the long term (Boyd and others, 2014a). Prescribed fire can be particularly valuable at maintaining sagebrush dominance over the long term, even though it results in short-term losses in areas being encroached by conifers (Davies and others, 2019).

Impacts of Altered Fire Regimes on Ecosystem Services

Beyond the direct costs of fire suppression, altered fire regimes can affect people in three major ways: (1) impacts to sagebrush communities and wildlife; (2) changes in critical services that people rely on for health and survival; and (3) impacts to recreational and cultural resources (table J2). Few studies quantify the effects of wildfires on ecosystem services (the way natural systems provide benefits to people, such as forage for livestock, water filtering, and so on [Boyd and Banzhaf, 2007; Brown and others, 2007; Venn and Calkin, 2008, 2011; Milne and others, 2014]), and even fewer consider sagebrush ecosystems. Estimates for 1 year of damages from the 2013 Rim Fire, which burned 103,055 ha (254,654 acres) of mostly forested land, were between $100 million and $736 million (Batker and others, 2013).

Fire impacts on wildlife populations and plant communities can result in indirect impacts on people if society values the existence of the species or communities affected. Economists recognize nonuse existence values, which refer to the benefits society derives from the survival of a species or plant community, independent of any associated active uses (for example, Freeman, 2003; Segerson, 2017). No published study estimates the existence value of sage-grouse or other sagebrush-obligate species (Eiswerth and van Kooten, 2009), but the nonmarket value of protecting habitat for other charismatic bird species has been estimated between approximately $15 and $60 per household, depending on the species (Richardson and Loomis, 2009). While values vary with contexts, similar values might be expected for the existence not only of sagebrush-obligate species but also of native plants.

Fire can also affect the availability of forage for livestock. Ranchers tend to face higher costs if they must adjust forage following a fire, and the likelihood of ranchers going out of business goes up with increasing fire frequency (Brunson and Tanaka, 2011). These costs can be significant; one case study of 647,520 ha (1.6 million acres) burned in northern Nevada in 1999 estimated $12.8 million in losses from lost livestock output, livestock deaths, and damage to fences (Riggs and others, 2001).

Wildland fires release smoke and carbon into the atmosphere. Exposure to wildfire smoke affects human health and welfare directly through induced illness. Wildfire smoke also incurs opportunity or real costs associated with behaviors taken to avoid that exposure, such as deferred recreation or exercise, increased air conditioning usage, and so on. While the cost of smoke-induced illness has been estimated at about $10 per exposed person per day, studies have also found that people are willing to pay between about $85 and $130 per day for a reduction of 1 day of smoke-induced symptoms (Richardson and others, 2012; Jones, 2018). The impact of altered fire regimes on carbon cycles corresponds to the net effect of carbon released into the atmosphere and the subsequent carbon sequestration during plant regrowth. Between 7 and 97 metric tons of carbon can be stored annually per acre of shrub/scrub ecosystems (Lacelle, 1997; Wilson, 2010), resulting in between $359 and $4,880 annual per acre benefits from carbon storage based on the $50.25 per metric ton estimate of the social cost of carbon (Interagency Working Group on the Social Cost of Carbon, 2013).

However, the long-term net effect of fires on total ecosystem storage of carbon in sagebrush ecosystems is not well understood, especially regarding the long-term response

Table J2. Summary of main potential effects of wildfire on the nonmarket goods and services provided by sagebrush (*Artemisia* spp.) ecosystems (Modified from Venn and Calkin, 2011, table 1).

Nonmarket good or service	Potential impacts to people
Wildlife and plant communities	Threats to existence values of species and communities
Livestock forage	Changes in forage availability affecting ranchers
Air quality	Induced illness from exposure to wildfire smoke
Carbon sequestration and storage	Release of stored carbon into atmosphere
Soil erosion	Sedimentation of water resources
Recreation opportunities	Changes to aesthetics of recreation areas
Cultural heritage	Changes in fire's role in cultural traditions and practices Damage to culturally important artifacts and sites

to multiple fires or ecosystem transition to other states (Miller and others, 2013). The available evidence suggests long-term net effects of fire on carbon cycles in sagebrush ecosystems are small in general, with most sagebrush ecosystems remaining carbon sinks because only small amounts of total ecosystem carbon are lost during fires in sagebrush ecosystems (Miller and others, 2013), and carbon fluxes and stocks tend to recover rapidly after fire in sagebrush ecosystems (Fellows and others, 2018; Flerchinger and others, 2020). However, certain environmental conditions can make some sagebrush communities carbon neutral (Flerchinger and others, 2020) and can turn communities dominated by annuals (cheatgrass and mustard) into net carbon sources (Prater and others, 2006).

Following a wildfire in sagebrush ecosystems, runoff and erosion by water may increase between 3 to 125 times, depending on plot scale and other contextual considerations (Miller and others, 2013). This in turn can lead to sedimentation of water resources, debris flows, and chemical water quality changes, and can potentially result in accruing reservoir dredging costs, infrastructure damage, and increased water treatment costs. Cost estimates for addressing sediment range widely from about $5 to $100 per cubic meter removed (American Society of Civil Engineers Task Committee, 1997; Jones and others, 2017). Water treatment costs increase by 0.19 percent with a 1 percent increase in turbidity, and a 1 percent increase in total organic carbon (TOC) increases water treatment costs by 0.46 percent (Warziniack and others, 2017). The costs resulting from erosion and debris flows also depend on the hydrology and climate of the affected area, with specific impacts depending on the magnitude of precipitation after a fire and on downstream values, such as infrastructure protection and fishery habitats (Haas and others, 2016; Jones and others, 2017).

Altered fire regimes can affect recreation through three main mechanisms: (1) changes to the aesthetics of recreation areas; (2) closures of trails and recreation areas; and (3) changes to opportunities for hunting and wildlife viewing. Changes to aesthetics can have an ambiguous effect on people’s usage of the burned areas; in some cases, recreationalists actually prefer to visit novel landscapes, including those affected by fire, whereas in other cases, recreationalists will avoid these burnt landscapes (Englin and others, 1996; Loomis and others, 2001; Hesseln and others, 2003, 2004). Time spent hunting can produce significant economic benefits but with large variability based on species and location; estimates range from about $40 per day for small game species to hundreds of dollars per day for big game species (Huber and others, 2018). Mean values for wildlife viewing, hiking, and off-highway vehicle use are of similar magnitudes—approximately $60, $78, and $76 per day, respectively (Rosenberger, 2016). However, simply multiplying an estimate of typical usage levels, absent a fire, by a per-use economic value would likely overestimate damages from a site closure because many would-be visitors might still recreate, albeit at different locations.

Finally, altered fire regimes can impact cultural heritage through changes to cultural traditions and practices, and potential damage of culturally important artifacts and sites. Damage to culturally important artifacts and sites could amount to substantial losses, particularly if irreplaceable resources are threatened. However, such impacts to cultural heritage are not particularly amenable to valuation. Few studies attempt to value cultural assets, and most that do focus on historical buildings, monuments, and artifacts (Venn and Calkin, 2011). Even if estimates from other contexts existed, benefit transfer is precluded in most cases because, almost by definition, important cultural sites cannot be substituted with other sites.

Fire-Suppression Costs

As fire seasons get longer, potentially more hectares will burn (fig. J5). Nationally in 2018, fires burned 3.54 million ha (8.77 million acres), which is 809,400 ha (2 million acres) more than the 10-year average. Within the Great Basin States of Idaho, Nevada, Utah, and extreme western Wyoming, 679,896 ha (1.68 million acres) or 80 percent of the 849,870 ha (2.1 million acres) burned in 2018 were in areas identified as sage-grouse habitat, meaning high-quality sagebrush.

As fires become more severe and consume larger acreages, fire-suppression costs rise. In 2017, fire-suppression costs were the highest on record for both the USDA Forest Service and the DOI agencies, exceeding $2.9 billion (Levy, 2018), more than five times the amount spent in 1985 when adjusted for inflation (see fig. J6 for DOI expenditures, exclusive of USDA Forest Service).

There are several factors leading to the rising costs of combating wildfires. Longer summers, shorter winters and springs, rising temperatures, and persistent drought have contributed to hotter-drier conditions that have lengthened the wildfire season.

For the sagebrush biome, however, the cost of fire suppression increases because of the proliferation of nonnative annual invasive grasses, which allows fire to spread rapidly and to resist suppression efforts. Fine fuel loadings may increase as much as 200–300 percent, as was the case in 2018 in the northern Great Basin (Newmerzhycky and Law, 2018). Combined with substantial carryover of fine fuels from the year prior, hot temperatures, low relative humidity, and windy conditions, fire behavior becomes explosive. Extreme fire behavior has been observed on recent fires in these fuel types with several fires growing more than 8,000 ha (20,000 acres) in a 24-hour burn period. Rapid rates of spread and high flame lengths often prevent a direct-attack strategy and greatly reduce the effectiveness of fuel breaks because conditions are too dangerous to place resources.

Often in the remote and rugged topography of the West, fire expands beyond conditions for initial attack because of the considerable amount of time required for resources to reach a reported fire. This can lead to the need for significant additional aerial and ground resources. For example, in July 2018, the Martin Fire, the largest single fire in Nevada's history, burned more than 176,000 ha (435,000 acres) within a 5-day period. There have been numerous other large sagebrush rangeland fires, each burning thousands of acres and costing millions of dollars.

Congress funds the annual fire-suppression accounts based on a rolling 10-year average, but with the consistently rising costs of suppression, Congressional funding is not enough during the fire season. The deficit is covered by transferring funds from other programs to the suppression account, known as "fire-borrowing," thereby making fewer and fewer funds and resources available to the very programs meant to proactively reduce the threat of wildfire and creating a perpetual cycle. Congress recently approved a measure to stop this fire-borrowing beginning in 2020 by raising the funding cap on suppression and allowing Federal agencies to tap into Federal Emergency Management Agency funds to fight catastrophic fires.

Burned Area Emergency Stabilization and Rehabilitation Costs

The U.S. Department of the Interior, BLM Emergency Stabilization (ES) and Burned Area Rehabilitation (BAR) programs assess damage and potential risks to landscapes damaged by wildland fires and identify and develop rehabilitation treatments to reduce or eliminate those risks. Collectively, the ES and BAR programs make up postwildfire recovery programs (referred to as "Emergency Fire Stabilization and Rehabilitation" [ESR]) whose purpose is to reduce the risk of resource damage caused by wildfire and promote recovery objectives. Although the ES and BAR programs are specific to Federal lands, the agencies work collaboratively with States and private landowners to leverage other funding sources for rehabilitation efforts and work across jurisdictional boundaries (for example, Utah's Watershed Restoration Initiative). All proposed and funded ES and BAR projects are reviewed and monitored for consistency with project objectives.

The ES program funds efforts to prevent further degradation of natural and cultural resources and to protect life, property, and other valuables. Funding for ES treatments are for no more than 1 year and are based on 10 percent of the DOI's 10-year rolling suppression expenditure average. However, to maximize opportunities for success, funding may extend into the second fall planting season after the wildfire. The BAR program

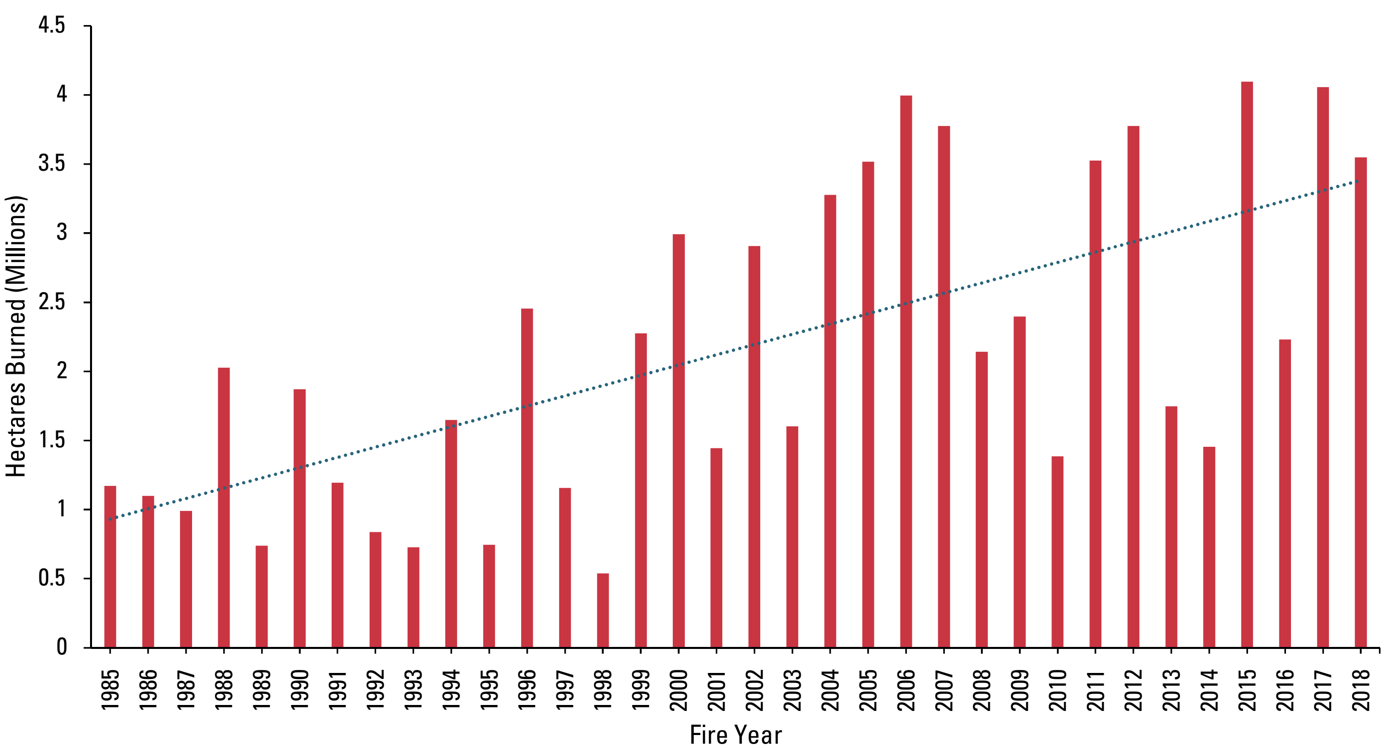

Figure J5. Millions of hectares burned annually by fire from 1985 to 2018 within the United States. The red bars represent millions of hectares burned in a given year. The dotted blue line represents the linear trendline of hectares burned for the time period. Data obtained from National Interagency Fire Center (2019b).

funds efforts to protect resources by restoring landscapes that are unlikely to recover naturally to management-approved conditions, consistent with land and resource management plan objectives. The BAR funding, which varies each year, is received as a line item in the wildland fire management budget and is available for up to 5 years from the wildfire-ignition date.

The BLM treats an average of 1 million ha (2.5 million acres) per year at a cost of $32.6 million dollars (fig. J7). However, it is difficult to use a cost-per-acre metric when attempting to compare ESR projects in a meaningful manner because the cost varies owing to differing needs for rehabilitation from region to region. For example, an ESR plan may only propose rebuilding some burned fences and some periodic monitoring, costing less than $16/ha (less than $40/acre), whereas another ESR plan is more intensive, proposing installation of numerous features to reduce soil erosion, chemical spraying, seeding (aerial and ground), planting seedlings, and more intense or frequent monitoring. Some of these plans may cost over $81/ha ($200/acre), and estimated costs for acres treated include a combination of new projects plus the continuation of projects from previous years. Furthermore, the acres treated each year are limited to available funding per year and not reflective of the acres actually needing ESR, which are increasing (fig. J5).

Other Costs Associated with Wildfire

In addition to fire-suppression costs, changes in ecosystem services, and costs for ESR, wildfires can also induce numerous other costs largely beyond the scope of this assessment, including direct costs of damage to private, commercial, and public-built infrastructure and indirect costs such as lost income, lost tax revenues, housing market impacts, and long-term psychological effects (Dale, 2010; Thomas and others, 2017; Barrett, 2018). For example, a recent analysis estimates the total annualized cost of wildfires in the United States at anywhere from $71.1 billion to $347.8 billion (Thomas and others, 2017). State and Federal agencies are responsible for paying the bulk of the suppression costs and, while costs are substantial, they make up only 9 percent of the total wildfire costs. Additional short-term expenses and long-term damages account for 91 percent of total wildfire costs. Of the 91 percent, short-term expenses such as relief aid, evacuation services, and home and property loss are approximately 35 percent, whereas costs related to long-term damages, which can take years to fully manifest, account for approximately 65 percent of wildfire costs.

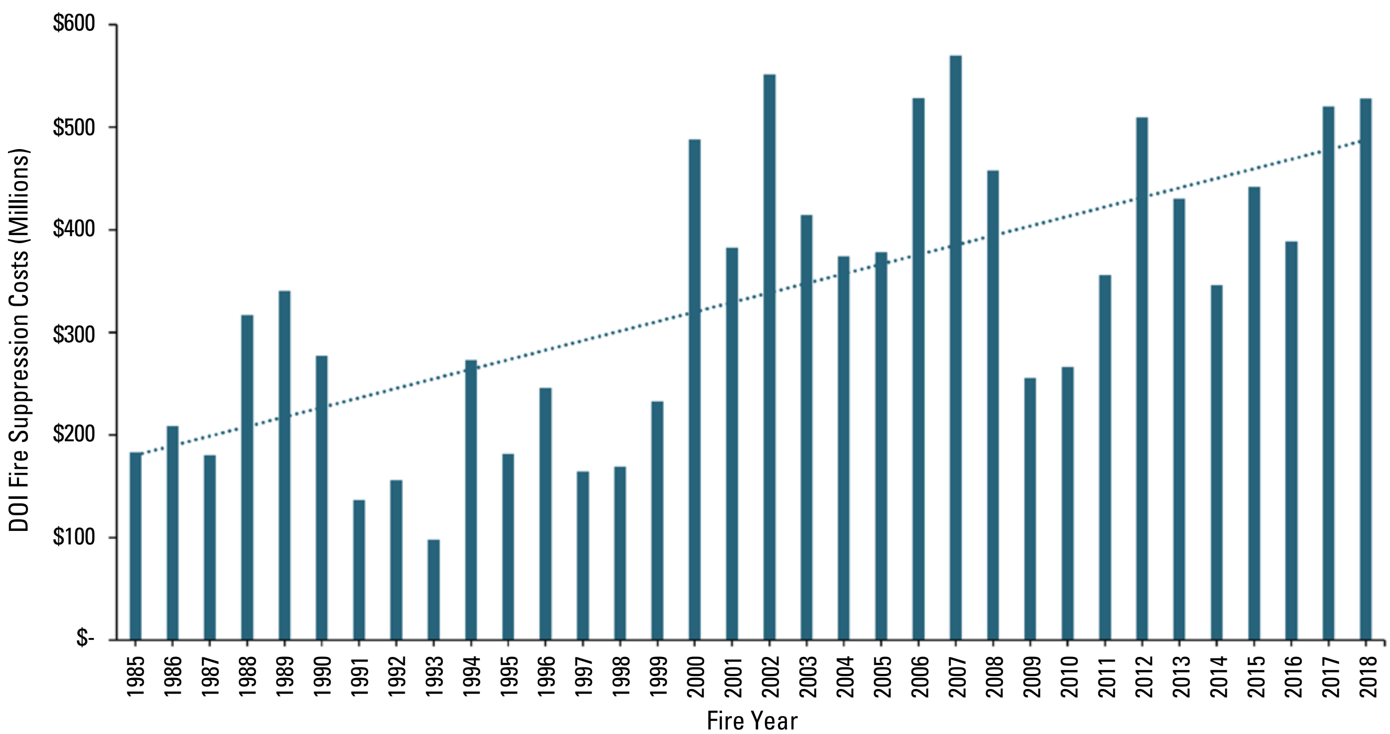

Figure J6. Millions of dollars spent on fire suppression by the U.S. Department of the Interior (DOI) from 1985 to 2018 (National Interagency Fire Center, 2019c). These values have been adjusted for inflation based on the value of the U.S. dollar in 2018 (United States Inflation Calculator, 2019). The dotted blue line represents the linear trendline of dollars spent for the time period.

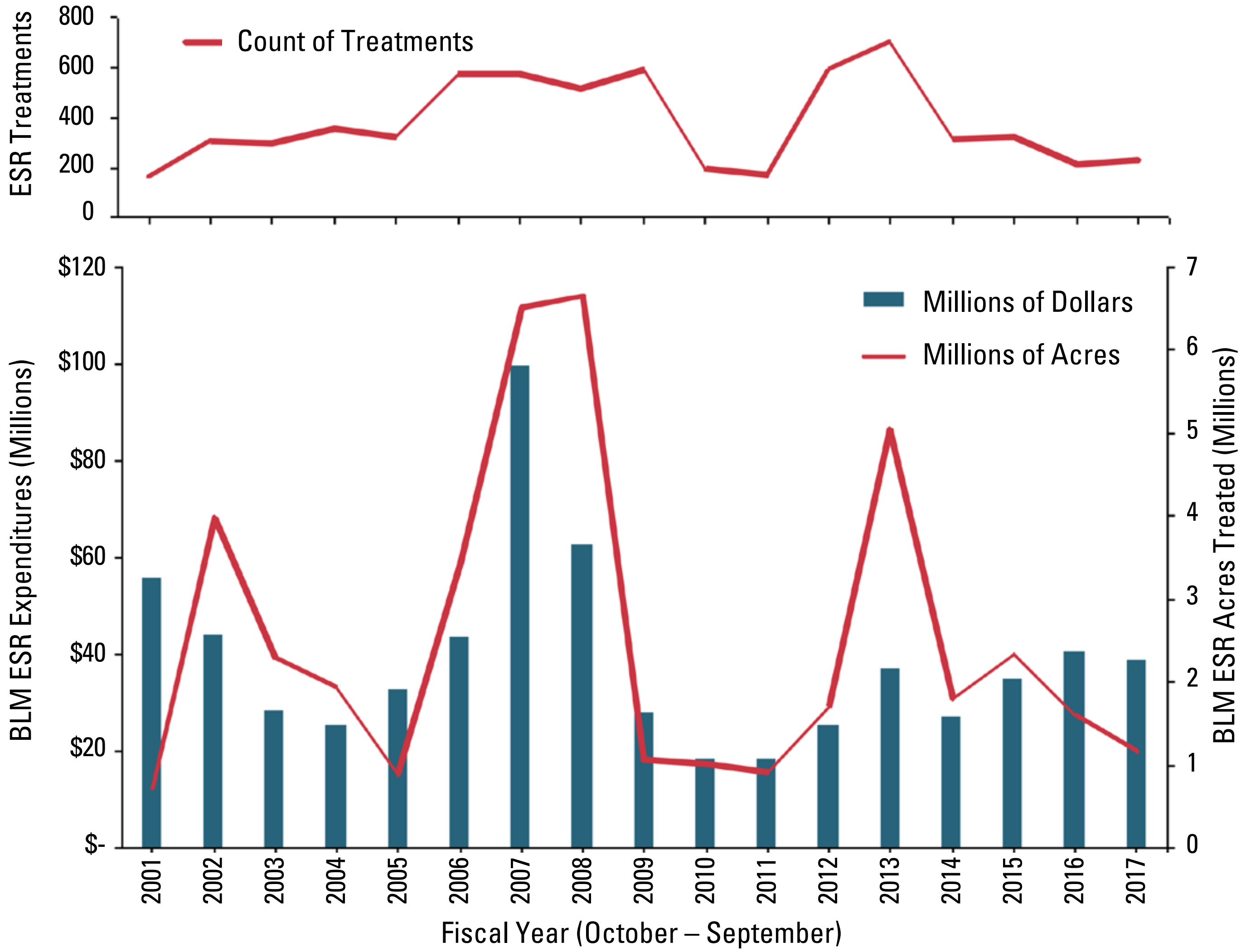

Figure J7. Number of treatments, expenditures, and acres treated by the U.S. Department of the Interior, Bureau of Land Management (BLM) Emergency Fire Stabilization and Rehabilitation (ESR) program from 2001 to 2017 (Bureau of Land Management, 2018a).

Current Coordination Efforts Among Federal, State, and Tribal Entities to Address Fire

The challenge of managing wildland fire in the United States is increasing in complexity and magnitude, and no one agency has the resources to address the growing issues and concerns associated with wildland fire. The DOI and USDA, together with Tribal governments, States, and other jurisdictions, are responsible for the protection and management of natural resources on lands they administer. Wildland fire is a cross-jurisdictional issue, yet uniform Federal and State policies and programs are lacking to ensure consistency in a safe and effective fire response. Since Federal firefighting resources are not enough to cover the full costs of fire response and suppression, cooperative relationships have become increasingly important across all land ownerships. The Federal Wildland Fire Management Policy (U.S. Department of the Interior and U.S. Department of Agriculture, 1995) mandates interagency cooperation and coordination. The policy states that fire management planning, preparedness, prevention, suppression, restoration and rehabilitation, monitoring, research, and education will be conducted on an interagency basis with the involvement of many cooperators and partners.

Several efforts continue to implement the interagency cooperation and coordination element of the Federal Wildland Fire Management Policy. The National Cohesive Wildland Fire Management Strategy (U.S. Department of the Interior and U.S. Department of Agriculture, 2014) outlined new approaches to coordinate and integrate efforts to restore and maintain healthy landscapes, prepare communities for wildland fire, and better address the Nation's wildland fire threats. The Integrated Rangeland Fire Management Strategy (IRFMS; U.S. Department of the Interior, 2015a) was developed specifically to address the issues of sagebrush rangeland fire and nonnative invasive annual grasses through broader wildland fire prevention, suppression, and restoration efforts and to ensure improved coordination with local, State, Tribal, and regional efforts in the threat of sagebrush rangeland fire. The IRFMS (U.S. Department of the Interior, 2015a), is one of the largest efforts to highlight the importance of collaboration, cooperation, and coordination among several Federal and State agencies, Tribes, and local governments, in addition to academia, other nongovernmental organizations, private landowners, and stakeholders. The Western Governors' Association initiatives—including the National Forest and Rangeland Management Initiative, the Species Conservation and Endangered Species Act Initiative, and more recently, the Biosecurity and Invasive Species Initiative—all discuss the issues of invasive annual grasses and the subsequent loss of greater sage-grouse habitat because of wildfire. These initiatives provide opportunities for greater collaboration among land-management agencies, States, Tribes, and other stakeholders on fire-related issues.

In August of 2018, the USDA Forest Service announced a new strategy for managing catastrophic wildfires and the impacts of invasive plant species, drought, and insect and disease epidemics. The associated report, "Toward Shared Stewardship across Landscapes—An Outcome-based Investment Strategy" (U.S. Department of Agriculture, 2018) outlines the USDA Forest Service's plans to coordinate with States to identify landscape-scale priorities for targeted fuel reduction treatments in areas with the highest fire risks. The National Cohesive Wildland Fire Management Strategy provides the foundation for building these relationships. While the USDA Forest Service has worked with States, Tribes, local communities, and collaborative groups to reduce fuels and improve forest conditions around communities, the focus is still largely on wildfire in forested ecosystems.

For all agencies, cooperation and coordination is accomplished through cooperative interagency agreements at all levels, from the national, State, and Tribal levels down to the local-community level. All western State governments have departments that are responsible for wildland fire management and maintain various capabilities and assets for fire suppression. Though cooperative agreements vary by State in terms of their policies and guidelines, State resources often are used to augment and support suppression efforts on Federal lands; similarly, Federal resources augment and support suppression efforts on State and private lands. For example, Colorado and Idaho maintain aviation assets (helicopters and single engine air tankers), and Utah maintains fire crew capability, including two hotshot crews. This coordination creates additional capacity and covers more ground for fire response and suppression actions across jurisdictions.

Prior to each yearly fire season, agencies coordinate and plan by holding preseason meetings, simulation drills, and exercises. During a fire incident, affected agencies may work together to quickly develop cost-share agreements and may participate in the overall wildfire strategy planning effort where management and incident objectives and requirements are identified, assessments completed, and decisions are documented throughout the fire incident. After the fire incident, after-action reviews are frequently performed among all partners with assigned resources to discuss what was planned for the incident versus what the actual response was during the incident, what the challenges were, and what strategies and tactics can be improved for the next incident. These reviews help identify issues or concerns with the goal of strengthening the coordination and communication efforts of all parties involved. Most, if not all, western States are integrated with their partners in fire prevention, public education, fire restrictions, fire information, and reducing the risk of wildfires to communities through the development and implementation of community wildfire protection plans (CWPPs).

Rangeland Fire Protection Associations (RFPAs) are unpaid volunteer groups of rural landowners trained and authorized to respond to wildfires. Initially, RFPAs were formed to address wildfires burning on private unprotected lands, often in remote locations. Western States continue to see the development of more RFPAs, and these types of efforts are needed where State and Federal resources are not able to respond quickly to a fire incident.

The DOI was recently given the authority to strategically transfer surplused firefighting assets (engines, radios, tools, and supplies) to local cooperators that routinely respond to fires on DOI-managed lands (for example, RFPAs and rural fire districts). Leveraging the knowledge, resources, and proximity of the local private landowner has been key for a quick response to wildfires that have become more common in sagebrush communities across the West. While these efforts demonstrate a commitment to address sagebrush wildfires in a coordinated manner, large wildfires will continue to be a management challenge as human populations and cities in the West continue to grow, nonnative invasive annual grasses expand, and climate change continues to increase environmental conditions.

Chapter K. Invasive Plant Species

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Executive Summary

Of the numerous and mounting threats to the health of sagebrush (*Artemisia* spp.) landscapes, none are more widespread or pervasive than invasive plant species. Invasive annual grasses, in particular, pose an immediate threat to the sagebrush biome because they respond quickly to disturbance and lead to more frequent and expansive wildfires which damage habitat as well as infrastructure and threaten human safety. Such invasions ignore land ownership and other jurisdictional boundaries and represent a common threat. Left unchecked, invasive plants degrade plant communities, wildlife habitat, and migratory corridors and threaten wildlife survival. They also can cause significant negative economic impacts. Current estimates indicate most invasive plant-management programs address less than 10 percent of the average annual rate of spread of invasive plants, almost entirely because of a lack of capacity and coordination for common priorities. The cost of managing these infestations increases annually and is commensurate with the exponential annual increase in spread of infestations.

Limited financial resources require that management actions used to ameliorate the impacts of invasive plant species be prioritized, focused, and implemented in a collaborative manner to ensure the greatest conservation and restoration benefits. Strategies of prevention and early detection must be considered across the landscape to prevent adding to an already overwhelming management burden. Only through sustained resources, integrating science and adaptive management within supportive policy, and focused partnerships will successful reduction of the threats posed by invasive plants to the productivity of the sagebrush biome be realized.

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[3]U.S. Geological Survey.

[4]University of Northern Colorado.

[5]University of Wyoming.

[6]Western Association of Fish and Wildlife Agencies.

[7]U.S. Department of the Interior, Bureau of Land Management.

[8]U.S. Department of Agriculture, Forest Service.

[9]Wyoming Department of Agriculture.

Introduction

Executive Order 13112 (signed in 1999 by President Clinton) defines an invasive species as "a species that is nonnative or alien to the ecosystem—the introduction of the species causes or is likely to cause harm to human health, or to the economy or environment." The term "invasive species" refers to plants, animals, insects, fungi, and bacteria, but in this chapter, the discussion is restricted to plants that are invasive to sagebrush (*Artemisia* spp.) ecosystems.

Many nonnative annual and perennial plants are invasive to sagebrush ecosystems (app. K1; Zouhar and others, 2008; Miller and others, 2011; Ielmini and others, 2015). However, invasive annual grasses (such as cheatgrass [*Bromus tectorum*] and medusahead rye [*Taeniatherum caput-medusae*]) are ecosystem disruptors, converting sagebrush habitats to annual grass monocultures, particularly within the Intermountain West and Great Basin ecoregions (fig. K1; Miller and others, 2011). Basin big sagebrush (*A. tridentata tridentata*) and xeric Wyoming big sagebrush (*A. t. wyomingensis*) communities are at the greatest risk for displacement by cheatgrass (Miller and Eddleman, 2001; Connelly and others, 2004; Chambers and others, 2007). There are other invasive exotic annual grasses within the *Bromus* genus (for example, red brome [*B. rubens*] and field brome [*B. arvensis*]) that have impacts similar to cheatgrass, albeit with a more limited extent. As the current cheatgrass-invaded areas become warmer, these species may expand into these areas (Bradley and others, 2016), compounding impacts to sagebrush. Medusahead rye and North Africa grass (*Ventenata dubia*; ventenata hereafter) are emerging annual grass threats within the sagebrush biome. Many perennial forb species invasive to sagebrush plant communities also occur at regional scales. Several native coniferous species are also encroaching into sagebrush landscapes (for example, juniper [*Juniperus* spp.] and pinyon [*Pinus* spp.]). The consequences of conifer encroachment are addressed in chap. M (this volume).

Invasive Plant Descriptions and Regulations

Noxious weeds are generally defined as those that can directly or indirectly cause problems for agriculture, natural resources, wildlife, recreation, navigation, public health, or the environment. Differences in terminology used to describe invasive plants are more than semantic; they impact how agencies treat invasive plants with respect to funding authority, management, and control. These differences also impact the priority the agencies have for management and resources that are available. For example, most States require an invasive plant to be listed on the "noxious weed" list for the use of State funds to be appropriated for weed management. From a regulatory standpoint, only invasive plant species listed on Federal or State noxious weed lists are required to be managed. For example, California, Colorado, Montana, Nevada, Oregon, Utah, and Wyoming list medusahead as a noxious regulated weed (U.S. Department of Agriculture, 2019), but in Washington, it is listed as a Class C noxious weed that doesn't require control, and in Idaho, it is not listed as a noxious weed at all.

Cheatgrass is not listed as a Federal noxious weed and is largely unregulated by the States (Ielmini and others, 2015). Colorado is the only western State that lists cheatgrass as a noxious weed (U.S. Department of Agriculture, 2019). State laws may provide some protection for sagebrush habitats, although large-scale control of invasive plants is not occurring, and rehabilitation and restoration techniques are mostly unproved and experimental (Pyke, 2011).

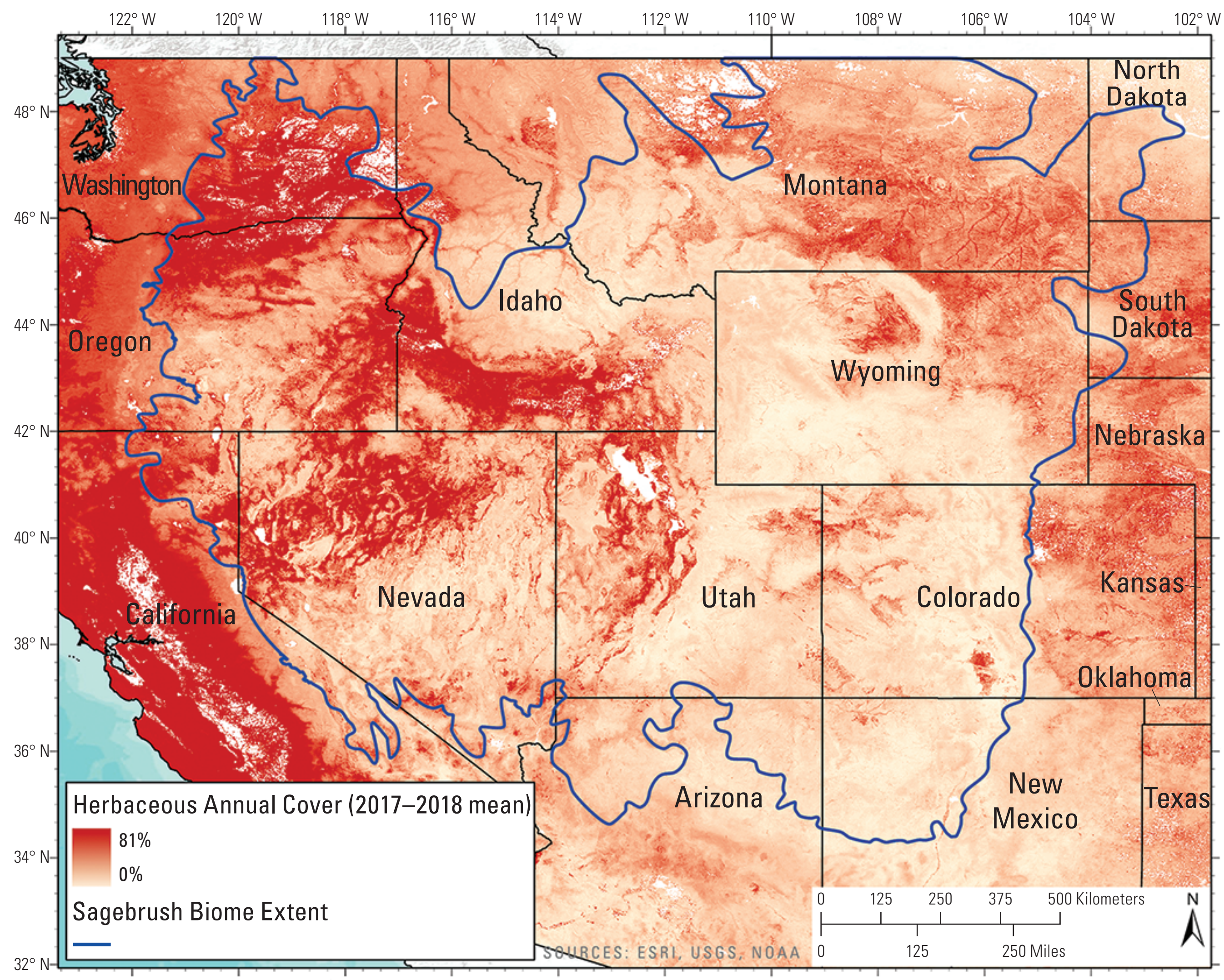

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Figure K1. Mean (or average) herbaceous annual cover in the western United States averaged across 2017 and 2018 (Jones, M.O., 2018). %, percent.

Spatial Extent and Distribution of Invasive Plants

The Western Governors' Association recently compiled a list of the top 25 terrestrial invasive plant species in the West (excluding Oregon, Idaho, and Utah). Eight of the 25 species can grow in sagebrush ecosystems, and cheatgrass was listed no. 2 overall. Cheatgrass is clearly the most widespread species; however, other species, such as no. 8 leafy spurge (*Euphorbia esula*), no. 12 hardheads (Russian knapweed; *Rhaponticum repens*), no. 16 yellow star-thistle (*C. solstitialis*), and no. 22 dyer's woad (*Isatis tinctoria*) may be local problems within sagebrush ecosystems (Western Governors' Association, 2018). While species such as yellow star-thistle and rush skeletonweed are severe problems west of the Rocky Mountains in Oregon and Idaho and are often associated with annual grasses, they have not yet invaded heavily into the eastern part of the sagebrush biome. Other invasive "watch" species and early detection and rapid response species that occur within the sagebrush biome are listed by State in table K1.1.

Medusahead fills a similar niche to other invasive annual grasses in more mesic communities with heavier clay soils (Dahl and Tisdale, 1975). Medusahead can also become abundant on some low sagebrush (*A. arbuscula*) sites below 1,500 meters (4,900 feet) in elevation (Miller and Eddleman, 2001), as well as in some big sagebrush communities (Miller and others, 1999). Medusahead is most common in the western United States from the United States-British Columbia (Canada) border south to California with significant invasions east to Oregon and Idaho. While the species is moving east, it is still restricted as a new invader and has potential for eradication in eastern parts of the sagebrush biome in Montana and Wyoming with a consistent monitoring effort and timely responses to new detections. Little sagebrush communities are most susceptible to medusahead invasion, whereas big sagebrush communities are more resistant (Young and Evans, 1970, 1971).

Ventenata is the most recent exotic annual grass to invade the sagebrush biome. As a relatively new invader, less information is available regarding its ecological niche, but it may fill a similar niche as medusahead (Jones, L.C., and others, 2018), invading areas of clay soils with poor drainage. The species may initially establish on moist sites but spread to drier sites (Fryer, 2017) and is well established in eastern Washington, western Idaho, and Oregon (Kerns and others, 2016). Thought to be less common in other western States, ventenata is being found more commonly than expected in northern Utah, western Montana, and northeastern Wyoming (Martin, 2005). Medusahead, cheatgrass, and ventenata coexist in distribution and requirements and the species interact with one another in response to land management (Young and Evans, 1970; U.S. Department of Agriculture, 2019). In addition, treatment for one species may result in weed succession for the other invasive annual grasses depending on site conditions and environmental factors. For example, ventenata is rapidly spreading and becoming dominant in sagebrush communities where cheatgrass or medusahead was formerly dominant (Bansal and others, 2014).

More than 50 species of exotic tap-rooted forbs that originate from Eurasia exist in the sagebrush biome. Exotic forbs, particularly perennials, are a taxonomically diverse group of invaders that are well represented on State noxious-weed lists and thus have received more concentrated eradication efforts in addition to regulation of spread (for example, with "weed-free" certification) compared to invasive annual grasses. Notable species include perennials such as knapweed (*Centaurea* spp.), thistle (*Cirsium* spp.), rush skeletonweed (*Chondrilla juncea*), whitetop/hoary cress (*Lepidium draba*), and other annuals/biennials that may not always be listed as noxious, such as Family Brassicaceae (mustards) and prickly lettuce (*Lactuca serriola*).

Critical to understanding the problem of invasive plants is understanding where they exist across the landscape. Species-specific distribution information can be found in several resources (table K1; see also Dingman and others, 2018, table 2). However, updated, comprehensive landscape-scale distribution information often does not exist, and much of the extant distribution information is incomplete (Ielmini and others, 2015).

Concerted efforts are being made to track specific invasive plant species across the extent of areas they have successfully invaded. For example, research projects underway through the Northwest Fire Science Consortium (an exchange network of the Joint Fire Science Program) are focusing on ventenata and its distribution and spread in the Blue Mountain ecoregion of Idaho, Oregon, and Washington. Cheatgrass infestations are monitored through the National Land Cover Database (U.S. Geological Survey, 2018d). For the other invasive plant species, monitoring is largely done via county or State reports to databases such as the PLANTS database (U.S. Department of Agriculture, 2019) or EDDMapS (University of Georgia, 2019). Agency-level datasets (for example, U.S. Department of Agriculture, Forest Service [USDA Forest Service] Forest Activity Tracking System and U.S. Department of the Interior, Bureau of Land Management [BLM] National Invasive Species Monitoring System) also spatially document invasive plant occurrences on public lands. The U.S. Geological Survey (USGS), in collaboration with Colorado State University, is developing the INHABIT tool that will provide modeling projections of potential suitable habitat and future occurrence for several invasive plant species (available at https://engelstad.shinyapps.io/dashboard_dev/).

A process has been developed to estimate the extent of cover of herbaceous annuals in the sagebrush biome by May of each year (fig. K2) that could be used for project planning and positioning resources in response to increased wildfire risk (Boyte and Wylie, 2017, 2018, 2019). The geographic coverage includes the Great Basin, Snake River Plain, State of Wyoming, and contiguous areas. The Rangeland Analysis

Platform (Jones, M.O., and others, 2018) provides another dataset and is an online mapping tool that can be used to evaluate or compare trends in the herbaceous annual functional group, which includes invasive and native annual grasses.

These broad-scale efforts are useful for rangewide national policy evaluations of management and budget needs, long-range temporal trends of high risk for specific regions, and for highlighting broad areas of potential low risk or low invasion. Many partners are also developing spatially explicit maps of existing levels of invasion and maps of suitable habitat at risk. These maps can help identify how these invaders move across the landscape and how they may impact resources to inform local-scale conservation delivery.

Feedback and Climate Effects

In addition to understanding where invasive species occur now, understanding where they are likely to be in the future is important when developing long-term management plans. The distribution of many invasive plants will likely shift with climate change. Bradley and others (2009) predicted that the range of spotted knapweed (*C. stoebe*) will expand in some areas, mainly in parts of Colorado, Idaho, Oregon, and western Wyoming, and will contract in other areas (for example, eastern Montana). They also predict the range of yellow star-thistle will expand eastward and the invasion risk of leafy spurge will likely decrease in several States, including parts of Colorado and Idaho.

Table K1. Summary of some major sources of information on invasive plant species, distribution, and management (adapted from Dingman and others, 2018, table 2).

Source	Developer	Contains
Fire Effects Information System	U.S. Department of Agriculture, Forest Service	Species autecology reviews
Canadian Journal of Plant Science	Agricultural Institute of Canada	Biology of Canadian weeds; biology of invasive alien species
PLANTS Database	U.S. Department of Agriculture, Natural Resources Conservation Service	Species taxonomy, links to further information, Federal noxious weed list
Early Detection and Distribution Mapping System (EDDMapS)	The University of Georgia—Center for Invasive Species and Ecosystem Health	Species information, national and State-level distributions, library of identification and management information
NatureServe Explorer	NatureServe	Use data explorer to find species ecology
GoogleScholar	Google	Search tool; subscriptions needed for some located articles
JSTOR (Journal Storage)	ITHAKA	Digital library of journals, primary sources, and books
California Invasive Plant Council	Cal-IPC	Species summaries, management information
State and Regional Invasive Plant Councils and Exotic Pest Plant Councils	Nongovernment agencies	Species summaries, management information, training opportunities (Accessible National Association of Exotic Pest Plant Councils)
State noxious weed lists	State agencies	Species information, often State-level distributions and status for species of concern to agriculture
State natural heritage programs	State agencies	Species information, often State-level distributions and status
Inventory and Analysis Program	U.S. Department of Agriculture, Forest Service	Vegetation plot data for forested landscapes, requires special access
Integrated Resource Management Applications (IRMA)	National Park Service	Digital data store for national parks
U.S. Department of the Interior, National Park Service (NPS) vegetation inventory	National Park Service	Vegetation plot data, including invasives, within parks
NPS treatment synthesis	Abella (2014)	Synthesis of publications on treatments on NPS lands
NPS National Invasive Species Management System	National Park Service	Web-accessible geospatial tool that is the NPS standard

If sagebrush ecosystems—particularly Wyoming big sagebrush communities—lack resilience to invasive annual grasses, conversion to a novel annual grassland steady-state is likely (Miller and others, 2011; Pyke and others, 2014). Once the transition to an annual-dominated alternative state has occurred, restoring native sagebrush communities is difficult with a low probability of success (Knutson and others, 2014; Shriver and others, 2018). Determining if and how invasive plants impact plant and wildlife communities is important for deciding if control treatments should be applied to a site, determining how treatments should be designed, and estimating the likelihood of treatment success. Invasive plants can greatly alter ecosystem processes to their own benefit, which negatively affects the likelihood of treatment success. In contrast, there are also many cases where invasive plants appear to persist at levels that do not cause appreciable changes to the plant community.

Concern with habitat loss and fragmentation owing to fire and invasive plants has mostly been focused on the western part of the sagebrush biome. However, climate change may alter the range of invasive plants (chap. L, this volume), potentially expanding this threat. The establishment of invasive annual grasses will then contribute to increased fire frequency in those areas, further compounding habitat loss and fragmentation. The fire-invasive feedback loop may be promoted by warmer and wetter winters (Bradley, 2009) and may result in a subsequent increase in establishment and growth of invasive winter annuals. These cycles may be exacerbated by rising atmospheric carbon dioxide concentrations, more nitrogen deposition, and increases in human activities that result in soil-surface disturbance and invasion corridors (Chambers and others, 2014a). As an example, cheatgrass already competes successfully against native perennial grasses because of its early maturation, short root systems to

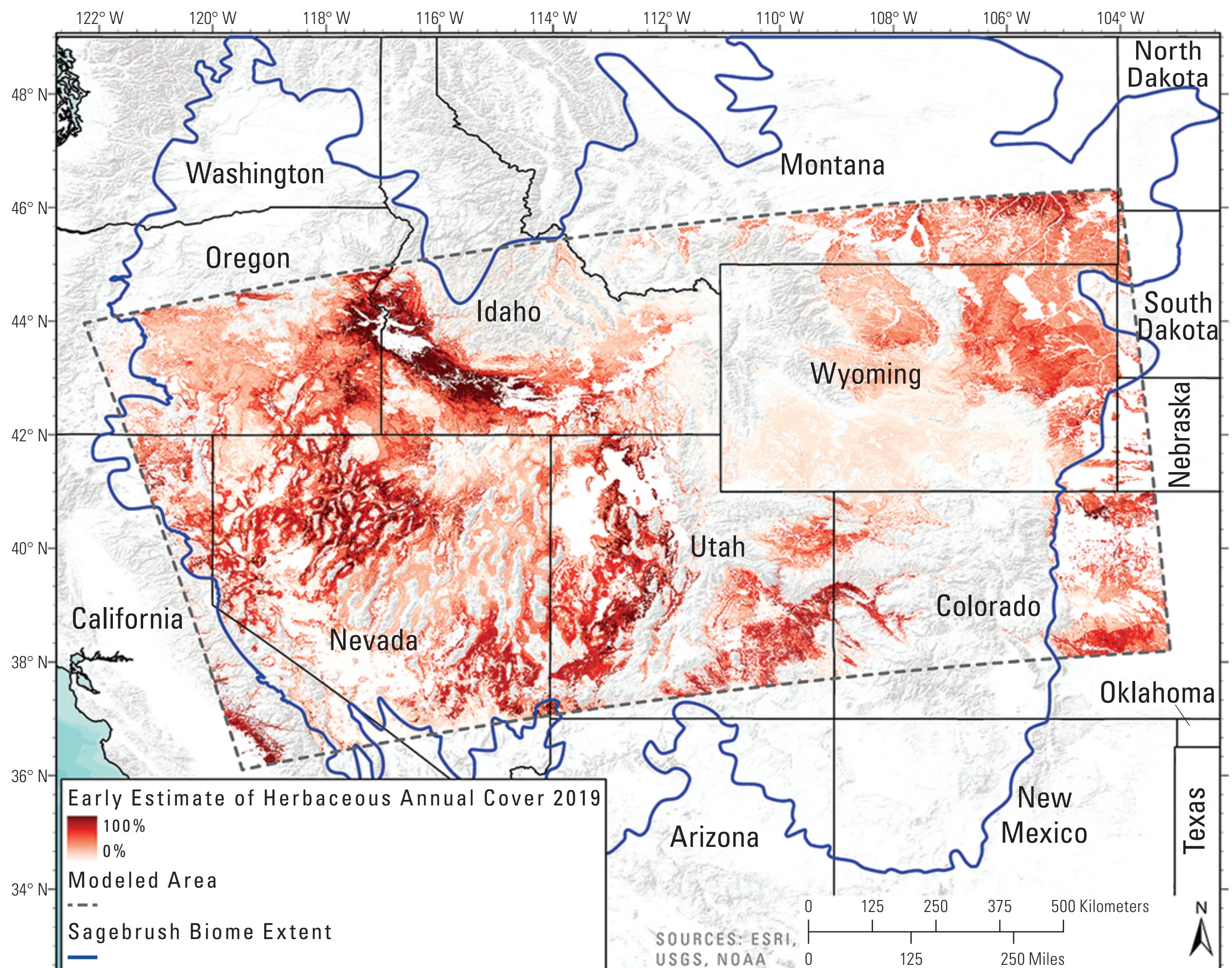

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Figure K2. Early estimates of herbaceous annual cover in the sagebrush (*Artemisia* spp) ecosystem in the Great Basin, Snake River Plain, State of Wyoming, and contiguous areas for May 2019. Data obtained from Boyte and Wylie (2019). %, percent.

collect water in soils, greater seed production, and ability to respond quickly to resources released during disturbance.

Changes in climatic suitability may create restoration opportunities in areas that are currently dominated by invasive plants (Bradley and others, 2009). Cheatgrass will eventually disappear from areas that become climatically unsuitable for this species, but this transition is unlikely to occur suddenly. Areas that become unfavorable to cheatgrass may become favorable to other invasive plants such as red brome, which is more tolerant of higher temperatures (Bradley and others, 2016). Invasions into native plant communities may also be sequential, as the initial invaders are replaced by a series of new invasive plants or by species adapting to new habitats within their range (Young and Longland, 1996). For example, areas along the Snake River Plain and the Boise Front Range in Idaho, once dominated by cheatgrass, have been invaded by medusahead. Rush skeletonweed, which is typically localized to disturbed areas in xeric sagebrush-grassland communities, is now invading areas dominated by medusahead (Sheley and others, 1999) and in postfire habitats (Kinter and others, 2007). Therefore, one cannot assume that areas that become unsuitable for cheatgrass will return to pre-invaded habitat conditions without significant restoration efforts. Modeling and experimental work are needed to assess whether native species could still occupy these sites if invasive plants are reduced or eliminated by climate change (Bradley and others, 2009).

Impacts of Invasive Plants on Sagebrush Plant Communities

Invasive plants impact sagebrush ecosystems by altering plant community structure and composition, and invasive plants may cause declines in native plant populations through competitive exclusion and niche displacement, among other mechanisms (Mooney and Cleland, 2001; Germino and others, 2016). Invasions are impactful in arid and semiarid ecosystems of the western United States when they transform perennial shrub-steppe communities into invasive annual grasslands or transform perennial grasslands into meadows dominated by invasive forbs. When compared to native species, the differing biological or ecological properties of invasive plants typically result in undesirable changes in disturbance regimes (such as fire), ecosystem services (such as forage or pollinators), ecosystem functioning (such as carbon and water cycles), productivity, nutrient cycling, and ecosystem diversity that causes the loss of ecosystem stability in time. Ecosystem changes can greatly benefit the invaders and create feedback that increases their chances of success and potential dominance.

The synergistic invasive annual grass/fire cycle is likely the most impactful feedback that occurs in the sagebrush ecosystem (Brooks and others, 2004; Germino and others, 2016). Although the historical frequency of fire continues to be debated (Baker, 2011; Miller and others, 2011), most agree the role of fire in the sagebrush ecosystem has changed substantially since post-European settlement (Pyke and others, 2016). The significant expansion of cheatgrass following fire has caused that species to garner the most attention of all invading plants in the sagebrush biome. Other attributes that focus attention on cheatgrass are that it can readily attain high levels of community dominance, effectively displace native species such as sagebrush and key forbs, and destabilize ecosystem productivity and other functions following fire. Most problematic, however, is that cheatgrass increases the occurrence of wildfire well beyond levels tolerable by native plant species (D’Antonio and Vitousek, 1992; Germino and others, 2016).

After fire, dominant invasive plants outcompete most native plants, especially natives establishing from seed (Miller and others, 2011), and thereby degrade and fragment existing sagebrush habitat. When cheatgrass becomes dominant in places where weather variability is high, cheatgrass productivity responds strongly to annual and seasonal weather variation.

Fire typically promotes dominance of invasive plants within plant communities, establishing an invasive plant-fire regime cycle (Brooks and others, 2004). Invasive perennial forbs generally persist or may increase following fire owing to life-history traits such as prolific seed production, persistent seed banks, and rooting characteristics, including the ability to sprout from rhizomes, root crowns, or adventitious buds (Ielmini and others, 2015). Annual invasive forbs with transient seed banks may be vulnerable to, and controlled by, fire during certain life-history stages, but many invasive annual and biennial forbs (for example, mustards) are among the first to establish after any disturbance, including fire (Piemeisel, 1951). Deep-rooted, creeping invasive perennials such as hardheads (Russian knapweed), squarrose knapweed (*C. virgate*), dalmation toadflax (*Linaria dalmatica*), and Canada thistle (*Cirsium arvense*) can increase following fire but do not impact sagebrush ecosystems on a landscape scale.

These invasive perennials can pose a significant threat to sagebrush habitats on a local scale (Ielmini and others, 2015). Invasive forbs, particularly perennials, are often secondary and more persistent invaders of sites already invaded by invasive annual grasses or even of sites that have been restored to a native bunchgrass community. Most of these invasive forbs have deeper tap roots than grasses and can remain active long after grasses have senesced. These invaders deplete soil water from beneath the root zone of grasses (Hill and others, 2006), which can be to the detriment of native perennials such as the deep-rooted sagebrush or native forbs. The impacts of invasive forbs vary considerably from simple species displacement by species such as prickly lettuce (Prevéy and others, 2010a), to the thorny nuisance of thistles and corresponding loss of utilization by livestock, and to the high toxicity of leafy spurge when eaten by native ungulates. Invasive forbs often have showy flowers and can strongly affect pollinator communities, sometimes attracting them (to the benefit of other native perennials), but possibly also competing with native plants for scarce pollinators (Brown and others, 2002).

Impacts of Invasive Plants on Wildlife Communities

Changes in vegetation composition and structure associated with invasive plants reduce and, in cases where monocultures occur, eliminate vegetation that sagebrush-dependent species use for food, cover, or as substrates for nesting or perching, which fragments and degrades habitat (Miller and others, 2011). Invasive plants impact the entire sagebrush biome to varying degrees. (Miller and others, 2011).

Changes in vegetation composition and structure associated with invasive annual grasses may indirectly affect local greater sage-grouse (*Centrocercus urophasianus;* hereafter sage grouse) populations by outcompeting native perennial plants after wildfires, reducing this important part of sage-grouse habitat. Prelaying and nesting females selectively feed on herbaceous forage (for example, Barnett and Crawford, 1994), and broods initially feed almost entirely on a variety of native forbs (for example, Klebenow and Gray, 1968; Drut and others, 1994) and associated insects (for example, Klebenow and Gray, 1968; Johnson and Boyce, 1991; Drut and others, 1994; Gregg and Crawford, 2009).

Sage-grouse in northwestern Nevada selected large expanses of sagebrush-dominated areas for nesting (Lockyer and others, 2015) and, within those areas, sage-grouse selected microsites with higher shrub canopy cover and lower cheatgrass cover (Lockyer and others, 2015). Nest-site selection was also negatively correlated with cheatgrass abundance in south-central Wyoming (Kirol and others, 2012), indicating that changes in species composition and vegetative structure associated with cheatgrass degraded sage-grouse habitat. Cheatgrass was not widespread, but when present, it was associated with anthropogenic features, suggesting female sage-grouse may not have selected against cheatgrass but instead may have avoided nesting areas dominated by cheatgrass because of human development and infrastructure (Kirol and others, 2012).

Sage-grouse population demographic studies in northern Nevada found that recruitment and annual survival were reduced by the presence of invasive annual grasses at larger spatial scales. Recruitment of male sage-grouse to leks was inversely correlated with the extent of invasive grasses within a 5-kilometer (km; 3.1 mile [mi]) radius of the lek (Blomberg and others, 2012). Recruitment to leks was strongly correlated with precipitation, but leks impacted by invasive grasses did not experience increases in recruitment in high precipitation years and in the highest precipitation year had recruitment rates one-sixth of that of nonimpacted leks (Blomberg and others, 2012). Survival of adult males was also inversely correlated to the amount of invasive grasses within a 5-km radius of the lek they attended (Blomberg and others, 2012).

At the landscape scale, studies are beginning to quantify the effects of invasive annual grasses on sage-grouse distribution and abundance. A strong negative association between sage-grouse occupying an area if cheatgrass was present was reported, even with less than 5 percent cheatgrass cover (Arkle and others, 2014). In a rangewide analysis, increasing cover of invasive plants was associated with declining trends in counts of males on leks. Few leks had greater than 8 percent invasive annual vegetation cover, suggesting that when the extent of the landscape dominated by invasive plants becomes relatively high, leks become inactive (Johnson and others, 2011).

Invasive grasses, particularly cheatgrass, diminish sagebrush sparrow (*Artemisiospiza nevadensis*) habitat and use (McAdoo and others, 1989; Brandt and Rickard, 1994), and the replacement of sagebrush with cheatgrass also reduces sage thrasher (*Oreoscoptes montanus*) densities (Reynolds and Trost, 1980; Krementz and Sauer, 1982; McAdoo and others, 1989; Brandt and Rickard, 1994). The replacement of sagebrush with grasses—whether through invasion (cheatgrass) or seeding-introduced Eurasian species such as wheatgrass (*Agropyron cristatum*) for reclamation and restoration—also reduces sage thrasher densities (Reynolds and Trost, 1980; Krementz and Sauer, 1982; McAdoo and others, 1989; Brandt and Rickard, 1994).

Although cheatgrass can be nutritionally beneficial to mule deer (*Odocoileus hemionus*) during the spring when rapid new growth occurs (Bishop and others, 2001), the long-term costs of cheatgrass invasion outweigh the seasonal nutritional benefits. Because cheatgrass invasion can cause major changes to vegetation cover (especially in Wyoming big sagebrush communities), forage availability on winter ranges for mule deer has been reduced. This phenomenon at least partially explains a twofold decline in mule deer abundance in northwest Nevada (Clements and Young, 1997).

Habitat loss and fragmentation resulting from wildfire, invasion of introduced annual grasses and invasive forbs (especially cheatgrass and Russian thistle), and conifer encroachment have negatively impacted the dark kangaroo mouse (*Microdipodops megacephalus*) through loss of population connectivity (Hafner and Upham, 2011). Additionally, the density and diversity of small mammals occupying the Snake River Plain, Idaho, were lower where cheatgrass was present (Hanser and Huntly, 2006).

Invasive plants are widespread, have the ability to spread rapidly, occur near areas susceptible to invasion, increase in areas of human development, and are difficult to control. For these reasons, invasive plants will likely continue to replace and reduce the quality of sagebrush wildlife habitat across the biome without focused strategies of collaborative conservation.

Impacts of Invasive Plants on Human Needs and Values

People who use the sagebrush ecosystem and the services it provides—whether the services be economic, cultural, recreational, or environmental—are impacted by invasive plants. Like the effects to native plants and wildlife outlined above, the impacts to human needs and values are mostly negative. These include those stemming from increases in fire frequency (Balch and others, 2013) that reduce livestock forage in the short term for both native and domestic ungulates (Launchbaugh and others, 2008; Brunson and Tanaka, 2011), cause the possible extirpation of native vegetation (DiTomaso and others, 2017), reduce air quality, destroy human-built structures, and cost governments and people millions of dollars annually, both for fighting fires and recovering from fires (Brunson and Tanaka, 2011). Impacts to humans from invasive plant species are likely to increase as invasive annuals and perennials expand their distribution, a process likely to be accelerated and exacerbated by climate change.

Cultural Impacts of Invasive Plants

One impact of sagebrush degradation for humans is the potential adverse effect on cultural practices that help to maintain cultural identities and contribute to human well-being (Pfeiffer and Voeks, 2008). The degradation of the sagebrush ecosystem by exotic plant invasions might adversely affect culturally relevant activities practiced by Native Americans, including sacred wildland gathering sites where the collection of medicinal and ceremonial plants occurs. If the vegetation composition is completely altered and sagebrush extirpation occurs, cultural practices that use sagebrush or other plants native to sagebrush ecosystems will be more difficult to undertake. These cultural practices could even cease.

Economic Impacts of Invasive Plant Species in the Sagebrush Ecosystem

Studies estimating the economic impacts of invasive plant species in rangeland and wildland systems of the arid and semiarid western United States are scarce (Duncan and others, 2004; Brunson and Tanaka, 2011). Researchers have instead focused attention on quantifying negative effects on livestock forage and outdoor recreation. Livestock forage availability is typically reduced when nonnative grasses, forbs, or other weeds invade an area, although the precise effects vary across time, space, and invasive plant species depending on the grazing animals under consideration. This point has been discussed specifically for invasive annual grasses such as cheatgrass (Pellant, 1996). However, the impacts of cheatgrass on forage availability, ranch management practices, and ranch profitability are variable and complex, owing to differences across ranches and seasons (Cook and Harris, 1952; Murray, 1971; Brunson and Tanaka, 2011).

Attitudes vary about the problems posed by cheatgrass invasion. For example, the invasion of cheatgrass into sagebrush ecosystems may afford some benefits to ranchers if they incorporate cheatgrass into spring livestock grazing programs. Cattle graze on cheatgrass during early spring when forage options are limited (Brunson and Tanaka, 2011), and it is palatable, but the period for grazing cheatgrass is relatively short (in the absence of protein supplementation), and the loss of forage after cheatgrass-driven fire is problematic. A study of attitudes in Colorado and Wyoming found that 51 percent and 66 percent of surveyed ranchers and natural resource professionals, respectively, perceived cheatgrass to be a moderate to severe problem, with variations in attitudes traced to differences across locations in elevation, climate, infestation extent, and other factors (Kelley and others, 2013). A study employing expert judgment surveys indicated that land-management specialists in the West believe cheatgrass does offer limited forage value, but the average respondent estimated that a cheatgrass monoculture reduces the yearlong ability of rangeland to support livestock by about 70 percent as compared to perennial grasses (Auton and others, 2000). Other exotic annual species, such as medusahead and especially ventenata, are more recently expanding in the wake of cheatgrass, but these species are even less palatable to livestock.

When an invasive plant species negatively affects forage availability for livestock, there are two broad types of economic effects on agricultural producers: (1) losses from reduced production and (2) out-of-pocket costs incurred to compensate for such losses, such as expenditures made to control invasive plant species, grow or purchase livestock feed elsewhere, or adopt more costly grazing management schemes. To account for the economic effects of invasive plant species, it is important to enumerate and consider both losses and costs. Estimating economic losses owing to reduced livestock forage in the sagebrush biome is complicated by incomplete information regarding the true values of several important variables, as well as how they vary spatially across this broad geographic realm. These variables include the number of acres infested by, and percent cover of, invasive annual grasses, expansion rates of infestations, percent reductions in livestock carrying capacity (for example, animal unit months [AUM]/acre), and baseline livestock stocking rates.

Despite uncertainty and incomplete information, researchers have attempted to estimate livestock grazing losses for smaller jurisdictional units. These bounding exercises provide a better indication of a likely range of potential losses rather than a point estimate that may be misleading. Estimates are usually conservative partly because they do not include the “secondary economic impacts” that would occur as a result of the primary, or direct, economic losses from reduced livestock grazing. For example, when farmers and ranchers carry out agricultural activities such as crop production and livestock grazing, they also purchase items such as fuel, seed, supplies, food, and a range of other goods and services necessary to support the agricultural enterprise and the farm or ranch family. Therefore, when agricultural production in a geographic area declines for any reason (including damages induced by invasive species), the result is not only (1) direct economic losses from reduced production, but also (2) a sequence of secondary losses because of reduced spending by the agricultural producer (and other producers and consumers in the rural economy) for a variety of goods and services. In the event of fire, the most acute, direct economic loss for ranchers is the loss of forage. On Federal leases, the loss of forage is likely to extend for 2 or more years while grazing is curtailed so vegetation can recover. Fire-based loss of forage is accentuated with increasing annual grass abundance in the negative feedback loop of cheatgrass and fire.

In addition to impacts from annual grasses themselves, it is important to consider that annual grass-dominated communities are open plant systems easily invaded by "the next weed that is introduced" (Young and Longland, 1996). Several perennial weeds have substantial negative economic impacts on grazing as well. For example, one study estimated that total losses of livestock forage value owing to yellow star-thistle on private land in California were approximately $10 million per year, with ranchers' out-of-pocket expenditures on yellow star-thistle control amounting to approximately $13 million per year (estimates from that study, which was conducted in 2003, have been updated here to 2018 U.S. dollars using the Consumer Price Index; Eagle and others, 2007). Together, the loss of forage and cost of control amounted to the equivalent of 6–7 percent of the total annual harvested pasture value for the State of California (Eagle and others, 2007).

Invasive annual grasses and other weeds in the sagebrush ecosystem can have substantial impacts on outdoor recreation activities such as fishing, hunting, hiking, wildlife viewing, and water-based recreation. Invasive annuals negatively affect a wide array of environmental attributes that support outdoor recreation, including but not limited to soil quality, water quality and quantity, plant diversity, forage and cover availability, and animal diversity and abundance. In addition, as with grazing, the perennial weeds that may invade annual grass-dominated landscapes have similar deleterious effects on ecosystem characteristics and functions that support recreation.

Several different types of information are required to estimate the impacts of invasive grasses and weeds on outdoor recreation. However, owing to the logistical challenges and expense involved with data collection, researchers generally do not have access to complete, high-quality data such as (1) invasive plant distribution data, (2) invasive weed percentage cover data, and (3) site-specific information on how recreators respond (for example, by visiting a site less often or enjoying their recreation less) when an infestation invades an area. Yet, it is important to develop and consider best estimates of the range of recreational impacts.

Some studies in the past have tended to take a single-species approach. For example, Leitch and others (1996) estimated losses in wildlife-related recreation expenditures owing to leafy spurge in a set of four States, ranging from $33,000 per year for Wyoming to $3.6 million per year for North Dakota (monetary estimates from that study have been updated here to 2018 U.S. dollars using the Consumer Price Index). Another study examining invasive plants and grasses in aggregate, rather than a single species, estimated the negative economic impacts (including secondary impacts from reduced indirect and induced spending) on wildlife-related recreation in Nevada likely ranged from $9 million to $18 million per year (updated to 2018 U.S. dollars using Consumer Price Index; Eiswerth and others, 2005). Using the most conservative findings for annual recreation losses, the predicted negative economic impacts over a future time horizon of 5 years ranged from about $44 million to $60 million in Nevada (updated to 2018 U.S. dollars using Consumer Price Index; Eiswerth and others, 2005). Such estimates are informative given that, at the time of that study, Nevada ranked 47th out of all States in terms of the total numbers of recreational days devoted to fishing, hunting, and wildlife watching. One would expect that, if the extent of invasive species and other factors are equal, the monetary values of negative impacts would be larger in otherwise similar jurisdictions having higher baseline levels of recreation.

Much of the uncertainty in economic analyses for any invasive plant species is because of knowledge gaps regarding infestation sizes. Therefore, ranges of current annual impact estimates are perhaps most useful as inputs for illustrating potential impacts over future time horizons. However, framing estimates about future losses is also challenging owing to the uncertainty regarding rates of future infestation expansion (Eiswerth and others, 2005).

Support for Invasive Plant Threat Reduction

Policy

Regulatory and operational responsibilities for invasive plant management are driven by an "all hands, all lands" approach across Federal, State, private, nongovernmental, Tribal, and corporate lands since invasions cross fence lines and jurisdictional boundaries. A variety of regulatory mechanisms and nonregulatory measures to control invasive plants exist. However, no single Federal law or combination of policies provides clear authority or coordination among Federal agencies to address invasive plant species (Corn and Johnson, 2013), and the extent to which these mechanisms effectively ameliorate the current rate of invasive expansion is unclear. Policy directives, guidance, and regulations enable State and Federal agencies and the remaining wildlife conservation community to work together to identify priorities, allocate resources, and take action, which includes (1) preemptively responding to invasions, (2) reducing wildfire risk with fuels management treatments, (3) limiting plant invasions postfire, and (4) performing restoration activities.

Legislative authorities (see “Primary Legislative Authorities Addressing Management of Invasive Plant Species” sidebar, below) such as the Federal Noxious Weed Act of 1974 (7 U.S.C. 2801 et seq.), and the Plant Protection Act of 2000 (7 U.S.C. 7701 et seq.) as amended by the Noxious Weed Control and Eradication Act of 2004 (118 Stat. 2320), provide the basis for a collaborative approach by authorizing the Secretary of the Department of Agriculture to cooperate with other Federal and State agencies, among others, in carrying out operations or measures to eradicate, suppress, control, prevent, or slow the spread of any noxious weed. State and Federal legislation has been developed to regulate high-risk invasive plants to protect the environment, private landowners, and industries. There are a wide range of programmatic activities that have been derived from this legislation, for example, programs to prevent and control plants regulated under these laws, or programs to develop noxious weed-free products for trade or sale.

Coordination of Federal invasive plant efforts is facilitated by the Federal Interagency Committee for Management of Noxious and Exotic Weeds, which was created in 1994 and consists of representatives from 16 Federal agencies with invasive plant management and regulatory responsibilities. In addition to enabling legislation, Federal invasive species management has been directed by a variety of Executive Orders (EO), such as EO 13112 signed in 1999 and, amended by EO 13751 in 2016, directs continued coordinated Federal prevention and control efforts and incorporation of human and environmental health, climate change, technological innovation, and other emerging priorities into Federal efforts to address invasive species. The National Invasive Species Council was created by EO 13112 to provide national level coordination among Secretarial departments and works with an Invasive Species Advisory Council to promote documentation and sharing of invasive species information. EO 13855 (2019), includes an invasive species management directive to promote active rangeland management to reduce wildfire risk.

Each Federal agency focuses on invasive species management on their respective fee title lands, supports efforts by others, and works with partners to address spread, early detection and rapid response, education, outreach, and research. The Natural Resources Conservation Service and U.S. Fish and Wildlife Service Partners for Fish and Wildlife Program provide technical assistance and funding to private landowners for a variety of conservation benefits, including control of noxious weeds. Federal invasive species management actions must comply with requirements under other authorities such as the National Environmental Policy

Primary Legislative Authorities Addressing Management of Invasive Plant Species (from Corn and Johnson, 2013)

Organic Administration Act of 1897 (16 U.S.C. 473–475, 477–482, 551).—Provides broad authority to the USDA Forest Service to protect National Forest System lands from a range of threats, including invasive species.

Federal Noxious Weed Act of 1974 (7 U.S.C. ch. 61 2801 et seq.).—States that each Federal agency shall (1) designate an office or person to develop and coordinate an undesirable plants management program for control of undesirable plants on Federal lands, (2) establish and adequately fund an undesirable plants management program, (3) complete and implement cooperative agreements with State agencies regarding the management of undesirable plant species on Federal lands, and (4) establish integrated management systems to control or contain undesirable plant species targeted under cooperative agreements.

Federal Land Policy and Management Act of 1976 (43 U.S.C. ch. 35 1701 et seq.), as amended.—Provides funds for range betterment, including weed control on certain National Forest System rangelands.

Cooperative Forestry Assistance Act of 1978 (16 U.S.C. ch. 41)*, as amended.*—USDA Forest Service may enter into cooperative agreements to assist other Federal, State, and private entities in controlling and managing invasive species on other Federal lands and non-Federal lands.

Public Rangelands Improvement Act of 1978 (43 U.S.C. ch. 37 1901 et seq.).—Provides funding for on-the-ground rangeland rehabilitation and range improvements on some of the rangelands managed by the USDA Forest Service.

Noxious Weed Control Act of 2004 (7 U.S.C. 7781).—Established a program to provide assistance through States to eligible weed management entities to control or eradicate harmful, nonnative weeds on public and private lands.

Noxious Weed Control and Eradication Act of 2004 (Public Law 108–412, 118 Stat. 2320).—Amended the Plant Protection Act to establish a grant program for financial and technical assistance to weed-management entities to control or eradicate harmful, invasive weeds on public and private lands. The law also authorizes USDA to enter into cooperative agreements with weed-management entities to fund weed-eradication activities and enable rapid response to outbreaks of noxious weeds.

Act of 1969 (NEPA; 42 U.S.C. 4321 et seq.). Often agencies are delayed from a timely response to an invasion or post-wildfire response because of NEPA requirements. The Western Weed Action Plan (Brown, 2019) highlights the need for identifying ways to comply with NEPA to facilitate invasive species management within the sagebrush biome in a more timely manner.

The Western Association of Fish and Wildlife Agencies published a report (Mayer and others, 2013) that evaluated challenges that hindered successful management of fire and invasive plant species in the West. This multi-agency, collaborative effort was followed by the U.S. Department of the Interior (DOI) Secretarial Order 3336 in 2015, "Rangeland Fire Prevention, Management and Restoration." The Integrated Rangeland Fire Management Strategy (IRFMS; U.S. Department of the Interior, 2015a) emphasized rangeland fire management as a critical priority for "protecting, conserving, and restoring the health of the sagebrush-steppe ecosystem and, in particular, greater sage-grouse habitat, while maintaining safe and efficient operations." A Federal interagency steering committee was formed to develop and implement policies and strategies for preventing and suppressing rangeland fire and for restoring sagebrush landscapes impacted by fire and invasive plant species.

In 2019, Secretarial Order 3372, driven by EO 13855, "Reducing Wildfire Risk on Department of Interior Land Through Active Management," included invasive species management and collaborative efforts among partners. Other products arose out of these collaborative efforts such as the interagency development of the DOI "Science Framework for the Conservation and Restoration of the Sagebrush Biome" (Chambers and others, 2017a) and "Western Invasive Plant Management: An Action Plan for the Sagebrush Biome" (Brown, 2019) that provide an approach and tasks for prioritization of conservation delivery among all stakeholders.

In February 2016, the Federal government released "Safeguarding America's Land and Waters from Invasive Species: A National Framework for Early Detection and Rapid Response" (EDRR; https://www.doi.gov/sites/doi.gov/files/National%20EDRR%20Framework.pdf). This serves as a first step in the development and implementation of a national EDRR program. Part 1 of the Science Framework (Chambers and others, 2017a) identified the importance of enhancing EDRR capabilities and described the components of a strategic, multiscale approach for managing invasive plant species and other threats to the sagebrush biome.

State laws often allocate responsibility of invasive species management to counties. In most cases, county weed programs and districts are governed by a board of county commissioners and have established legal and personnel infrastructure to support local weed-control activities. County weed-control programs usually operate under State authorities and primarily function as local governmental entities to enforce noxious weed laws; with the county board of commissioners usually providing the legal authority and oversight. Private landowners and industry are legally required to comply with State weed laws, although enforcement is often limited. Resources are also limited, and often counties try to work with landowners and industry collaboratively rather than only through regulatory means.

Many Federal and State agencies contract weed management efforts out to local county weed management programs, whereby county employees perform weed control on State and Federal roadways and other sites. This approach has been described as a substantial and underlying flaw in western weed management (Mayer and others, 2013; Mayer, 2018). The private, local, county, State, and Federal invasive plant management program infrastructure provides insight into the similarities and significant differences in how various operational levels are organized and function. This variation reflects the inconsistency across the sagebrush biome in governance structures, policies, priorities, partnerships, available information, communication systems, and resources available for management.

Funding

Over 20 Federal departments and agencies have responsibilities, authorities, and programs that deal with some facet of terrestrial invasive plant management. Appropriated funds are allocated annually for invasive plant management under various natural resource programs across the Federal agencies. Programs vary in how they can obligate funds toward invasive species management projects with different stipulations for spending. Federal allocations are dependent on the congressional budgeting process and therefore may not be consistent or sustained across years. While there are mechanisms available (for example, National Invasive Species Council) to provide unified support for increasing Federal funding for invasive plant management and research, these mechanisms have not been successful in implementing a sustained funding program. Although the scale and long-term impact of invasive plant invasions across the sagebrush biome greatly exceeds that of wildfire, the perceived risk and threat of invasive plants has not reached the same level of recognition, operational infrastructure, and funding priority. Unlike fire prevention, invasive plant species prevention has not become a social norm in western States.

Investments in invasive plant species prevention and control could realize significant reductions in fire-related costs in some sagebrush landscapes, depending on the likelihood of restoration success, future probability of fire and other site-specific factors (Taylor, M.H., and others, 2013). When prioritizing landscapes for invasive plant control or other restoration efforts, the social costs of conservation action and inaction should be quantified and considered as well.

Federal funding for invasive plant management activities is lacking throughout the sagebrush biome and has resulted in the curtailment, or a significant reduction in scope and scale, of many Federal management and research programs. The lack of adequate Federal infrastructure, sustained multiyear funding, and operational capacity severely hampers the ability to effectively deal with threats from invasive plant species that degrade or eliminate sagebrush ecosystems across the western United States (Brown, 2019). Increasing operational capacity and streamlining regulatory mechanisms may lead to more effective invasive plant management and increase the ability of land managers to successfully address the spread of invasive plants.

Partnerships

Local weed management programs perform most of the on-the-ground weed control and public education throughout the sagebrush biome. These programs take a variety of forms, such as county weed programs, county weed districts, and Cooperative Weed Management Areas and Weed Management Areas (hereafter, CWMAs) of local landowners, State and Federal agencies, and volunteers working together. County weed programs receive some county funding and have regulatory authority but are bounded by county lines and tend to be found in States with a history of agriculture and institutionalized weed management (Hershdorfer and others, 2007). County weed districts employ taxation to fund weed control authorized by State statute or voter-approved legislation. Weed districts are usually governed by volunteer weed control boards that administer the noxious weed control program according to State weed laws. Weed boards also set county weed control priorities and adopt county noxious weed lists.

Counties often lack the staff and resources needed to coordinate activities across multijurisdictional lines in addition to performing their primary duties of weed control activities, educating the public, and enforcing local or State weed laws. Additional burden on counties can result from conducting weed management programs on behalf of Federal and State land-management agencies through contracts. While funding is provided, these projects can detract from the county's ability to effectively complete their own work. Local governing entities in some States also receive State and Federal grants that can bolster funding for county weed control infrastructure and personnel. However, grant funding may require the counties to focus on high-priority, State-regulated invasive plants (such as List A and List B noxious weeds) which may detract from local programs focusing on locally prevalent species to the detriment of sagebrush ecosystems.

In some States, county programs may help to coordinate weed control between agencies and neighboring landowners. Management activities are often performed with no shared, central goals for management or measurable benchmarks to demonstrate progress. In some cases, private landowners conduct weed control with little communication with the county weed control office. Local weed regulations and ordinances may not be regularly enforced owing to a lack of staff and funding, or alternatively to encourage cooperation and compliance (Hershdorfer and others, 2007; Kokotovich and Zeilinger, 2011). Cooperative approaches to weed management have emerged in response to this disconnect. CWMAs are cooperative partnerships between neighboring private and public land managers that develop and employ strategies to manage weeds collectively within a common area. The CWMAs are local, multijurisdictional organizations across the country (see CWMA map developed by the North American Invasive Species Network [updated December 2018, https://www.naisn.org/cwmamap/]). Because CWMAs and county programs employ a localized and largely stakeholder-driven approach, management may be successfully carried out within an adaptive framework, allowing groups enough flexibility to incorporate new information or changing conditions.

Although the value of using such a cooperative approach across the landscape is well accepted, the sustainability and effectiveness of CWMAs is highly variable in the United States. The success of a CWMA often depends on the strength of the partnership agreements, the individual capacities of the partners, the ability to maintain consistent funding from year to year, and the personalities of the people involved. Faltering or failed CWMAs in the western United States are generally attributed to a lack of sustained funding and a lack of staff or volunteers. Either because CWMAs have not been initiated or because of failure of those that have initiated, there are gaps in cooperative weed management coverage across relatively large geographic areas.

CWMAs, as well as county weed programs and districts, play an important role in weed management. Although the effectiveness of local programs is sometimes equivocal, their success is critical to long-term ecosystem management on landscapes that vary dramatically in space and time within the sagebrush biome. The different program attributes that may contribute to invasive plant control efficacy include interagency coordination, strong local regulations and enforcement, funding, and volunteer participation.

Postfire Invasive Species Management

Land management agencies have a long history of rehabilitating burned rangelands using a variety of treatments designed to reduce the risk of postfire threats to life, property, and resource values. Prior to 1985, postfire rehabilitation practices focused on reducing erosion and flood potential. As the threat of invasive plant species increased, postfire rehabilitation policies and practices were changed to reduce the potential threats of nonnative plants. In 1985, BLM policy was changed to allow land use plan objectives to be used in developing rehabilitation objectives (for example, reestablish existing wildlife habitat) and to seed fire resistant plant materials if invasive plant species would establish (Bureau of Land Management, 2007).

Current DOI policy includes reducing the postfire threat of invasive plants. For example, BLM's Emergency Stabilization and Rehabilitation (ESR) program (Bureau of Land Management, 2007) addresses invasive plants in both the stabilization (treatments conducted in the first year following a wildfire) and rehabilitation (efforts conducted within 5 years of fire containment to repair or improve fire-damaged lands). Stabilization treatments may now include herbicide applications as a form of site preparation to control invasive annuals prior to seeding desirable vegetation. Rehabilitation treatments also include subsequent noxious weed inventories, control efforts, and monitoring for up to 5 years. Washing vehicles and equipment that has been used to prevent the spread of invasive species is required. A revised BLM ESR Handbook is being developed to incorporate the integration of ESR activities with multiyear investments in the restoration of fire-damaged lands, large-scale herbicide treatments, and activities to control invasive annual grasses. Better control of these invasive species will improve the success of treatments through better use of science and research and will improve the availability of native plant seed and the success in establishing native plants after wildfires.

The USDA Forest Service's Burned Area Emergency Response (BAER) program is designed to identify imminent postwildfire threats and take immediate actions, as appropriate, to manage unacceptable risks. This includes postfire invasive plant species detection surveys and, if warranted, invasive plant species rapid response treatments within the first year following fire containment.

The Land Treatment Digital Library is a USGS spatially explicit database of land treatments that were implemented by the BLM between 1940 and 2015. Of the 4,580 projects implemented by BLM, the majority (43 percent) were postfire rehabilitation treatments (Pilliod and others, 2017b), which included invasive species management. A new tool, the Land Treatment Exploration Tool (https://www.usgs.gov/centers/fresc/science/land-treatment-exploration-tool) taps into the Land Treatment Digital Library to use past treatment results to facilitate adaptive management of new land treatments, such as postfire rehabilitation projects.

The effectiveness of postfire rehabilitation projects in meeting invasive plant-management objectives has not been extensively studied. A review of 13 published papers to evaluate the effectiveness of postfire seedings in reducing the invasion and dominance of invasive plant species by Pyke and others (2013) found that 26 percent of the seedings reduced, 68 percent were neutral regarding, and 5 percent increased the cover, frequency, or density of invasive plant species (primarily cheatgrass). Reduction of weed abundance as a result of postfire seeding is highly variable and related to the conditions of the site and the plants established by seeding or recovery of residual plants after the fire. An evaluation of drill seeding perennial bunchgrasses on 61 BLM ESR projects in the Great Basin found it was only effective in reducing invasive plants (primarily nonnative *Bromus* species) on higher elevation sites with higher precipitation (Knutson and others, 2014). Seeding treatments at lower elevation sites with lower precipitation were more likely to be dominated by invasive annual grasses because of a lower establishment of perennial grasses.

Research and Restoration

Most invasive plant species research related to the sagebrush ecosystem has shifted from evaluating methods of direct control of invasive annual grasses toward understanding, sustaining, and restoring ecosystems that are resilient to disturbances and resistant to invasive plants, especially annual grasses, as is described in chap. R (this volume). The development of resistance and resilience maps using soil resources as developed and proposed by Chambers and others (2016a) has encouraged management, restoration (Pyke and others, 2015a), and research applications of the concepts. State-and-transition models with common disturbance responses are also being grouped to aid managers in understanding common threats to the ecosystem and potential management solutions for maintaining resistance to invasive plant species (Stringham and others, 2016).

Maintenance of perennial herbaceous species, especially native perennial grasses, has become an area of emphasis within the literature (Chambers and others, 2017a; Strand and others, 2017). Best practices regarding the restoration of native perennial bunchgrasses has moved from direct seeding to applying chemicals or hormones to enhance seeding or to combine seeding with application of herbicides to control invasive plants (Madsen and others, 2016; Davies and Johnson, 2017; Davies and others, 2017, 2018). Reestablishment of biological soil crusts (Condon and Pyke, 2016, 2018) is an emerging restoration and management approach focused on building and maintaining resilience and resistance.

The Actionable Science Plan (Integrated Rangeland Fire Management Strategy Actionable Science Plan Team, 2016) summarizes the priority science needs related to the control of invasive plant species with an emphasis on cheatgrass. The IRFMS (U.S. Department of the Interior, 2015a) and Actionable Science Plan were developed, in part, to help prioritize and direct research across the sagebrush biome.

Management for Threat Abatement

Prevention

Prevention has been shown to be one of the most economical approaches to managing invasive plant species (Leung and others, 2002). Although prevention is an economically viable management strategy, it is difficult to support because of the need to fund management activities where invasions are already a significant problem. This is partly owing to the difficulty in quantifying the economic return of the prevention of an invasion. Most funding agencies report invasive plant management success in acres that have already been treated. In an attempt to create a comparative metric to evaluate treated acres, some local level groups use the concept of protected acres to show the success of a prevention program in comparison to a post-hoc management program. This metric is intended to show the number of acres that have been saved from a future invasion owing to prevention. Although this approach could be a viable alternative or comparator to the "acres treated" metric, there is no universally accepted empirical method to determine how to calculate protected acres and how to compare this to treated acres.

One approach to invasive plant prevention is controlling point sources of propagules. An example of this approach is the National Weed Free Forage and Gravel Program and supporting standards developed by the North American Invasive Species Management Association. This program certifies both hay and gravel as weed free, allowing them to be used in areas considered high priority or that have high susceptibility to invasion. This program has been adopted by 28 States and Canadian provinces. Programs such as this cannot be successful without widespread to universal adoption from land-management agencies and other landowners because seed movement cannot be mitigated if all surrounding areas are not committed to the same standards. Currently, most Federal land-management agencies have not developed a clear policy for the use of a similar prevention program but do include the concepts as best management practices.

Another current prevention campaign, PlayCleanGo, educates the public on responsible outdoor recreation practices in the face of invasive plants with the message of cleaning outdoor equipment before departing an area. Unlike EDRR, this program does not require expertise in plant or seed identification, meaning the public can be active members in promoting and implementing these efforts. However, the success of programs like PlayCleanGo is difficult to measure, and there may be reluctance to adopt programs that have not proven to be successful. Although over 500 partners have adopted this messaging, much like weed-free forage, there is not wide-scale adoption of this program from Federal land-management agencies. The U.S. Fish and Wildlife Service Refuges and Fish and Aquatic Conservation Programs have been working with the National Park Service, the North America Invasive Species Management Association, and Wildlife Forever to develop a national invasive species prevention memorandum of understanding (MOU) in order to work more closely together to prevent the establishment of invasive species within Federal lands and waters. The MOU will be open for other parties to sign on in the future. The specific prevention program is not necessarily critical to the successful early management of invasions; however, a clear regional if not national priority on the prevention of invasive plant introduction and movement could be one of the best programmatic shifts for invasive plant management.

Early Detection and Rapid Response

Once established, invasions are difficult to fully eradicate (Rejmánek and Pitcairn, 2002). Invasions should be addressed as early as possible before a threshold is reached where perpetual management is needed to contain the species. Unfortunately, techniques to identify this threshold for any invasive plant species in any particular system are not available. Until this information is available, active monitoring for new invasions with associated response programs is the best measure to prevent spread.

Early detection and rapid response efforts are most important in areas that are highly susceptible to invasion, such as sagebrush communities that are three times more susceptible to invasion than any other associated ecosystem, in part because of the prevalence of roadways that act as propagule vectors in the region (Pollnac and others, 2012). Detection of new invasions is difficult because, unlike a wildfire that is easy to identify and is universally considered threatening to humans, plants are difficult to identify and do not pose an immediate threat (Dewey and others, 1995). Thus, immediate or early detection is more difficult and a response, if any, often occurs after a well-established population of the invasive plant exists. The low human population density of much of the western United States makes prompt detection even more difficult. For example, in Wyoming, if expectations for detection were evenly distributed across its population, every person would be responsible for monitoring approximately 100 acres.

There is no universal reporting mechanism to alert land managers of nascent invasions. This is a needed item as the number of on-the-ground land managers decreases, meaning the number of acres being monitored per manager is increasing. Thus, the public is becoming a more important member in the process of early detection. Unless there is a clear, direct, and ideally universal mechanism for reporting such threats, the likelihood of a new report being directed to the appropriate agency is unlikely. Remote-sensing technologies can increase the number of acres that can be visually monitored by an individual. Far-earth technologies have only recently been able to identify large-scale invasive plant populations, often beyond the point of early detection (Müllerová and others, 2016). They are also limited by seasonal flexibility of imagery and common obstructions such as cloud cover and vegetation layering. New near-earth technologies that use smaller platforms such as unmanned aerial systems are reducing the cost of data collection, increasing image resolution, and increasing potential image redundancy (Lowman and Voirin, 2016). They are still limited in area that can be covered and in the backend technologies to efficiently interpret large aerial images. However, these challenges may be addressed quickly given the current rate of evolution of these systems.

Identifying the areas that are most susceptible to invasions could potentially reduce the acreage needing monitoring and create a framework for systematic monitoring that more efficiently allocates resources and time. Niche modeling aims to identify where a species is likely and unlikely to invade (Petitpierre and others, 2012). Distribution data can be used to develop such models and can periodically be updated as new distribution data is collected.

Currently, systems for rapid response are not standardized across management units or agencies. Therefore, even when detection of nascent invasions is successful, a management response is not guaranteed. For example, once a new single invasive plant population is identified, there is no clear protocol on surveying the population's spatial extent beyond the initial identification, which can result in improper initial management. A standardized system of early detection reporting and rapid response would help in avoiding scenarios where known invasive plant populations go unmanaged.

Realistic goals are important to correctly allocating resources. If an invasion is beyond the early detection and rapid response phase and eradication is no longer feasible, goals must shift to recognize this and respond accordingly. Unlike eradication commitments which may require as little as a few years to successful completion, the mitigation of impacts and spread of invasive plant species is a long-term commitment. Because long-term financing for restoration is difficult to acquire, responding before perpetual management is required is critical to the long-term success of managing invasive plant species in sagebrush communities.

Single Species Versus Multi-Invasion Management and Weed Succession

Many invasive plant-management programs focus on a particular species. For example, in Wyoming, all special management programs at the county level are developed to target a single species. However, this may not be the best approach for systems that have many well-established invasive plant species. Instead, managing complexes of well-established invasive plant species may benefit from multispecies management programs. The majority (69 percent) of restoration and conservation efforts across the West focused on invasive plant management are working in systems that are not invaded by a single invasive plant species but are instead invaded by many species (Kuebbing and others, 2013).

The cumulative effects of multiple invasions on plant community dynamics is not well understood. Multiple invaders may create a greater negative impact to the system than either invader alone (often referred to as synergies), or they may act additively (Tekiela and Barney, 2017). If synergies exist between invaders, then management of both species is far more beneficial than targeting only one invader. This has been shown in systems of the eastern United States (Kuebbing and others, 2016), but no information exists on the potential combined impact of the most problematic invasive plant species in sagebrush ecosystems. Alternatively, synergies may not exist and impacts of invaders may not be cumulative (Pearson and others, 2016a). A holistic approach to management is important as targeting a single species for control while ignoring another can lead to secondary invasion (Pearson and others, 2016b) or the "invasion treadmill," (Thomas and Reid, 2007) where each attempt at controlling an invasion leads to another invasive plant species becoming dominant.

Thresholds and Treatments

Efficient management of invasive plants requires understanding thresholds in the likelihood of (or resistance to) invasion in addition to understanding threshold levels of abundance of exotic plants for desirable ecosystem functioning. These thresholds are also needed to establish objectives for treatments because they are the basis for determining treatment success. As an example, following the 2015 Soda Fire in Idaho and Oregon, the BLM, along with many agency partners, determined that the unifying objective of treatments was to promote resistance to annual grass invasion and resilience to future fires. Partners determined that this was most likely to occur if exotic annual grasses were less than 20 percent and perennial grasses were greater than 20 percent of total plant community cover and if sites trended toward greater dominance of perennial grasses over time.

Thresholds were also created based on the abundance and size (mean height and basal diameters) of perennial grasses, which were used to gauge their maturity and their ability to provide resistance and resilience in light of grazing, drought, reburning, and other disturbances (Germino and others, 2019). These thresholds were used to judge treatments as being successful or not. Moreover, the thresholds were initially proposed to be the basis for deciding whether resumption of livestock grazing should be permitted. However, the agency partners also recognized the breadth and depth of the data in supporting science publications were quite limited compared to the environmental and taxonomic variability within the rugged, greater than 100,000-hectare (247,097-acre) landscape. As a result, the vegetation data were used to guide management decisions, but adherence to thresholds was loosened for the BLM and other partners.

Prioritizing Areas for Management

The expansion of invasive plants continues, and land managers, communities, States, and agencies are forced to select management areas to apply resources, treatments, and restoration efforts. Areas are prioritized for proactive invasive plant management and for response to disturbance based on several factors ranging from resources of concern in an area, community needs, size of invasion, and willing partners to opportunities for success based on the resilience to recover and resistance to re-invasion of a site. Often groups and agencies do not have the same objectives or measures of success, which prevents successful collaborative conservation or successful threat reduction. Limitations of management persistence over several years and long-term funding (Mayer, 2018) force the targeting of leveraged resources for restoration efforts in high-priority areas (Brown, 2019).

The Science Framework (Chambers and others, 2017a) provides an approach for prioritizing areas for effective management. Although the approach was developed with a focus on invasive annual grasses, it is applicable to other invasive plants where information exists on the environmental characteristics necessary for their establishment, growth, reproduction, and persistence. The framework's approach is based on (1) the likely response of an area to disturbance or stress because of threats and management actions (that is, resilience to disturbance and resistance to invasive annual grasses), (2) the capacity of an area to support the target species or resources, and (3) the predominant threats. A geospatial process can then be used that involves overlaying key data layers including resilience and resistance to invasive annual grasses as indicated by soil temperature and moisture regimes (Maestas and others, 2016). Geospatial data on invasive plant species distribution and abundance can be used in conjunction with other threats in the analyses to (1) evaluate the level of risk of vegetation types and communities to invasion, (2) further refine target areas for management, and (3) determine the most appropriate type of management actions. Applying tools like this can help managers make informed decisions on approaches to address the invasive threat (Crist and others, 2019).

Site-Specific Management Options for Invasive Plants in Sagebrush Ecosystems

Control options for invasive plants in upland habitats are either broadly applied to landscapes or are precisely targeted and fall into the following categories: (1) reduction with mechanical or other disturbance (such as prescribed fire or targeted grazing), (2) eradication with chemicals or biological control agents, or (3) augmentation of community resistance to invasion by maintaining intact native plant communities (Sheley and Smith, 2012; Crist and others, 2019). Using principles of integrated pest management, many projects use more than one of these forms of control and often apply them in stages. Projects that enable multiple interventions over multiple years are especially important in variable environments, such as the vast areas of the western United States that receive less than 12 inches of annual precipitation and, more importantly, that have exceptionally high year-to-year variability in precipitation and thus in productivity (Hardegree and others, 2018).

For species such as exotic forbs, spot-spraying of postemergent herbicides such as aminopyralids (for example, Milestone), along with biocontrol releases such as insects (for example, flower weevils [*Larinus* spp.] for knapweeds), are the most common management responses. These approaches are successful in thwarting new invasions and at least temporarily reducing exotic forbs in more disturbed sites with high seed pressure. Compared to the mentioned responses, there is less focus on preemptive management or restoration of native or resident plants to bolster plant community resistance to exotic forb invasion. Such an approach would focus on establishment of a diverse community of species with different rooting and phenology rather than restoring sites to just low-diversity grass communities.

For exotic annual grasses, a different approach with herbicides is possible, in which pre-emergent herbicides are used selectively to kill emerging seedlings. Herbicides such as imazapic reliably reduce exotic annual grasses with minimal impact to perennials (Applestein and others, 2018a), although studies are still needed on how to coordinate herbicide use with desired seedings. Indaziflam is a relatively new pre-emergent that may provide a longer control period than imazapic. Cheatgrass control for longer periods can deplete the seed bank and may increase rangeland recovery success. Biocontrol options for exotic annual grasses are currently limited. While cheatgrass die-off is known to cause stand failure over large areas as a result of the interactive effects of five fungal pathogens, these pathogens are highly dependent on weather and thus are not perceived to be useful as a planned bioherbicide (Meyer and others, 2016). Restoration following natural cheatgrass die-offs presents an unplanned (and thus difficult to capture) opportunity (Baughman and others, 2017).

A number of pathogens affect exotic annual grasses. Fungi on cheatgrass (Meyer and others, 2016) and weed-suppressive bacteria (WSB) have emerged as prospective, but highly uncertain, biopesticides for reducing exotic annual grasses in sagebrush and other ecosystems (Germino and Lazarus, 2020). Screening of strains of the soil bacterium *Pseudomonas fluorescen*s have revealed strains (D7, ACK55) that under controlled lab conditions (petri dishes) apparently suppress the root growth of exotic annual grasses but not of native species. While ACK55 has apparently shown desired effects in the field (Kennedy, 2017, 2018), many other studies on strains D7 and MB906 did not demonstrate an effect across a network of experimental sites (reviewed by Germino and Lazarus, 2020). It is unclear why other studies have not been able to replicate the effects shown in the two studies in the field (Kennedy, 2017, 2018), although USGS scientists have recently reproduced the effects of ACK55 in petri dishes (Lazarus and others, 2020). These new studies indicate that competition with native soil microbes is unlikely to be the reason why these weed-suppressive bacteria fail in soil. Instead some other factor may be affecting their viability in real soil and effects on plants in the field.

The hope for WSB is that it can provide a seminatural, nonchemical, and selective reduction of exotic annual grasses that lasts longer than the short-term effects of herbicides. This enduring effect is highly desired because it would allow for (1) bunchgrass recovery (which generally requires more than 2 years) and (2) prolonged relief from annual grass competition. Currently, adequate information on the basic biology and ecology of soil microorganisms in sagebrush ecosystems is lacking and any use of WSB should be experimental in nature. D7 is a registered biopesticide and efforts to register ACK55 are underway at the time of this writing.

Seeding of perennial grasses is common in disturbed areas where perennials have been depleted and provides competitive pressure against exotic annuals (chap. R, this volume). However, these treatments may create grasslands of often nonnative grasses, such as crested wheatgrass, or cultivars of native plants, such as Snake River wheatgrass (*Elymus wawawaiensis*) that both alter wildlife habitat value and are difficult to diversify (Pyke and others, 2015a). That said, perennial grasslands are likely preferable to monocultures of invasive annual grasses and attendant increases in wildfire, which may in turn jeopardize relatively intact plant communities within a landscape. Within the annual grass zone, decisions on plant materials used in reseeding efforts involve balancing the likelihood of exotic annual grass dominance with establishment likelihood of seeded species and the contribution of seeded species to invasion resistance.

Seeding can be done using aerial broadcast or drill seeding. Drill seeding leads to higher seeding success but also imposes a soil disturbance that can increase exotic annuals if the seeded species do not persist. Chain harrowing after aerial broadcast is commonly used to provide soil burial of seeds on steeper slopes. Following these treatments, rest from livestock is considered essential to protect young seedlings. However, there is currently little guidance on how to determine when bunchgrasses have become mature enough to withstand grazing (and drought and other stresses) and still provide resistance to invasion. Also, protecting seeding investments from reburning helps ensure that the ratio of annual to perennial grasses does not increase. This protection may, in some cases, be partly attained by installing fuel breaks (Shinneman and others, 2019) or through the use of prescribed livestock grazing (Davies and others, 2015).

Appendix K1. Nonnative Invasive Plants in Sagebrush Ecosystems

Table K1.1. Nonnative invasive plants in sagebrush (*Artemisia* spp.) ecosystems listed from highly invasive to weakly invasive (from Crist and others, 2019, app. 3, p. 233).

[EDRR, early detection and rapid response; CA, California; CO, Colorado; ID, Idaho; MT, Montana; NV, Nevada; OR, Oregon; SD, South Dakota; UT, Utah; WA, Washington; WY, Wyoming]

Plant	Scientific name	EDRR potential in sagebrush habitat	Habitat	Negative impacts
Medusahead	*Taeniatherum caput-medusae*	CA, CO, ID, MT, NV, UT, SD, WA, WY	Occurs in sagebrush-grass or bunchgrass communities that receive at least 9–12 inches (23–30 centimeters) of precipitation; often invades after disturbance; does well in clay soils that shrink, swell, and crack and openings in chaparral vegetation types	Low palatability for livestock owing to high silica content, which confers competitive advantage over native plants; awns can injure eyes and mouths of animals; dense, long-lasting litter layer creates fire risk and reduces seed germination of other species
Cheatgrass	*Bromus tectorum*	Local and regional EDRR potential	Wide ecological amplitude from salt desert in the Great Basin to coniferous forests in the Rocky Mountains; areas in which most precipitation arrives in late winter or early spring are most susceptible; often occurs in disturbed areas and areas with dry sandy soils with little competition	Increases fine fuels and fire risk; can outcompete many perennial native plant species and replace many annual species; reduces production of perennial grasses for livestock forage but can be grazed in winter or spring; sharp seeds may cause eye injuries
North Africa grass	*Ventenata dubia*	CA, CO, ID, MT, NV, SD, UT, WA, WY	Occurs in bunchgrass, sagebrush, and meadow communities	Can outcompete perennial bunchgrasses; low palatability for livestock owing to high silica content; matures early in the season and is likely to pose fire risks
Spotted knapweed	*Centaurea maculosa*	CA, NV, OR[1], SD, UT, NV, WA	Occurs over a wide range of elevation and annual precipitation; does well in forest-grassland interface on deep, well-developed soils, with dense stands occurring in moist areas on well-drained soils including fields, roadsides, and disturbed and degraded rangeland	Very competitive and can form dense stands that result in higher surface-water runoff and soil erosion; excludes desirable vegetation, thereby reducing livestock and wildlife forage
Yellow star-thistle	*Centaurea solstitialis*	CA, CO, MT, NV, OR[1], SD, UT, WY	Occurs on deep, loamy soils and south-facing slopes with 12–25 inches (30–64 centimeters) of annual precipitation; found in open disturbed sites, rangeland, roadsides, and open woodlands	Highly competitive and develops dense, impenetrable stands. Reduces forage production for livestock and wildlife; can be grazed before spine development, but poisonous to horses
Iberian starthistle	*Centaurea iberica*	CA, CO, ID, MT, NV, OR, SD, UT, WA, WY	Occurs on riverine banks, along watercourses and in other moist areas	Unpalatable—spines restrict access to the plant and deter grazing

Table K1.1. Nonnative invasive plants in sagebrush (*Artemisia* spp.) ecosystems listed from highly invasive to weakly invasive (from Crist and others, 2019, app. 3, p. 233).—Continued

[EDRR, early detection and rapid response; CA, California; CO, Colorado; ID, Idaho; MT, Montana; NV, Nevada; OR, Oregon; SD, South Dakota; UT, Utah; WA, Washington; WY, Wyoming]

Plant	Scientific name	EDRR potential in sagebrush habitat	Habitat	Negative impacts
Purple starthistle	*Centaurea calcitrapa*	CA, CO, ID, MT, NV, OR, SD, UT, WA, WY	Can inhabit a wide range of conditions, including fertile alluvial soils, pasture, range, open forest, and riparian areas	Unpalatable—spines restrict access to the plant and deter grazing
Diffuse knapweed	*Centaurea diffusa*	CA, NV, OR[1], SD, UT	Wide ecological amplitude for elevation, aspect, slope, and soil properties; maximum invasiveness is in shrub steppe, rangelands, and forested benchlands; often occurs on well-drained soils	Increases soil erosion and surface runoff; replaces wildlife and livestock forage but has some forage value through the bolting stage; dispersal similar to tumbleweeds (for example, *Salsola tragus*)
Leafy spurge	*Euphorbia esula*	CA, NV, OR[1], UT, WA	Found in disturbed sites, roadsides, rangelands, and riparian areas with semiarid to mesic conditions. It has wide ecological amplitude and occurs on many soil types; high genetic variability allows it to easily adapt to local growing conditions	Highly competitive and can form dense clones that suppress native plants and reduce forage; milky sap is toxic and can irritate skin, eyes, and digestive tracts of humans and other animals; sheep and goats graze it and can tolerate the toxins
Rush skeletonweed	*Chondrilla juncea*	CA, CO, MT, NV, SD, WY	Found in rangelands and pastures and along roadsides; occurs in very dry to very wet environments on disturbed soils and well-drained, sandy textured, or rocky soils	Can form dense monocultures and displace native plants, reduce livestock forage, and spread from rangeland to adjacent cropland; wiry stems can interfere with harvest machinery
Dalmatian toadflax	*Linaria dalmatica*	CA, NV, SD, WA	Tolerates many soil types and is found on well-drained, coarse-textured soils and sandy loams, as well as heavier soils; does best in cool, semiarid climates on dry, coarse soils with neutral to slightly alkaline pH and south- to southeast-facing slopes; occurs in rangelands, disturbed areas, roadsides, and forest clearings; can move into undisturbed prairies and riparian habitats	Aggressive invader capable of forming dense colonies and outcompeting native grasses and other perennials; decreases forage for livestock and wildlife; if sufficient quantities are ingested, quinazoline alkaloids can pose toxicity problems to livestock, but goats and sheep are tolerant; can increase soil erosion, surface runoff, and sediment yield in invaded bunchgrass communities
Sulphur cinquefoil	*Potentilla recta*	CA, ID, NV, SD, UT, WA, WY	Wide ecological amplitude. Found in conifer, grassland, shrubland, and seasonal wetland ecosystems; occurs along roadsides and in other disturbed sites, but also will invade low-disturbance sites	Low palatability for grazing animals, possibly from phenolic tannins in leaves and stems; can become a dominant part of plant communities
Russian knapweed	*Rhaponticum repens*	NV, OR[1], WA	Found in pastures, in rangelands, and along streambanks and roadsides; will invade croplands; occurs on many soil types but prefers moist soils that are not excessively wet	Allelopathic and very competitive, forming dense stands; reduces forage for livestock; low palatability for livestock and toxic to horses

Table K1.1. Nonnative invasive plants in sagebrush (*Artemisia* spp.) ecosystems listed from highly invasive to weakly invasive (from Crist and others, 2019, app. 3, p. 233).—Continued

[EDRR, early detection and rapid response; CA, California; CO, Colorado; ID, Idaho; MT, Montana; NV, Nevada; OR, Oregon; SD, South Dakota; UT, Utah; WA, Washington; WY, Wyoming]

Plant	Scientific name	EDRR potential in sagebrush habitat	Habitat	Negative impacts
Squarrose knapweed	*Centaurea virgata*	CA, CO, ID, MT, NV, OR, SD, WA, WY	Found in fields, roadsides, disturbed sites, grasslands, and big sagebrush (*Artemisia tridentata*) bunchgrass- and juniper (*Juniperus* spp.)-dominated rangelands; extends into salt desert shrub, particularly in sandy or gravelly washes and on dry, rocky, south-facing slopes; will invade fairly pristine mountain brush types and juniper/Idaho fescue (*Festuca idahoensis*) rangeland; also, will invade abandoned dry wheat (*Triticum* spp,) fields, crested wheatgrass (*Agropyron cristatum*) seedings, burned areas, and improperly grazed areas	Highly competitive; can endure drought at either temperature extreme, is fire tolerant, and has excellent seed dispersal and rapid response to soil resources released by fire; rosettes grow slowly for years before flowering, creating basically a vegetative seedbank; similar palatability and nutritive value to diffuse or spotted knapweed; sheep and cattle may graze it when other annual forage is sparse; dense stands can exclude desirable vegetation and wildlife in natural areas
Whitetop, hoary cress	*Lepidium draba*	Not listed as an EDRR species by any of the States	Found in disturbed open sites, on ditch banks, and along roadsides; well-adapted to moist habitats, especially subirrigated rangeland, pastures, wetlands, and riparian areas; tolerates a wide range of soil types and moisture conditions; often found in disturbed areas with other invasive species	Can form dense monocultures and is difficult to control owing to large and deep roots and rhizomes; can dramatically reduce biodiversity and forage production and can invade cropland and reduce yields; plants contain glucosinolates, which can form toxic compounds; unpalatable to livestock
Yellow toadflax	*Linaria vulgaris*	CA, SD, UT	Found in riparian areas, rangeland, disturbed areas, roadsides, and forest clearings; often occurs on moister sites; tolerates many soil types varying from coarse gravels to sandy loams but is also found in heavier soils; can move into undisturbed prairies and riparian habitats	Highly competitive for soil moisture with winter annuals and shallow-rooted perennials; aggressive invader capable of forming dense colonies and outcompeting native grasses and perennials; decreases forage for livestock and wildlife; if sufficient quantities are ingested, quinazoline alkaloids can pose toxicity problems to livestock, but goats and sheep are tolerant
Dyer's woad	*Isatis tinctoria*	CO, MT, NV, SD, UT, WA, WY	Occurs in disturbed sites, roadsides, pastures, forests, and rangeland often on dry, rocky, or sandy soils; invades undisturbed natural areas as well as alfalfa (*Medicago* spp.) and small grain fields; also found along waterways; adapted to the arid climate and alkaline soils of the West	Palatable to cattle only before bolting; grazing can be done before flowering to minimize seed production; can spread at an annual rate of 14 percent and reduce grazing capacity by an average of 38 percent; capable of invading and increasing density on well-vegetated range sites even in the absence of grazing or disturbance

Table K1.1. Nonnative invasive plants in sagebrush (*Artemisia* spp.) ecosystems listed from highly invasive to weakly invasive (from Crist and others, 2019, app. 3, p. 233).—Continued

[EDRR, early detection and rapid response; CA, California; CO, Colorado; ID, Idaho; MT, Montana; NV, Nevada; OR, Oregon; SD, South Dakota; UT, Utah; WA, Washington; WY, Wyoming]

Plant	Scientific name	EDRR potential in sagebrush habitat	Habitat	Negative impacts
Mediterranean sage	*Salvia aethiopis*	CO, ID, MT, NV, UT, WA, WY	Found in degraded big sagebrush communities, rangeland, openings in ponderosa pine (*Pinus ponderosa*), and disturbed sites, including roadsides; also occurs in floodplain and riparian areas following overgrazing, excessive trampling, and soil erosion; often inhabits moderate to deeper soils with good drainage; often associated with sites dominated by annual grasses	Unpalatable to grazing animals and, although not considered toxic, reduces forage production on rangeland and pastures; tumbleweed mobility facilitates rapid spread in degraded communities; may attain understory dominance in sagebrush/cheatgrass communities
Scotch thistle	*Onopordum acanthium*	CA, WA	Found in disturbed areas, rangeland, forest clearings, abandoned cropland, high rodent activity areas, and along river and stream corridors and roadsides; best suited to areas with high soil moisture during germination; often associated with cheatgrass	Can form dense stands over large acreages and decrease desirable forage; sharp spines deter livestock and wildlife from grazing; dense stands can prevent movement by livestock, wildlife, and humans; grazing of young plants may occur in early stages of infestation, but overgrazing promotes Scotch thistle
Barilla, saltlover (known as halogeton)	*Halogeton glomeratus*	CA, NV, SD, WA	Occurs in dry, arid regions and is adapted primarily to alkaline and saline soils	Foliage contains soluble sodium oxalates and can be toxic to livestock, especially sheep, when large quantities are ingested
Musk thistle	*Carduus nutans*	CA, WA	Found in disturbed open sites, roadsides, pastures, and annual grasslands; occurs over a wide range of environmental conditions, ranging from saline soils in low elevation valleys to acidic soils in high elevations; potentially intolerant of shading from neighboring plants	Can form dense stands over large areas and decrease desirable forage; sharp spines deter livestock and wildlife from grazing; dense stands can prevent movement by livestock, wildlife, and humans; allelopathy can reduce growth of desirable pasture species in an area much greater in diameter than the musk thistles themselves; may take 15 years of treatment to decrease germination
Common crupina	*Crupina vulgaris*	CA, CO, ID, MT, NV, SD, UT, WA, WY	Occurs in grasslands, pastures, rangeland, canyons, disturbed riparian areas, and gravel pits; adapted to many temperature and moisture regimes and soil types; infests sites with cheatgrass	Highly competitive for limited soil moisture; dense populations reduce and displace desirable forage species for livestock and wildlife and can contaminate hay; seeds can survive ingestion by animals and remain viable in soil up to 3 years; most livestock avoid grazing; can displace perennial bunchgrasses and lead to soil erosion because of less effective soil stabilization

[1]Oregon species that is a State-listed B-noxious weed and is established in some areas. However, in areas that are currently known to lack the listed invader, it is considered an EDRR species.

Chapter L. Climate Adaptation

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Executive Summary

Increases in the frequency and magnitude of extreme climate events in the 21st century likely will create more ecologically significant droughts (especially hot droughts) and floods than experienced in the recent past. However, because there is substantial variability across climate projections among models, across seasons, and across space, the models help with understanding possible scenarios and possible outcomes affecting ecosystems and humans. All 10 climate models examined in this chapter project increases in temperature, and the magnitude of increase (1–3 degrees Celsius [°C; 1.8–5.4 degrees Fahrenheit (°F)] between 2020 and 2050, 2–5 °C [3.6–9.0 °F] or as much as 3–7 °C [5.4–12.6 °F] for 2070–2100) is reasonably consistent across seasons and locations, whereas approximately 90 percent of these models indicate slight increases in precipitation.

The interaction of rising temperatures and potential modest increases in precipitation are expected to influence patterns of drought and moisture availability within the sagebrush (*Artemisia* spp.) biome. Cool-season recharge of soil moisture is likely to be sustained, although more precipitation will come as rain, potentially resulting in higher moisture availability earlier in the year. However, warmer temperatures will prompt earlier soil drying, leading to longer periods of hot and dry conditions in summer. Climate projections indicate that large decreases in the abundance of sagebrush will occur in the hottest and driest regions within the sagebrush biome, but the geographic extent of loss is uncertain. Furthermore, potential increases in the abundance of cheatgrass (*Bromus tectorum*) are likely in cooler, wetter parts of that species' range, and decreases are likely in the hottest and driest parts of its range. However, those hot and dry locations may be vulnerable to invasion by other nonnative annuals such as red brome (*B. rubens*). Fewer days with precipitation in summer and declines in overall summer precipitation have likely contributed to recent increases in the amount of sagebrush burned. In the next 30–40 years, longer and hotter fire seasons, and more extreme fire weather are predicted to lead to a significant increase in the probability of very large fires, particularly in the Pacific Northwest.

The ecological importance of riparian zones, seeps, springs, and other wetlands are disproportionately large relative to their size. Similarly, climate change and other anthropogenic impacts on mesic systems may affect ecosystem function disproportionately, especially if these systems serve as local buffers and climate refugia. Native animal species' ability to persist as climate changes likely will depend on their phenotypic plasticity and evolutionary rates. Land use, including human appropriation of water and activities that fragment native vegetation or open space, may further constrict adaptive responses. Climate-driven stresses also are likely to impact the capacity to support herds of domestic livestock, although human intervention in breeding, nutrition, and movement may reduce the effects of climate change on livestock compared to the effects on most native species. Climate adaptation strategies include informed selection of seed sources for restoration and consideration of resistance and resilience information when prioritizing areas for restoration or other management.

Introduction

Average annual temperature over the contiguous United States has increased by 0.7 degrees Celsius (°C; 1.2 degrees Fahrenheit (°F) for the period 1986–2016 compared to 1901–1960 (Vose and others, 2017). Warming temperatures, increased frequency of heat waves, and possibly drought have likely contributed to longer fire seasons, more extreme fire weather, and consequently, larger amounts of sagebrush (*Artemisia* spp.) burned each year. Future climate warming and alterations in timing of seasonal precipitation may impact the distribution of sagebrush and invasive plants, and further increase the frequency and severity of fires and duration of fire seasons. The degree and spatial extent of these impacts of warming climates on the sagebrush biome will depend on the degree and rate of warming and changes in timing and amount of precipitation.

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Climate Change Trajectories and Impacts

Climate Projections

Details on projected changes in climate across several ecoregions encompassing the sagebrush biome are provided by Chambers and others (2017a; see sec. 5.2 and app. 3). Representative concentration pathways (RCPs) are scenarios used for global climate projections. These scenarios include time series of emissions and concentrations of the full suite of greenhouse gases (GHGs) and aerosols and chemically active gases, as well as land use/land cover (see https://www.ipcc-data.org/guidelines/pages/glossary/glossary_r.html).

In this chapter, the results are summarized for a representative set of climate models that simulate two general climate scenarios: moderate increases in greenhouse gas emissions (RCP4.5) and more substantial increases (RCP8.5). Over the entire sagebrush biome, climate models simulating both RCPs project average increases in temperature of 1–3 °C (1.8–5.4 °F) in the near term (2020–2050) and increases in average temperatures of 2–5 °C (3.6–9.0 °F) under RCP4.5 and 3–7 °C (5.4–12.6 °F) under RCP8.5 in the far term (2070–2100). The models project that the greatest average temperature increases (more than 6 °C [10.8 °F] from 2070 to 2100 under RCP8.5) will occur in the center and far northeastern edge of the current range of big sagebrush (*A. tridentata*). Winter temperature increases are projected to be greatest in the northeastern part of big sagebrush range. Spring temperature increases, by contrast, are projected to be greatest in the central and southern part of the range.

Climate-change projections for precipitation in the sagebrush biome, and virtually all biomes, are more uncertain than projections of temperature change. Although the median projections indicate increasing mean annual precipitation—with the greatest increase (approximately 20 percent under RCP8.5) by the end of the century—different models project changes from a slight (less than [<] 10 percent) decrease to a 50 percent increase. Spring precipitation is projected to increase most in the northeastern part of the range of big sagebrush, and summer precipitation is projected to increase most in the southern and western range of big sagebrush. Most climate models project that the proportion of precipitation falling between May and October will decrease, especially in the northern part of the region. Projected historical and future values of these and other climate variables are available at https://www.sciencebase.gov/catalog/item/5850549ae4b0f24ebfd9368f.

A recent study described the current and projected 21st century climate changes at approximately 900 sites (Palmquist and others, 2016a), representing the current distribution of big sagebrush (Schlaepfer and others, 2012a). This study examined climate projections from 10 general circulation models (GCMs), a number likely to represent greater than (>) 80 percent of the variation in all climate models in CMIP5 (Coupled Model Intercomparison Project Phase 5—data source for climate data; McSweeney and Jones, 2016). The GCMs that were selected represent the most independent (Knutti and others, 2013) and best performing (for the western United States; Rupp and others, 2013) subset of GCMs. For these 900 sites, the mean annual temperature from 1980 to 2010 averaged 6.7 °C (44 °F) and is projected to increase 2.7 °C (4.9 °F) by 2030–2060 (range among 10 climate models used in this study: 1.9–3.3 °C [3.4–5.9 °F]) and 5.4 °C (9.7 °F) by 2070–2100 (ranges 4.7–6.5 °C [8.5–11.7 °F]). Mean annual precipitation at these sites averaged 353 millimeters (mm; 13.9 in.) from 1980 to 2010 and is projected to increase by 27 mm (1.1 in.) from 2030 to 2060 (ranges from −23 to 74 mm [−0.9–2.9 in.]; 90 percent of models projected increasing precipitation) and 45 mm (1.8 in.) from 2070 to 2100 (ranges from 1 to 156 mm [<1.0–6.1 in.]).

Climate Distributions and Extremes

Elevated temperature extremes have already been documented for the western United States and Canada (Vose and others, 2017), and projections suggest that rising temperatures in coming decades will be accompanied by continued increases in heat wave frequency and severity (Wuebbles and others, 2014). Similarly, the length of intervals without precipitation has increased over the past several decades (Groisman and Knight, 2008; Diffenbaugh and others, 2017) and is projected to continue increasing in the 21st century, especially in the southern part of the sagebrush biome (Polade and others, 2014). These dry intervals, combined with rising temperatures, will result in longer, hotter droughts in the western United States and Canada (Dai, 2013), including the sagebrush biome (Palmquist and others, 2016b). Simultaneous with increased severity of droughts, the frequency and severity of major precipitation events has been increasing and is projected to continue increasing in coming decades (Pfahl and others, 2017; Prein and others, 2017).

Soil Temperature and Moisture

Sagebrush ecosystems are characterized by a cool-season recharge of soil moisture (Schlaepfer and others, 2012b), so potential changes in winter precipitation as snow (especially when accompanied by rising temperatures) may alter patterns of moisture availability during the growing season. Furthermore, changes in snowpack dynamics are heavily influenced by temperature, so projections are relatively consistent among climate models. In their examination of representative big sagebrush sites, Palmquist and others (2016b; fig. L1) found that an average of 74 percent of precipitation currently falls as rain and that rising temperatures under RCP8.5 are projected to increase that proportion by 8 percent during 2030–2060 (range among climate models: 5–13 percent) and by 16 percent during 2070–2100 (range: 14–18 percent). Average maximum snow-water equivalent at these sites is projected to decrease from 45 mm (1.8 in.) in 1980–2010 to 31 mm (1.2 in.) in

2030–2060 (range: 20–39 mm [0.8–1.5 in.]) and 18 mm (0.7 in.) in 2070–2100 (range: 11–24 mm [0.4–0.9 in.]). These changes alter patterns of soil moisture, leading to increases in the amount of water available to plants during spring and decreases in the amount of water available to plants during summer. This may lead to overall longer warm-season dry soil periods.

Soil temperature and moisture regimes in sagebrush ecosystems are used to assess resilience to disturbance and resistance to nonnative invasive species (Chambers and others, 2014b; Pyke and others, 2015b; Chambers and others, 2016b; Maestas and others, 2016; Chambers and others, 2017a). Recent work (Bradford and others, 2019) characterized the potential impact of climate change on the soil temperature and moisture variables that are the foundation of these assessments. Results suggest substantial increases in soil temperature that are reasonably consistent across climate models. Higher temperatures will expand the area of mesic (ranges from 8 to 15 °C [14.4–27.0 °F]) and thermic (ranges from 15 to 22 °C [27–39.6 °F]) soil temperatures while decreasing the area of cryic (ranges from 0 to 8 °C [0–14.4 °F]) and frigid (<8 °C [<14.4 °F]) temperatures, with the overall effect of decreasing the extent of areas with high resilience and resistance. Simultaneously, shifts toward cool season moisture lead to an increase in the area with cool-season (xeric) moisture conditions and a decrease in the area with warm season (ustic) conditions.

Plant Community Impacts

Single Species Approaches

Much of the research assessing the impact of climate change on sagebrush-dominated plant communities focuses on how precipitation or temperature may affect the distribution or abundance of a focal species (climate suitability models). The two species receiving most of the attention are big sagebrush and cheatgrass. The most common approach is to model current species distributions as a function of climate and other environmental drivers, then project future changes in habitat amount and quality as a function of projected changes in the environment. Studies applying this approach (for example, Schlaepfer and others, 2012a; Still and Richardson, 2015) to big sagebrush estimate declines of the species' occurrence in areas that are relatively low in elevation, warm, and dry (for example, the southern Great Basin and Colorado Plateau). Species' occurrence is estimated to increase in areas that are relatively high in elevation, cool, and wet (for example, montane areas and parts of the northern mixed prairie). Both Schlaepfer and others (2012a) and Still and Richardson (2015) projected substantial decreases in area for sagebrush.

However, similar studies that focused on cheatgrass abundance rather than occurrence found that precipitation seasonality had a greater influence (Bradley, 2010; Boyte and others, 2016; Brummer and others, 2016). Cheatgrass is

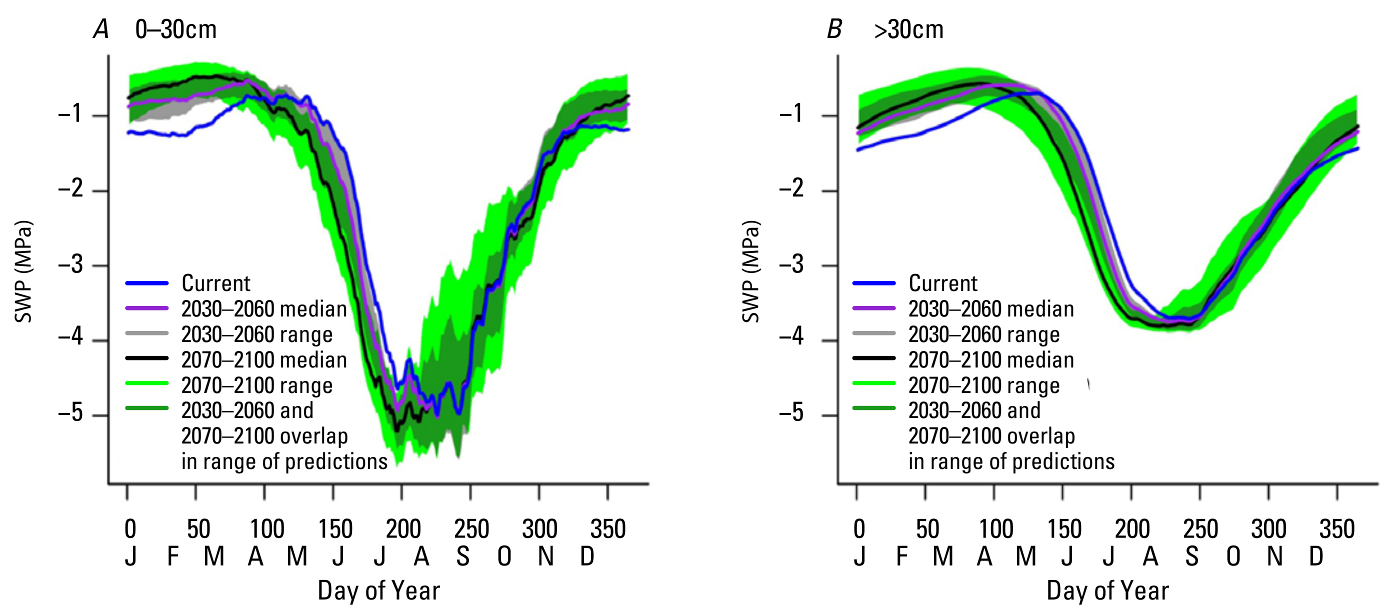

Figure L1. Mean daily soil water potential (SWP), based on 10 global circulation models (GSMs), for *A*, upper (0–30 centimeters [cm; 0–11.8 inches {in.}]) and *B*, lower (>30 cm [11.8 in.]) soil layers for current conditions (1980–2010), 2030–2060, and 2070–2100 across 898 sagebrush (*Artemisia* spp.) sites in the western United States. For 2030–2060 and 2070–2100, daily median values and the daily minimum and maximum values predicted from all 10 GCMs are shown. The overlap in the range of GCM predictions for 2030–2060 and 2070–2100 is in dark green. After Palmquist and others, 2016b. cm, centimeter; >, greater than; MPa, megapascals [pressure]; J, January; F, February; M, March; A, April: M, May; J, June; J, July; A, August; S, September; O, October; N, November; D, December.

currently most abundant in parts of the sagebrush biome with relatively hot and dry summers and where precipitation is received mostly during autumn and spring. The implication is that a change in precipitation seasonality could alter cheatgrass abundance, but predictions about changes in precipitation timing and amount are highly uncertain.

Climate suitability models depend heavily on potentially inaccurate assumptions, such as current distributions being in equilibrium with climate, and rarely provide information about abundance or the dynamics of climate change. Experiments can provide data that allow more direct inference about the effects of specific environmental drivers. Experimental manipulations of temperature and snowpack indicate that cheatgrass fitness likely increases as temperature increases (Concilio and others, 2013; Compagnoni and Adler, 2014a, 2014b; Blumenthal and others, 2016). Experimental manipulations to reduce winter and early spring precipitation limited increases in cheatgrass density (Prevéy and others, 2010a). Increasing winter precipitation through experimental irrigation greatly enhanced big sagebrush abundance over 20 years, provided soils were deep (> 1 meter [3.3 feet]; Germino and Reinhart, 2014).

A multimodel comparison of climate change impacts on sagebrush abundance (Renwick and others, 2018) yielded different inferences than the climate suitability models. Four models estimating the effects of climate change, including time series models (Kleinhesselink and Adler, 2018), mechanistic models (Schlaepfer and others, 2015), and a distribution model generated by Renwick and others (2018), were compared by Renwick and others (2018). The models were built with different data sources and reflected different underlying processes. The outputs consistently projected little change or an increase in sagebrush abundance over much of the species' current range, with decreases projected only in the hottest, driest parts. Both field measurements and modeling also have indicated that sagebrush and cheatgrass have substantial impacts on the microclimatic attributes of sites (Valayamkunnath and others, 2018) such as soil water availability, thereby affecting other plants in the community (Wilcox and others, 2012).

The study of physiological thresholds is another approach for learning about plant responses to climate. For example, the survival of different populations of sagebrush in common gardens is explained best by their adaptation to low temperature (Chaney and others, 2017; Lazarus and others, 2019). These thresholds for freezing damage may help explain patterns of mortality in sagebrush seedlings established from planting stocks after wildfire (Brabec and others, 2017; Lazarus and others, 2019). A response threshold to freezing temperatures also explains differences in the geographic distributions of cheatgrass and red brome (Salo, 2005; Bykova and Sage, 2012).

Impacts to Riparian Systems—Wetland and Meadow

Riparian zones, seeps, springs, and other wetlands make up a small proportion of the sagebrush biome, but they are essential to ecosystem function, the viability of many species of plants and animals, and numerous land uses. For example, about 80 percent of terrestrial animal species in the Great Basin (Thomas and others, 1979), including 66–75 percent of the breeding bird species (Martin and Finch, 1995), are associated with riparian areas for breeding, feeding, or shelter (for example, Dobkin and Wilcox, 1986; Krueper and others, 2003; Earnst and others, 2012).

The extent to which climate change will directly affect the area and configuration of riparian zones and other wetlands is difficult to project. Nevertheless, even if total precipitation changes little, increases in temperature (leading in part to increases in evapotranspiration) and decreases in the proportion of precipitation falling as snow will alter the amount of water availability seasonally and will likely intensify human appropriation of surface water and groundwater (Seager and others, 2007), particularly in the Great Basin part of the biome. Many sources of surface water throughout the Great Basin already are fully appropriated, and water is being reallocated from agricultural to domestic use as exurbanization spreads across the Intermountain West (Brown and others, 2005). Accordingly, the availability of water to support riparian functions, species, and uses is likely to decrease.

In some cases, land use has a stronger effect on riparian species and function than climate does, although the two types of causes interact. For example, recruitment of aspen (*Populus tremuloides*) in the northwestern Great Basin over the past century was much more strongly associated with grazing by domestic livestock than with climate (Beschta and others, 2014). The numerous springs and seeps that are supplied by groundwater, and species and communities in the surrounding areas, also will continue to be affected directly by human uses of water. Groundwater storage has not decreased appreciably over the past century in the Great Basin, and therefore, losses of groundwater are more likely attributable to land use than to climate change (Brutsaert, 2012).

Responses of terrestrial, riparian-associated species to climate change are difficult to project in part because changes in the structure and composition of riparian vegetation have different effects on different species (Strong and Bock, 1990; Dickson and others, 2009). For example, some species respond strongly to the extent of riparian areas, whereas others respond more strongly to the contiguity or fragmentation of riparian areas (Fahrig, 2013). Abundance and recruitment are likely more sensitive than species presence to changes in the amount or fragmentation of riparian cover (Fleishman and others, 2014). Moreover, many riparian areas in the Intermountain West are naturally fragmented. Species that evolved in naturally fragmented systems may have different responses to habitat area and fragmentation than species in human-fragmented systems. As climate changes, the microclimate in some riparian areas may provide a biological buffer from some effects of climate change. For instance, low-elevation ravines are cooler and wetter than surrounding areas and may provide refugia for limber pine (*Pinus flexilis*) in the Great Basin (Millar and others, 2018).

Biological Soil Crusts

Relatively few studies have attempted to assess the long-term impacts of changing climate on competitive interactions within sagebrush-dominated plant communities. One approach to evaluating the potential dynamics of future plant communities, an examination of competition for water by plant functional groups, identified several potential changes in biomass (Palmquist and others, 2018). In particular, biomass of big sagebrush was projected to decline by roughly 30–50 percent in the low-elevation, hotter, and drier areas by 2100, with smaller declines expected in the short term. By contrast, projections suggested that sagebrush biomass may increase by 20–30 percent in high-elevation, cool, and relatively wet locations.

Biological soil crust communities (BSCCs) occur between sparsely distributed woody plants in sagebrush ecosystems and can comprise large parts of the flora cover, particularly where herbaceous vegetation is lacking (Rutherford and others, 2017). The crusts, which are formed by algae, fungi, cyanobacteria, lichens, and bryophytes, occur in semiarid areas. They stabilize soils and increase nutrient cycling, water infiltration, and establishment of vascular plants (Root and others, 2017). With potential changes in climate—and therefore changes in fire regimes and potential invasion by nonnative plants—the species richness, abundance, and cover of BSCCs is likely to change, in turn affecting hydrological and biogeochemical functions (Rutherford and others, 2017). Consequences of a reduction in cover may include soil destabilization, increased albedo (reflection of sunlight), and increased redistribution of dust, all of which could increase rates of snowmelt (for example, Painter and others, 2018; Zhang and others, 2018).

Measurements of BSCCs at four sites in Idaho 12–16 years postfire suggested reductions in percent cover and abundance of several functional groups of plants (for example, squamulose lichens, vagrant lichens, and tall turf mosses), and a 65 percent reduction in species richness (Root and others, 2017). Although the study did not find that fires reduced the overall representation of functional groups of vascular plants, BSCCs require at least one to two decades to recover after fire. With potential changes in climate, and therefore fire regimes and invasion of nonnative species, BSCCs could experience multiple stresses.

Few studies have investigated how BSCCs may change owing to changes in climate. However, their functional importance in semiarid ecosystems is well understood (Ferrenberg and others, 2017), and therefore, manipulations can suggest some of the consequences if their cover, abundance, and composition change. For example, a 10-year study (2005–2015) in the Colorado Plateau established 20 different 5-square meter (m^2; 54 square foot [ft^2]) control sites and treatment sites in which water input and temperature were manipulated to simulate projected climate changes: a 1.2 mm (0.05 in.) increase in summer precipitation and a 2 °C (3.6 °F) temperature increase for 3 years followed by a 4 °C (3.6 °F) temperature increase for 7 years (Rutherford and others, 2017). Treatments were selected to meet climate model projections for 2098 (Christensen and others, 2004). The results indicated as much as a 33 percent increase in albedo in all three treatment types (increased water, increased temperature, and increased water and temperature), which resulted in loss of darkly pigmented, late succession species and increases in cyanobacteria (early successional, lightly pigmented species). Ecosystems and interactions among their biotic and abiotic elements are complex, but increases in the magnitude and rate of warming will likely have negative consequences in many semiarid ecosystems.

Climate Change as One of Multiple Interacting Stressors

The previously referenced studies focused on the direct effects of changes in precipitation or temperature on species and communities but did not address the potential for climate change to interact with—and exacerbate—additional threats to species such as land use change, biological invasions, and changes in fire dynamics. For example, Renwick and others (2018) projected increases in sagebrush abundance in cool, moist parts of the species' range. However, their models did not consider the possibility that warming also might cause an increase in cheatgrass abundance in the same locations, leading to increases in fire and, ultimately, substantial reductions in sagebrush abundance. Large increases in the abundance of cheatgrass and nonnative forbs occurred when sagebrush was experimentally removed from plots (Prevéy, 2010a, b). The effects were exacerbated in study locations where the most precipitation fell during winter (Prevéy and others, 2010a, b), which is projected for much of the core range of big sagebrush (Abatzoglou and Kolden, 2011). Such interactions could amplify, offset, or overwhelm the direct effects of precipitation and temperature on individual species, but little research exists to help understand these potential effects.

Effects of Climate Change on Wildfire

Sagebrush ecosystems are highly variable because they occur over large gradients of climate, topography, soils, vegetation types, and plant functional groups (fig. L2; see also chap. J, this volume). Fire occurrence in any given year is a function of fuels (biomass), the availability of those fuels for burning, fire weather, and ignition sources (Bradstock, 2010). Fire regimes can be altered by changes in the composition of plant functional groups, the amount and availability of biomass for burning (Abatzoglou and Kolden, 2013), and ignitions that are either caused by humans or lightning (Fusco and others, 2015). Invasion of nonnative annual grasses, which are highly flammable and increase fuel continuity, can alter plant functional group composition and increase the amount and availability of fuels following high-precipitation years. Fire size and intensity is strongly influenced by fire weather and fire behavior (Bradstock, 2010). Warmer and drier conditions are often required to decrease fuel moisture sufficiently for large wildfires to burn. Thus, increases in atmospheric carbon dioxide concentrations that result in changes in climate and fire weather (for example, longer and hotter fire seasons and more extreme fire weather) have the potential to influence fire regimes in sagebrush ecosystems (Abatzoglou and Kolden, 2013; Stavros and others, 2014).

Declines in summer precipitation and the number of days with measurable precipitation have likely been a primary driver of increases in area burned across the western United States (Holden and others, 2018). Recent analyses of fire patterns in pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) land-cover types in the semiarid western United States demonstrated that fire seasons started earlier and ended later from 1984 to 2013 in the Sierra Pacific, Central Basin and Range, and Mojave Basin and Range ecoregions (Board and others, 2018). In many of the ecoregions, the area burned during the fire season was related to temperature, precipitation, and soil moisture in the preceding year because of their effects on fine-fuel abundance (Abatzoglou and Kolden, 2011).

Generalized linear models and statistically downscaled climate projections for two representative concentration pathways (RCP4.5 and 8.5) projected significant increases in the probability of very large wildfires during the mid-21st century (2031–2060; >20,234 hectares [ha; 50,000 acres];

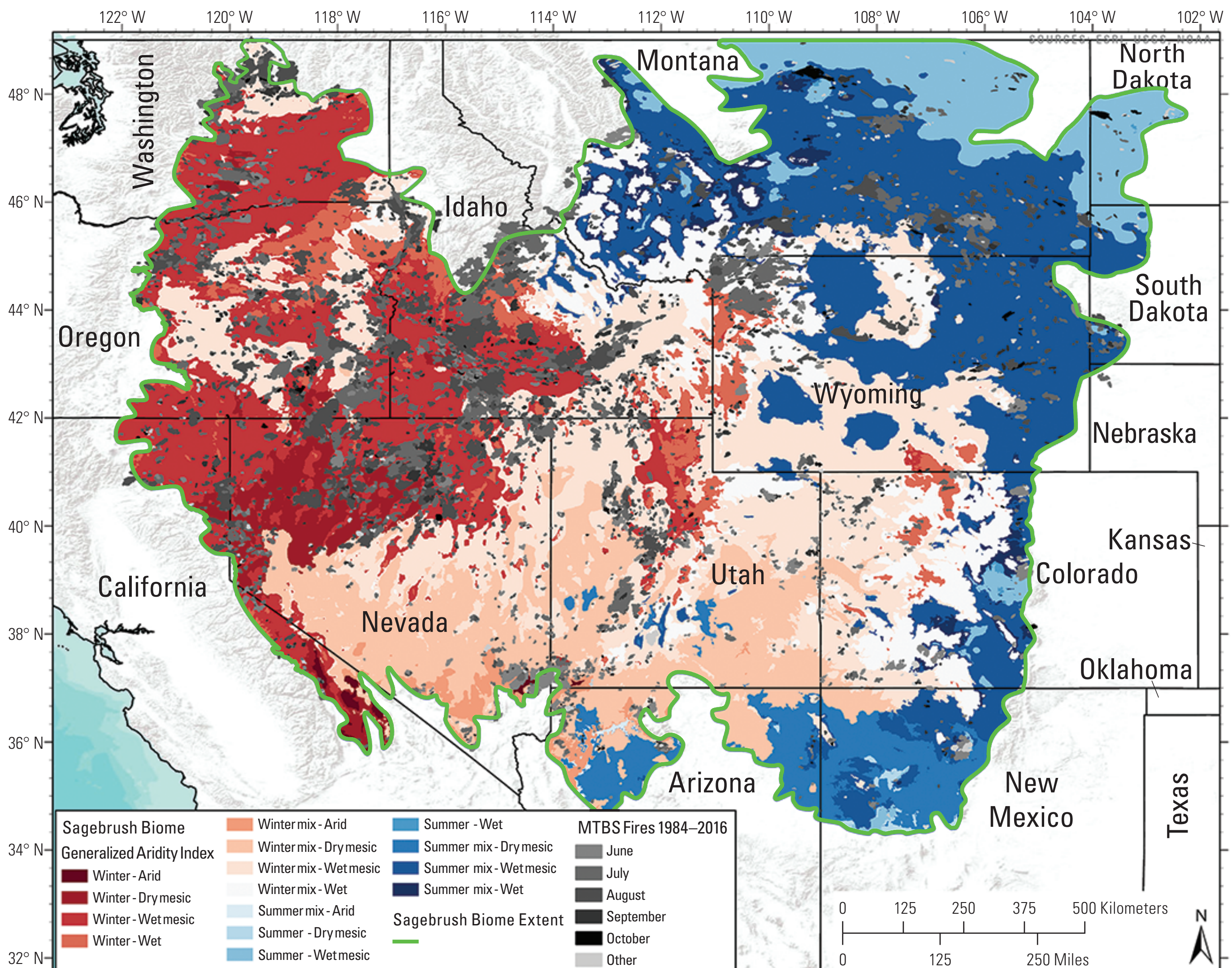

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Figure L2. A generalized aridity index customized for the sagebrush (*Artemisia* spp.) biome (adapted from Dobrowski and others, 2013) based on the timing of precipitation (winter or summer) using 30-year normal annual values (from PRISM Climate Group, 2019), overlaid with the locations of large fires that occurred during 1984–2016 (from Monitoring Trends in Burn Severity, 2018).

Stavros and others, 2014). In mesic areas such as the Pacific Northwest, model agreement was high, and the occurrence of weeks with very large wildfires in a given year was 2–2.7 times more likely. The number of weeks with at least one very large wildfire in fuel-limited systems, such as the western Great Basin, was only 1.3 times greater, but model agreement was low. Therefore, increases in the likelihood of very large wildfires are greater in areas where fire is associated with unusually hot and dry conditions, such as the Pacific Northwest, than in areas where fire is related to conditions in previous years, such as much of the western Great Basin.

Wildlife and Livestock Impacts

Wildlife Impacts and Adaptive Capacity

Conservation planning for climate change, including climate-change vulnerability assessment, has tended to focus on population climate exposure rather than on species sensitivity and adaptive capacity (Butt and others, 2016). Adaptive capacity and land use are likely to have a substantial effect on responses to climate change of native animals in the region in which sagebrush dominates, including but not limited to big sagebrush, black sagebrush (*A. nova*), low sagebrush (*A. arbuscula*), and silver sagebrush (*A. cana*). Species adapt in response to environmental changes (Thomas and others, 1996; Skelly and others, 2007), and these adaptations may be rapid (on the order of years) or slow (on the order of decades; MacDonald and others, 2008; Willis and MacDonald, 2011). Adaptive responses may reflect phenotypic plasticity (the ability of individuals to increase their probability of survival and reproduction by responding to environmental cues), dispersal ability, or adaptive evolution (Reed and others, 2011; Beever and others, 2016). Plasticity is heritable, and therefore can also evolve. Species with relatively high phenotypic plasticity are generally more resilient to environmental change, including climate change, than those with relatively little plasticity (Møller and others, 2008; Willis and others, 2008).

The explicit study of the extent of phenotypic plasticity in wild animals and the extent to which such plasticity is adaptive is rare (Hall and Chalfoun, 2019). An understanding of underlying genetic variation in traits related to persistence as climate changes is even more limited (Culp and others, 2017). The development of new genomic resources, however, may facilitate a better understanding of the adaptive potential of species (Oyler-McCance and others, 2016). Such resources now exist for several species that inhabit sagebrush-dominated areas (Oh and others, 2019). For example, genomic analyses revealed evidence of adaptive variation in genes linked to heat stress, response to viral pathogens, and digestion of plant defense compounds (such as those in sagebrush) in Gunnison sage-grouse (*Centrocercus minimus*; Zimmerman and others, 2019). The extent to which this variation may affect the ability of species to adapt to increasing temperatures or to potential climate-induced changes to its habitat is uncertain. Phenotypic plasticity, however, may be more strongly associated with whether populations persist in the face of climate change than with evolutionary capacity (Dawson and others, 2011).

Many of the animal species that currently inhabit the Intermountain West persisted through relatively rapid and substantial changes in climate and land cover over tens of thousands of years. However, the anticipated rate of widespread climate change from 2010 to 2100 generally exceeds that documented in paleoecological records from the past approximately 2 million years. Therefore, some populations or species, especially those with relatively long generation times, may not be able to evolve genetically with the current pace of climate change (Hoffmann and Sgrò, 2011; Sih and others, 2011). Some species that inhabit open, exposed environments in deserts, including those that occupy relatively low-elevation sagebrush steppe in the United States and Canada, may be among the most vulnerable to changes in climate because they may already be close to their physiological limits (Vale and Brito, 2015).

Changes in climate variability may affect phenology—the timing of seasonal biological events (Parmesan and Yohe, 2003; Gienapp and others, 2013). For example, differences among species in phenological responses to climate variability may affect species interactions including competition, predation, symbiosis, and disease (Yang and Rudolf, 2010). Both plasticity and topographic heterogeneity may reduce the likelihood that asynchronous phenology will reduce the viability of species in the Intermountain West. Additionally, phenological changes may be more likely at relatively high and mesic elevations than at relatively low and xeric elevations (Fleishman and others, 2013).

Livestock Impacts and Adaptive Capacity

Climate-driven stresses on domestic livestock have the potential to reduce the number of young produced or the amount of weight gained (Thornton and others, 2009; Gaughan and Cawdell-Smith, 2015; Rojas-Downing and others, 2017) and therefore to reduce farm or ranch income. This issue is receiving increased attention in both scientific and agricultural communities. Adaptation in this case is largely human-mediated and involves the selection of livestock breeds with traits that are resilient to contemporary and projected climate (for example, heat tolerance; also body size and "muscling"). Adaptation also involves modified management strategies (for example, grazing rotations, stocking rates, protein supplements) that aid in climate response.

Climate change may also impact livestock production by causing an increase in the frequency and severity of droughts and floods, which may reduce available forage and lead to changes in grazing management. Existing programs to help producers manage drought, such as grass banks, drought insurance, more flexible operations (yearlings rather than cow-calf operations), seasonal drought forecasts, and spatial bet-hedging strategies, will become even more important (Finch and others, 2016).

Indirect Climate Impacts

One of the greatest ways in which climate change in arid biomes may affect wildlife and livestock is indirect, from human appropriation of surface water and groundwater. Although per capita municipal water use is declining across much of the western United States, human populations are increasing, and the production of food and energy in the region generally requires considerable inputs of fresh water (Udall, 2013). It is likely that increases in temperature and changes in the timing and amount of snow across the sagebrush biome will reduce water availability for both humans and animals, even if the total amount of precipitation remains fairly constant.

As noted above, climate change interacts with other environmental changes that function as stressors to many species, including changes in land use, species composition, and disturbance processes. Although the scientific community continues to explore whether native species with similar evolutionary histories, life-history traits, and vegetation associations have similar and predictable responses to environmental change, empirical evidence is limited. The greatest good for the greatest number of native species will likely be accomplished by actions that follow first principles of conservation, such as minimizing loss and fragmentation of natural ecosystems by human activities and minimizing the creation of hard edges between vegetation types. In the sagebrush biome, maintaining riparian ecosystems may be especially beneficial to a high proportion of native taxa.

Diseases and Impacts to Wildlife and Humans

As climate and land use continue to change, the distribution, frequency, and virulence of infectious diseases that are either carried by or expressed in native wild animals, domestic animals, and humans across the sagebrush biome are also expected to change. Infectious diseases are the product of interactions among hosts, pathogens, and vectors, and changes in climate may directly affect the distribution, life cycle, and physiological status of hosts (Gallana and others, 2013). However, given the complexity of systems and possible adaptations, there is no consensus on how infectious diseases may respond to climate changes (Liang and Gong, 2017). The physiological changes in hosts may include phenotypic acclimation or genotypic adaptation, but with many interactions and stressors, nonlinear responses of infectious diseases to changing climates are likely (Gallana and others, 2013). Changes in temperature, precipitation, and humidity affect vector abundance and transmission of pathogens. Land use, pollution, and social and economic systems also change in response to climate change, which can affect the geographic and temporal distribution of infectious diseases (Algeo and others, 2014).

In the western United States, fleas and rodents serve as vectors of sylvatic plague (*Yersinia pestis*), which can spread to pets and humans. Prairie dogs (*Cynomys* spp.) are the most common vector in the western United States. Models suggest general reductions of the plague in prairies in the United States but indicate potential shifts of the bacteria to higher latitudes and elevations (Algeo and others, 2014). Chronic wasting disease occurs primarily in the western United States among elk (*Cervus canadensis*), mule deer (*Odocoileus hemionus*), and white-tailed deer (*O. virginianus*). Climate-driven changes in these species' ranges may increase the frequency of their interactions with other ungulates, such as woodland caribou (*Rangifer tarandus caribou*; Algeo and others, 2014).

Hantavirus pulmonary syndrome occurs when humans contact Hantavirus particles associated with feces of murid rodents, such as deer mice (*Peromyscus maniculatus*), which most commonly occurs in the southwestern United States (Algeo and others, 2014). The occurrence of hantavirus pulmonary syndrome fluctuates with population cycles of deer mice, which are responsive to El Niño events. Therefore, climate changes will likely affect distributions and population cycles of deer mice (Algeo and others, 2014) and may increase the occurrence of hantavirus pulmonary syndrome in humans.

West Nile virus (*Flavivirus* spp.), which currently occurs on every continent except Antarctica, causes neurological symptoms in birds (notably greater sage-grouse [Walker and Naugle, 2011]), horses (*Equus caballus*), and humans. Mosquitoes (mainly those of the genus *Culex*) are the primary vectors of West Nile virus. Ticks are a much less common vector (Hoover and Barker, 2016). Temperature and the availability of overwintering sites play a major role in population sizes of mosquitoes. The incidence of West Nile virus has increased significantly since 1996. Given a scenario of RCP4.5 in 2070, West Nile virus is likely to expand across all continents (Hoover and Barker, 2016). Similarly, an assessment of potential risks of West Nile virus in southwestern Wyoming, north-central Montana, and possibly northeastern Wyoming, given six projections of climate in 2030 suggested that transmission is likely to increase in July and August (Schrag and others, 2011).

Climate Change Adaptation

Vulnerability and Adaptation Concepts

Climate vulnerability, the degree to which a system is susceptible to adverse effects of climate change—which may include climate variability and extremes (Intergovernmental Panel on Climate Change, 2007)—can be estimated at a variety of ecological, spatial, and temporal scales with standard vulnerability assessments (Glick and others, 2011). Vulnerability is a function of the sensitivity of a particular system to climate changes, its exposure to those changes, and its capacity to adapt (Intergovernmental Panel on Climate Change, 2007). The potential of natural and human systems to adapt to climate change can be increased by promoting ecological resilience; maintaining ecological function, including ecosystem services; and supporting other elements of biological diversity (Glick and others, 2009). Given the uncertainties associated with projecting future climates and with the adaptive capacity of species and ecological function, some traditional adaptive management approaches are well-suited to guide resource management in response to climate change.

Ecological Models Incorporating Climate

Many modeling approaches aim to characterize historical, current, and future interactions between climate and ecological condition. Climate envelope models are projections of changes in the distributions of individual species (such as sagebrush [Schlaepfer and others, 2012a], cheatgrass [Bradley and others, 2016], or birds [Langham and others, 2015]) under different climate change scenarios. This family of models assume that species-environment relations are spatially homogeneous and permanent (Parra and Monahan, 2008) and, at least implicitly, that climate is the primary driver or limiting factor of species' distributions. Also, these models rarely account for heterogeneity in topography and microclimate that is common across the Intermountain West and which affects the distributions of numerous taxonomic groups (for example, Weiss and others, 1988; Frey and others, 2016). Models that reflect these assumptions can overestimate the distributions of species that are locally adapted (Reed and others, 2011) and underestimate species' capacity for adaptation (Visser, 2008; Chevin and others, 2010; Reed and others, 2013). Furthermore, future values of climate variables may be outside the boundaries of values during the period of observation. Values outlying the boundaries would thereby increase the uncertainty of projections based on associated statistical models.

Climate change velocity models (Carroll and others, 2015; Hamann and others, 2015) evaluate the exposure of an organism to climate change. Climate velocity is calculated by dividing the rate of climate change by the rate of spatial climate variability to hypothesize a speed at which species must migrate over the surface of Earth to maintain constant climate conditions. Forward velocity models measure the distance from a single location (potential source of organisms) to multiple future destinations and focus on species or populations. In other words, these models measure the speed at which an organism would need to move to maintain the same climate niche.

Backward velocity models consider the distance between multiple locations or sources and a single future destination and therefore focus on sites (for example, where source genotypes currently are located [time t] that will be climatically matched with an area of interest at time t+1; Carroll and others, 2015). Velocity modeling approaches are limited by poorly understood relations between climate and species plasticity, and although they explicitly account for variation in local topography, they generally assume distance is a proxy for climate exposure and ignore climate-topographic gradients that may hinder or prevent species movement (Dobrowski and Parks, 2016).

Applying Concepts in the Sagebrush Biome

Coarse-Resolution Approaches

A number of vulnerability assessments have been developed for the sagebrush biome (app. L1; table L1.1). Assessments of climate impacts tend to focus on either specific ecosystem components or questions (such as a single species response, see above) or hypothesize generalized responses to climate change and related drivers of change. The former often are published in the peer-reviewed literature, whereas the latter generally appear in agency reports. The U.S. Department of the Interior (DOI) Bureau of Land Management [BLM] initiated rapid ecoregional assessments (REAs) that covered nearly the full extent of the sagebrush biome. Individual States have also evaluated climate-change threats in State Wildlife Action Plans. For example, Idaho identified species of greatest conservation need; evaluated threats, including those resulting from climate change; and recommended management strategies and actions (Idaho Department of Fish and Game, 2017). An assessment of vegetation responses in the sagebrush biome was provided by Reeves and others (2018a) as part of a set of fairly general vulnerability assessments led by the U.S. Department of Agriculture (USDA) Forest Service (Forest Service; for example, Halofsky and others, 2018a, b).

BLM conducted REAs (https://landscape.blm.gov/geoportal/catalog/REAs/REAs.page) for many of the ecoregions in the conterminous United States where sagebrush is a dominant species. From 2010 to 2015, authors of the REAs collated much of the available digital information on the past or projected effects of change agents (fire, development, nonnative invasive species, and climate) and conservation elements (coarse-resolution elements include major resources or ecosystems, fine-resolution elements were species) to address management questions, such as how a certain conservation element may respond to interactions among certain change agents. The analysis team for each REA convened with land managers and scientists to create a conceptual model of the response of the various conservation elements to change agents and to establish management questions. The management and science team then reviewed each step of the REA process, from data gathering to analysis and reporting. Not all REAs addressed the effects of change agents and adaptation potential in a consistent manner, which precludes applying them collectively to draw inferences across the entire sagebrush biome.

As an example of how climate was evaluated in some REAs, the Central Basin and Range REA provided watershed-level analyses on the overlap among climate responses; the existing distribution of invasive, nonnative grasses; and wildfire risk for several types of sagebrush communities as defined by LANDFIRE (for example, Intermountain Basin Montane Sagebrush Steppe, Intermountain Basins Big Sagebrush Shrubland, and Great Basin xeric mixed sagebrush shrubland; fig. L3).

Managing for Resilience and Resistance

Enabling ecosystem adaptation to climate changes and promoting ecosystem resilience to disturbance are essential for effective management (Chambers and others, 2019a, b). A widely used approach focuses on four types of climate adaptation strategies: resistance, resilience, response, and realignment (Millar and others, 2007; Halofsky and others, 2018a, b; Chambers and others, 2019c; Snyder and others, 2019). Resistance strategies aim to increase the capacity of ecosystems to retain their fundamental structure, processes, and functioning in the face of climate change-related stressors such as longer and hotter drought, more frequent and intense wildfire, outbreaks of insects at frequencies or magnitudes with which most native plants did not evolve, and diseases with which plants and animals did not evolve. Resistance strategies typically are only a short-term solution but often describe the intensive and localized management of rare and isolated species (Heller and Zavaleta, 2009). Resilience strategies aim to minimize the severity of climate change impacts by reducing climate vulnerability and increasing the capacity of ecosystem elements to adapt to climate change and its effects. Response strategies seek to facilitate spatially extensive ecological transitions in response to changing environmental conditions and may include realignment, which is the use of restoration practices to ensure ecosystem function in a changing climate.

Key steps in developing adaptation strategies and actions include obtaining the information on regional climate change projections, resource conditions, and threats; evaluating the relative resilience of ecosystems and high-value resources to climate change and interacting threats; prioritizing areas for management; developing and implementing adaptation strategies and actions; and monitoring the effectiveness of adaptation actions and adjusting management actions as needed (based on Peterson and others, 2011).

The approach used in the Science Framework for Conservation and Restoration (Chambers and others, 2017a; Crist and others, 2019) allows researchers to assess potential effects of climate change and interacting disturbances on sagebrush ecosystems and high-value resources (Chambers and others, 2019b). Geospatial analyses overlay key data to quantify and visualize the locations and extents of high-value species' habitats and resources, such as the probability of occurrence of breeding habitat for greater sage-grouse (*C. urophasianus*). Probable ecosystem response to disturbance and management treatments can be evaluated through a resilience and resistance index that is based on soil temperature and moisture regimes. Dominant threats can be assessed, such as cover of nonnative invasive annual grasses, burn probability, or density of active oil and gas wells. Climate change projections can be used to evaluate future suitability and potential interactions with invasive species and fire. These analyses and overlays can inform land managers' selection of management strategies and target areas for adaptive management.

Recent downscaled climate projections for the sagebrush biome are available (see Chambers and others, 2017a, app. 3). Also, current and future patterns in soil temperature and moisture regimes have been characterized for the sagebrush biome and provide information on how relative resilience to disturbance and management actions and resistance to nonnative invasive annual grasses are likely to change in sagebrush ecosystems (Bradford and others, 2019). Other important data layers are projections of changes in the distributions of individual plant species, such as sagebrush (Schlaepfer and others, 2012a) and annual grasses and forbs (Bradley and others, 2016; Jones, M.O., and others, 2018), under different climate change scenarios.

Climate change projections can be factored into land management prioritizations and strategies (Chambers and others, 2019a). If continued increases in climate change (for example, increases in temperature and shifts in the timing and amount of precipitation) and associated ecological responses are expected to be small, areas can be prioritized to support populations of a given species at ecoregional levels, and management can be used to build local resilience to climate change. If changes in climate are already documented and projected to be large (for example, rapid warming, uncertain snowpack, extreme drought in the next few decades), more proactive strategies may be needed to facilitate ecosystem adjustments.

Restoration

Principles and techniques for restoration of sagebrush ecosystems following fire or other disturbance are discussed in chapter R (this volume); this section provides a discussion of challenges to restoration posed by climate change. Threats such as colonization or expansion of nonnative plants and wildfires most likely will be exacerbated by warming and a higher proportion of precipitation falling in winter. Consequently, active restoration of plant communities to reduce fire occurrence—or to encourage establishment of desirable perennial plant species after fire—will become increasingly necessary. Fuel-reduction treatments and postfire

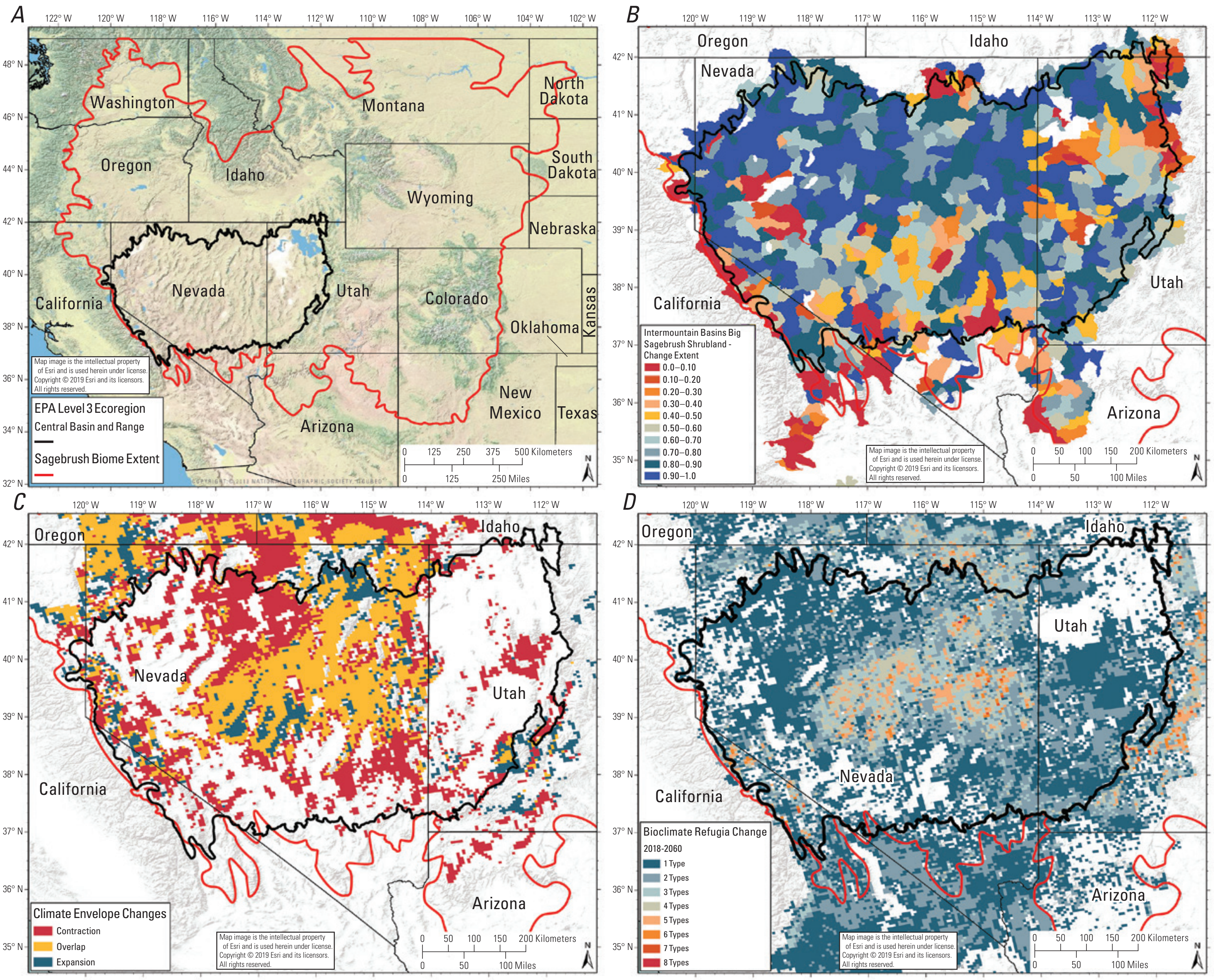

Figure L3. Maps showing *A*, aspects of a changing climate in the Central Basin and Range ecoregion and examples of climate change including *B*, percent change in the extent of Intermountain Basins Big Sagebrush (*Artemisia tridentata*) Shrubland; *C*, locations where the climate envelopes in which species currently occur may be located in the future; and *D*, different bioclimate refugia changes under predicted climate change (Bureau of Land Management, 2019d). EPA, Environmental Protection Agency.

restoration (including rehabilitation) will likely continue to be the largest investments into conservation of sagebrush ecosystems. From 1950 to 2017, more than 9,000 such land treatments were conducted over 3.8 million ha (9.3 million acres) in the Great Basin alone (Pilliod and others, 2017b). Three general considerations regarding climate are key in restoration:

- Climate affects the response of sites to restoration treatments and, conversely, restoration affects the response of sites to climate and the resilience of an ecosystem.
- Restoring perennial species and, potentially, increasing the genetic diversity of seeded or transplanted species may facilitate ecosystem functioning as the environment continues to change (Edwards and others, 2019).
- Consideration of climate during selection of treatments for particular objectives and locations increases the likelihood of success.

The resilience of sagebrush ecosystems or their ability to recover after disturbances, such as wildfire, and their resistance to invasion by nonnative plants is strongly affected by climate, soils, and attributes of the predisturbance plant community (chap. R, this volume, “Resilience and Resistance” sidebar; Chambers and others, 2014a, 2019b). The first consideration for climate adaptation when planning for restoration is prioritization of where treatments are conducted relative to spatial variation in vegetation and long-term climate. A resilience matrix allows land managers to consider both general and spatial resilience when prioritizing areas for management actions (fig. L4; Chambers and others, 2017a). The resilience matrix facilitates estimation of both (1) the locations where conservation and restoration activities are likely to have the greatest benefits and (2) the types of activities most likely to be effective. This decision tool will be most useful when applied in conjunction with an understanding of recent climate changes and projections for the future.

Long-term climate variation or directional changes in temperature, precipitation, and wind exert strong effects on restoration outcomes (Hardegree and others, 2018). Drought or unfavorable timing of precipitation relative to necessary temperatures for growth results in many seeding failures (for example, Brabec and others, 2015). Storm patterns are highly variable among years, and their timing relative to vegetation recovery strongly affects soil stability and restoration (for example, whether sowed seed germinates and transplants survive) via erosion from water or wind (Germino, 2015). Hydrological changes, including the delivery of annual precipitation in fewer but more intense events, are likely to exacerbate erosion and effectively reduce the hydrothermal time required for germination and seedling establishment (Roundy and others, 2018). Treatments such as herbicides, which are most commonly applied before seedlings emerge, are quite sensitive to the timing of application relative to temperature, moisture, and wind, and identifying suitable weather windows can be a considerable challenge.

Weather forecasting tools are increasingly available and can help determine when to apply treatments (chap. R, table R3, this volume). The National Weather Service Climate Prediction Center provides a 3-month outlook of weather and a suite of forecasting tools; the National Weather Service Fire Weather Center announces red flag warnings; the National Interagency Coordination Center provides Significant Wildland Fire Potential Outlooks (7-day and monthly); and a suite of forecasting tools are available on Dr. John Abatzaglou’s website (https://climate.northwestknowledge.net/RangelandForecast/index.php) at the University of Idaho and the Northwest Climate Toolbox (https://climatetoolbox.org/). There are practical limitations to timing postfire restoration treatments to optimize temperature and moisture, such as the fleeting availability of freshly burned and bare soil and emergency fire response funds. Repeat application of treatments such as seeding can be an important means of improving success regardless of weather after seeding. Any restoration treatment should be considered a learning opportunity given the uncertainty of its outcomes, particularly in relatively warm and dry sites (sites with low resilience and resistance) where multiple interventions over many years usually are necessary for success (for example, Shriver and others, 2018). Accordingly, an adaptive management cycle is essential (Wiechman and others, 2019).

Planting a selection of climatically appropriate seed sources, possibly from relatively warmer and drier areas, is a basic climate-adaptation strategy (Richardson and Chaney, 2018). The U.S. National Seed Strategy outlines key needs and steps for avoiding risks of climate maladaptation of seeded or planted species under current or future climate conditions. Given the extensive seedings that occur in sagebrush ecosystems, these concerns are very relevant. Seeds in these ecosystems are either wildland collected (for example, those of sagebrush and some forbs), wildland collected and then farm-reared to increase seed quantity (most forbs and many grasses), or developed from propagated lines and then widely available for use (for example, the Anatone cultivar of bluebunch wheatgrass [*Pseudoroegneria spicata*]).

Seeds of nonnative species also are commonly used in restoration (for example, crested wheatgrass [*Agropyron cristatum*], Lewis flax [*Linum lewisii*], clover [*Trifolium* spp.]). Use of nonnative species sometimes is rationalized based on their low cost and the severity of threat from nonnative grasses. Many of the species used in restoration seed mixes are widespread. They typically have high intraspecific diversity, and therefore it is important to obtain locally adapted subspecies (for example, Mahalovich and McArthur, 2004, for sagebrush). Furthermore, population-level variation may not be associated with subspecies identity but rather with adaptive variation, including local adaptation, which may be underestimated owing to the short duration of many common-garden experiments. This type of experiment occurs when seeds from different populations are planted in the same location to discriminate between genetic and environmental differences (for example, Germino and others, 2019).

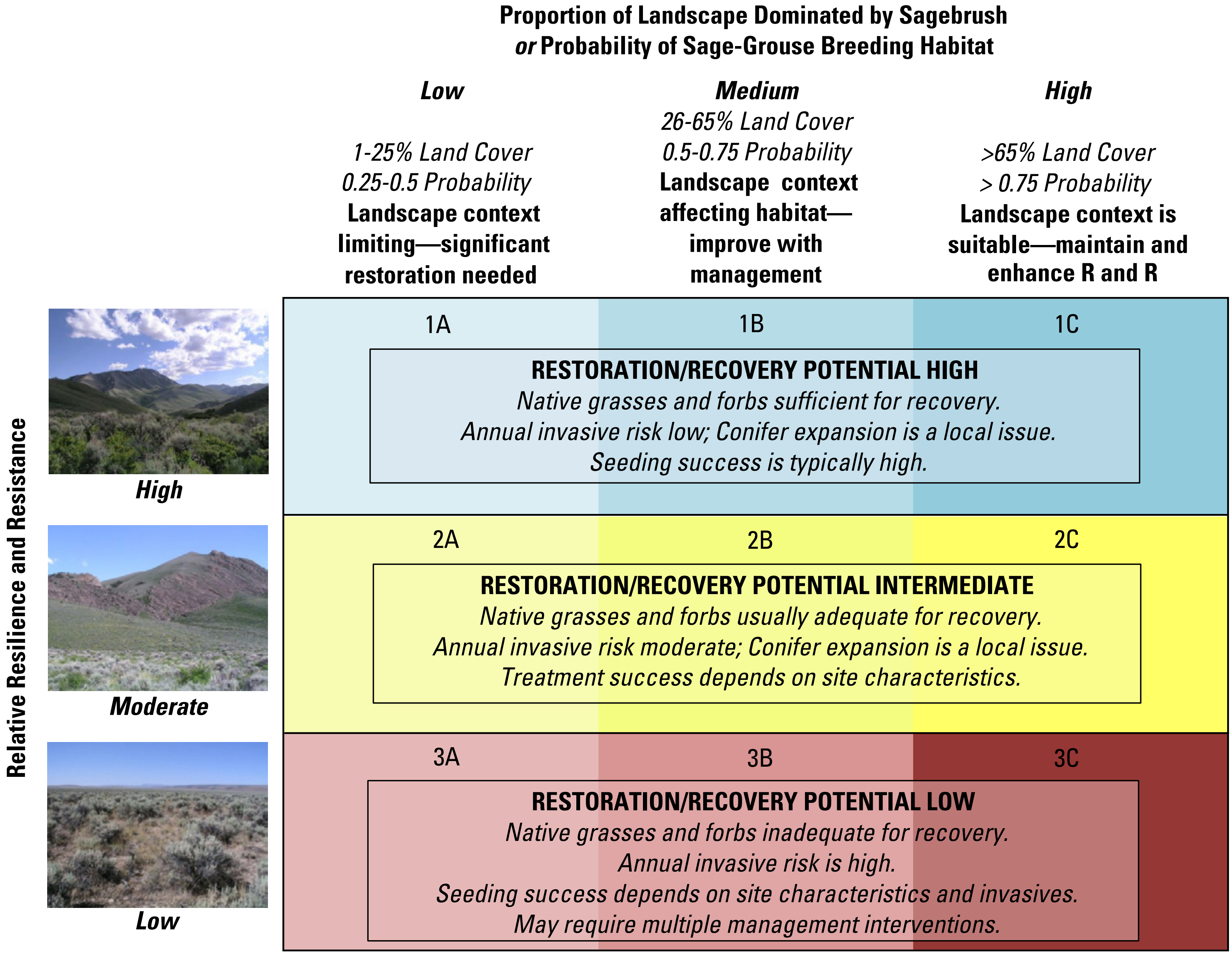

Figure L4. Decision matrix for determining management strategies based on a landscape's resilience to fire and resistance to nonnative invasive annual grasses (rows) and spatial resilience and resources or habitat quality (columns). Adapted from Chambers and others (2017a). %, percent; >, greater than.

Local seed sources are often not an option for restoration of large burned areas, particularly for aerial seeding in the first year or two after fire. Provisional seed zones (Western Wildland Environmental Threat Assessment Center, https://www.fs.fed.us/wwetac/threat-map/TRMSeedZoneMapper.php) have climates similar to those of the burned areas and are useful first approximations for matching the climate of seed sources and planting sites (Bower and others, 2014). Empirical seed zones for a few species have been identified on the basis of common garden studies or genetic information and are the best available guidance for seed selection (Erickson and others, 2004; Johnson and others, 2013). Diversifying seed mixes may be another way to hedge against risks of maladaptation and the uncertainty of future climate. Diversification may be achieved either with multiple populations (seed lots) or propagated lines for a particular species or with multiple species of functional groups of interest (for example, Barr and others, 2017). The Seedlot Selection Tool (https://seedlotselectiontool.org/sst/) is useful for matching sources to planting areas.

Direct manipulation of soil moisture or temperature for restoration, such as with mulching, generally is not feasible for large treatment areas. Biological soil crusts can strongly affect the amount of water available to the soil, and spatially constrained trials have demonstrated that soil crusts can be restored in sagebrush ecosystems (Condon and Pyke, 2016). Efforts to determine whether the techniques can be applied over larger areas are underway. Aggregating seeds into pillows or coating them with hormones or other compounds that influence water absorption and retention can accelerate or delay the seasonal timing of germination (Madsen and others, 2016). Seeding sagebrush into areas among or within restoration projects that have favorable climate resulting from their topography, soils, or biological communities can mitigate climate stresses. For example, north-facing slopes or higher-elevation sites with fertile soils (organic content from prefire shrubs or from the absence of restrictive layers) and limited competition from grasses can result in a greater establishment of sagebrush from seed (Chambers and others, 2017a; Germino and others, 2018). Providing sufficient time for recovery of restored grasses and forbs by restricting grazing by domestic livestock or wild horses (*Equus caballus*) or burros (*E. asinus*) may enable these species to develop the size and root systems that are key for enduring drought.

Current Programs and Activities

Many resource management agencies are transitioning to climate adaptation (Smith and Travis, 2010; Archie and others, 2012; Center for Climate and Energy Solutions, 2012). Under Executive Order 13514 and in coordination with the Interagency Climate Change Adaptation Task Force (ICCATF), all Federal agencies are required to "manage the effects of climate change" (Center for Climate and Energy Solutions, 2012). Prominent Federal agencies that manage lands in the sagebrush biome, including the Forest Service; U.S. Department of the Interior, National Park Service; and U.S. Fish and Wildlife Service, have agency-wide strategic plans for climate adaptation, and the Departments of the Interior and Agriculture have department-level plans. These strategic plans continue to be used for general guidance, referenced for annual policy-level reporting and appear in land use planning documents (for example, see rapid ecoregional assessments, https://landscape.blm.gov/geoportal/catalog/REAs/REAs.page). However, institutional implementation has been slow (Kemp and others, 2015). Federal agency personnel reported that their organizations tend to adapt to climate change through existing management strategies that already are widely implemented (Kemp and others, 2015), in part because managers feel they lack consistent science, guidance, time, and resources to apply emerging adaptation practices. Between 33 and 56 percent of agency personnel surveyed reported that they did not know the degree to which climate change adaptation plans differ from prior management plans (Archie and others, 2012).

Federal resource management staff report actions consistent with these data. When weighed against uncertain future budgets and multiple resource objectives, treatments that cover large areas are often selected over treatments that cover small areas. The latter generally use more expensive, climate-adapted seed mixes. The extent at which treatments occur does not consider landscape climate change effects, but typically considers more localized data such as annual weather variation, antecedent conditions, local slope and aspect, and wild horse or livestock grazing management (that is, timing, season, and duration of use) in the vicinity.

Maintaining and enhancing ecological connectivity may be one of the more effective ways to ameliorate the consequences of climate change on plant and animal populations. Connectivity over extensive areas will be critical in enabling species' ranges to shift in response to climate changes (Heller and Zavaleta, 2009) and to maintaining adaptive capacity via gene flow (Sexton and others, 2011). Research (for example, Buttrick and others, 2015; Crist and others, 2017; Cross and others, 2018) of spatially extensive connectivity and permeability has the potential to inform spatially explicit conservation that maximizes genetic and demographic persistence of sagebrush-associated species.

Each State Wildlife Action Plan (SWAP) revision relevant to the sagebrush biome identifies climate change as a factor for management consideration. Characterization of climate change varies among State plans, from direct threat to pervasive factor, and most SWAPs offer a set of climate adaptation strategies for consideration. Resource management in practice is more likely to be informed by climate adaptation principles than explicitly guided by them. Adaptations, when they occur, typically are integrated with—or modified from—traditional management activities. For example, managers are more likely to be cognizant of changing bird and pollinator behaviors and phenologies than changing climate patterns and, thus, may delay mowing as a result of observing extended nesting by grassland birds. These fine-resolution actions generally are not documented as climate adaptation.

Appendix L1. A Selection of Climate Vulnerability Assessments and Adaptation Strategies Relevant to the Sagebrush Biome

Table L1.1. A selection of climate vulnerability assessments and adaptation strategies relevant to the sagebrush (*Artemisia* spp.) biome.

[-, unspecified]

Title	Year	Geography	Relevant targets
Conservation Assessment of Greater Sage-grouse and Sagebrush Habitats (https://www.fws.gov/greatersagegrouse/documents/Research/WAFWA_Conservation_assessment_2004.pdf)	2004	Badlands and Prairies, Great Basin, Northern Rockies, Southern Rockies, and Colorado Plateau	Sage-grouse, sagebrush
Using the NatureServe Climate Change Vulnerability Index—A Nevada Case Study (https://www.natureserve.org/biodiversity-science/publications/using-natureserve-climate-change-vulnerability-index-nevada-case)	2009	Great Basin	-
Management Planning in Light of Climate Change—Grassland Wildlife in the Great Plains LCC (https://www.cakex.org/sites/default/files/documents/Rowland%20LTA%20rally_10.3.10_GPLCC.pdf)	2010	Badlands and Prairies	Grasslands
Climate Adaptation Priorities for the Western States—Scoping Report (https://www.cakex.org/sites/default/files/documents/WesternGovernorsAssociation.pdf)	2010	Badlands and Prairies, Great Basin, Northern Rockies, Southern Rockies, and Colorado Plateau	All lands
Hydrologic Vulnerability of Sagebrush Steppe Following Pinyon and Juniper Encroachment (https://www.researchgate.net/publication/258498583_Hydrologic_Vulnerability_of_Sagebrush_Steppe_Following_Pinyon_and_Juniper_Encroachment)	2010	Badlands and Prairies, Great Basin, Northern Rockies, Southern Rockies, and Colorado Plateau	Hydrology
Managing Changing Landscapes in the Southwestern United States (https://www.cakex.org/sites/default/files/documents/TNC_Managing_Changing_Landscapes_SW.pdf)	2010	Great Basin; Southern Rockies and Colorado Plateau	Sagebrush species
Bear River Climate Change Adaptation Workshop Summary (https://www.cakex.org/sites/default/files/documents/SWCCI-BearRiver-Climate-Adaptation-Wkshp-FINAL-Report-Nov-2010.pdf)	2010	Great Basin	Wetlands
A Geospatial Assessment on the Distribution, Condition, and Vulnerability of Wyoming's Wetlands (https://www.sciencedirect.com/science/article/pii/S1470160X1000021X)	2010	Northern Rockies	Wetlands
Climate Change Vulnerability Assessments, Lessons Learned from Practical Experience—Practitioner's Responses to Frequently Asked Questions (https://www.cakex.org/sites/default/files/documents/McCarthy%202010%20Climate%20Change%20Vulnerability%20Assessment%20CC%20VA%20Lessons%20Learned_2010_0.pdf)	2010	Southern Rockies and Colorado Plateau	-
Vulnerability Assessment and Strategies for the Sheldon National Wildlife Refuge and Hart Mountain National Antelope Refuge Complex (https://www.fws.gov/refuges/whm/pdfs/SheldonHartNWR_RVA_Report.pdf)	2011	Great Basin	Sagebrush; sage-grouse
Gunnison Basin Climate Change Vulnerability Assessment (http://www.cnhp.colostate.edu/download/documents/2011/Gunnison-CC-Vulnerability-Assessment_and_Appendices-FULL_REPORT-Jan_9_2012.pdf)	2011	Southern Rockies and Colorado Plateau	Sagebrush; Gunnison sage-grouse

Table L1.1. A selection of climate vulnerability assessments and adaptation strategies relevant to the sagebrush (*Artemisia* spp.) biome.—Continued

[-, unspecified]

Title	Year	Geography	Relevant targets
Anticipating Climate Change in Montana's Sagebrush-Steppe and Yellowstone River Systems (https://www.cakex.org/case-studies/anticipating-climate-change-montanas-sagebrush-steppe-and-yellowstone-river-systems)	2012	Badlands and Prairies	Sagebrush steppe
Final Memorandum II-3-C—Northwestern Plains Rapid Ecoregional Assessment (https://landscape.blm.gov/REA_General_Docs/NWP-REA_II-3-C_MainText_App%20A_Final.pdf)	2012	Badlands and Prairies	Shrubland
Vulnerability of Riparian Ecosystems to Elevated CO_2 and Climate Change in Arid and Semiarid Western North America (https://onlinelibrary.wiley.com/doi/full/10.1111/j.1365-2486.2011.02588.x)	2012	Badlands and Prairies, Great Basin, Northern Rockies, Southern Rockies and Colorado Plateau	Riparian
National Fish, Wildlife and Plants Climate Adaptation Strategy (https://toolkit.climate.gov/tool/national-fish-wildlife-and-plants-climate-adaptation-strategy)	2012	Badlands and Prairies, Great Basin, Northern Rockies, Southern Rockies and Colorado Plateau	All lands
A Climate Change Vulnerability Assessment of California's At-Risk Birds (https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0029507)	2012	Great Basin	Sage-grouse, birds
Final Memorandum II-3-C—Middle Rockies Rapid Ecoregional Assessment (https://landscape.blm.gov/REA_General_Docs/MIR_REA-II-3-C_MainReport_andAppxAandB.pdf)	2012	Northern Rockies	Shrubland, steppe, and savanna
Colorado Plateau Rapid Ecoregional Assessment (https://landscape.blm.gov/REA_General_Docs/COP_Final_Report_Body.pdf)	2012	Southern Rockies and Colorado Plateau	Sagebrush
Central Basin and Range Rapid Ecoregional Assessment—Final Report (https://landscape.blm.gov/REA_General_Docs/CBR_1_ReportBody.pdf)	2013	Great Basin	Semidesert shrub and steppe, species
Ecological Assessment Report—Northern Great Basin Rapid Ecoregional Assessment (https://landscape.blm.gov/REA_General_Docs/NGB_REA_Main_Report_and_App_A1.pdf)	2013	Great Basin	Sagebrush, species
Integrating Climate and Biological Data into Land Management Decision Models to Assess Species and Habitat Vulnerability—A Collaboration for Greater Sage-Grouse and their Habitats Final Report (https://www.sciencebase.gov/catalog/item/5761d9c4e4b04f417c2d30f4)	2014	Badlands and Prairies	Sage-grouse
Sierra Nevada Ecosystem Vulnerability Assessment Briefing—Sagebrush (http://ecoadapt.org/data/documents/SierraNevada_Sagebrush_VABriefing_23Oct2014.pdf)	2014	Great Basin	Sagebrush
Assessing the Future Vulnerability of Wyoming's Terrestrial Wildlife Species and Habitats (https://www.nature.org/media/wyoming/wyoming-wildlife-vulnerability-assessment-June-2014.pdf)	2014	Northern Rockies	Sagebrush
Climate, Land Management and Future Wildlife Habitat in the Pacific Northwest (https://cascprojects.org/#/project/4f8c64d2e4b0546c0c397b46/5006e784e4b0abf7ce733f4d)	2015	Great Basin	Sage-grouse
Northwest Regional Climate Hub Assessment of Climate Change Vulnerability and Adaptation and Mitigation Strategies (https://www.cakex.org/sites/default/files/documents/Northwest%20Vulnerability%20Assessment%20Final.pdf)	2015	Great Basin	Rangelands
Assessing the Vulnerability of Vegetation to Future Climate in the North Central U.S. (https://cascprojects.org/#/project/4f83509de4b0e84f60868124/504a01afe4b02b6b9f7bd940)	2016	Badlands and Prairies, Great Basin, Northern Rockies, Southern Rockies, and Colorado Plateau	Vegetation

Table L1.1. A selection of climate vulnerability assessments and adaptation strategies relevant to the sagebrush (*Artemisia* spp.) biome.—Continued

[-, unspecified]

Title	Year	Geography	Relevant targets
Final Project Report—Assessing Climate Change Vulnerability and Adaptation in the Great Basin (https://www.sciencebase.gov/catalog/item/58d2e1cce4b0236b68f84fc0)	2016	Great Basin	-
Mid-Latitude Shrub-Steppe Plant Communities—Climate Change Consequences for Soil Water Resources (https://pubs.er.usgs.gov/publication/70171093)	2016	Great Basin, Northern Rockies; Southern Rockies and Colorado Plateau	Soil water
Changes to Watershed Vulnerability under Future Climates, Fire Regimes, and Population Pressures (https://cascprojects.org/#/project/4f8c64d2e4b0546c0c397b46/531dc54de4b04cb293ee7806)	2016	Great Basin; Northern Rockies, Southern Rockies, and Colorado Plateau	Water resources
Southern California Riparian Habitats—Climate Change Adaptation Actions Summary (https://www.cakex.org/sites/default/files/documents/EcoAdapt_SoCalAdaptationSummary_Riparian_FINAL_small.pdf)	2016	Southern California	Riparian
Upper Snake River Tribes Foundation Climate Change Vulnerability Assessment (https://uppersnakerivertribes.org/app/uploads/files/usrt-climate-assessment.pdf)	2017	Great Basin	Sagebrush, riparian, mule deer, and jackrabbits
Climate Change Vulnerability and Adaptation in South Central Oregon (http://adaptationpartners.org/scoap/docs/SCOAP_GTR_Final.pdf)	2017	Great Basin	Shrubland and grassland
Responding to Ecological Drought in the Intermountain Region (https://www.climatehubs.usda.gov/sites/default/files/r4-droughtfactsheet.pdf)	2017	Great Basin	Rangelands
Wyoming Basin Rapid Ecoregional Assessment (https://landscape.blm.gov/REA_General_Docs/WYB_Report.pdf)	2017	Northern Rockies	Sagebrush steppe, species
Potential Climate Change Impacts on Greater Sage-Grouse Connectivity in the U.S. Northern Rockies (https://www.sciencebase.gov/catalog/item/5867e0d4e4b0cd2dabe7c76a)	2017	Northern Rockies	Sage-grouse
Vulnerability Assessment of Ecological Systems and Species to Climate and Land Use Change within the North Central Climate Change Center and Partner Land Conservation Cooperatives Final Report (https://www.sciencebase.gov/catalog/item/58dd78eee4b02ff32c6859b2)	2017	Northern Rockies	Species
Vulnerability Assessment of Sagebrush Ecosystems: Four Corners and Upper Rio Grande Regions of the Southern Rockies Landscape Conservation Cooperative (https://lccnetwork.org/sites/default/files/Sagebrush%20Vulnerability%20Assessment%20SRLCC_Final.pdf)	2017	Southern Rockies and Colorado Plateau	Sagebrush
Vulnerability of Sagebrush Ecosystem to Climate Change within the Green River Basin (https://www.sciencebase.gov/catalog/item/55b7931de4b09a3b01b5fa0f)	2017	Southern Rockies and Colorado Plateau	Sagebrush
Climate Change and Rocky Mountain Ecosystems (https://www.springer.com/us/book/9783319569277#aboutBook)	2018	Northern Rockies	-
Vulnerability and Adaptation to Climate Change in the Northern Rocky Mountains (http://adaptationpartners.org/nrap/)	2018	Northern Rockies	-
Climate Change Vulnerability and Adaptation in the Intermountain Region—Part 1 (https://www.fs.fed.us/rm/pubs_series/rmrs/gtr/rmrs_gtr375_1.pdf)	2018	Northern Rockies, Great Basin	Sagebrush

Chapter M. Conifer Expansion

By Jeremy D. Maestas,[1] David E. Naugle,[2] Jeanne C. Chambers,[3] Jason D. Tack,[4] Chad S. Boyd,[5] and Joe M. Tague[6]

Executive Summary

Coniferous trees, principally juniper (*Juniperus* spp.) and pinyon pine (*Pinus* spp.), have increased considerably in cover and density in the western United States since European settlement with wide ranging consequences for sagebrush (*Artemisia* spp.) ecosystems. A continuum of vegetation types exists across the region, from conifer-encroached shrublands to persistent pinyon-juniper woodlands and savannas. This chapter focuses on the issue of conifer expansion into sagebrush shrublands and ensuing woodland succession, not the infill of persistent woodlands and savannas. Detrimental effects of conifer expansion on sagebrush ecosystem vegetation composition and productivity, wildlife, water and nutrient cycles, carbon storage, resilience to fire, and resistance to cheatgrass (*Bromus tectorum*) invasion are well-documented. Unprecedented partnerships have formed in recent years to address conifer expansion impacts across ownerships in the sagebrush biome. While significant conifer reduction has occurred in some strategic priority areas, the overall proportion of conifer being reduced through management and wildfire across the region remains relatively small.

Conifer removal is one of the few restoration practices known to be effective for restoring and maintaining a variety of sagebrush ecosystem functions and sagebrush-dependent plant and animal communities, but the degree of efficacy varies by treatment method, pretreatment site type and ecological conditions, location of treatment, follow up treatments, and posttreatment management. Understanding ecological site and stand characteristics is critical when evaluating conifer cover changes and determining appropriate management responses. Carefully crafted management prescriptions across the spectrum of woodland to shrubland—based on ecological site potential and historical stand conditions and dynamics—are needed to address all species habitat requirements at a whole watershed scale in the appropriate places on the landscape. A nuanced and holistic approach is likely necessary to balance multispecies habitat needs across the spectrum of woodland to shrubland.

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[6]U.S. Department of the Interior, Bureau of Land Management.

Introduction

Around the world, native trees are expanding into previously grass- and shrub-dominated systems (Nackley and others, 2017) contributing to the loss of rangelands (fig. M1). In the sagebrush (*Artemisia* spp.) biome, coniferous trees—principally pinyon pine (*Pinus* spp.) and juniper (*Juniperus* spp.; hereafter pinyon-juniper)—have increased considerably in both cover and density and are considered a persistent threat to sagebrush communities in some areas in the sagebrush biome (Chambers and others, 2017a). The focus of this chapter is primarily on factors that have contributed and continue to contribute to conifer expansion into sagebrush shrublands and ensuing woodland succession, along with the impacts of this expansion on ecosystem processes and wildlife. This chapter does not focus on the infill of persistent woodlands and savannas.

Nature and Extent of Conifer Expansion

Pinyon-juniper woodlands occur over an extensive area (greater than [>] 40 million hectares [ha; >100 million acres]) and in a wide variety of environmental conditions across the western United States, but three fundamentally different vegetation types have been described based on canopy structure and understory conditions: (1) persistent woodlands, (2) savannas, and (3) wooded shrublands (fig. M3; Romme and others, 2009). Increases in tree cover and density have resulted in both infill of persistent woodlands, savannas, and wooded shrublands leading to stand closure, as well as tree expansion into sagebrush ecosystems that historically did not support trees. This expansion has resulted in land cover type conversion from shrubland to woodland. Conifer expansion has been especially pronounced in the Great Basin where tree-ring analyses suggest a twofold to sixfold increase in woodlands since European settlement (Miller, R.F., and others, 2008). However, the extent of pinyon-juniper increase varies across the biome and effects are more localized in some ecoregions such as the Colorado Plateau (see reviews in Romme and others, 2009; Miller and others, 2019).

In parts of the sagebrush biome, other conifer species are also expanding locally, such as fir (*Abies* spp.) and ponderosa pine (*Pinus ponderosa*). Ninety percent of tree expansion is estimated to have occurred in sagebrush ecosystems (Miller and others, 2011). Although it is difficult to quantify expansion without site-specific data, remotely-sensed,

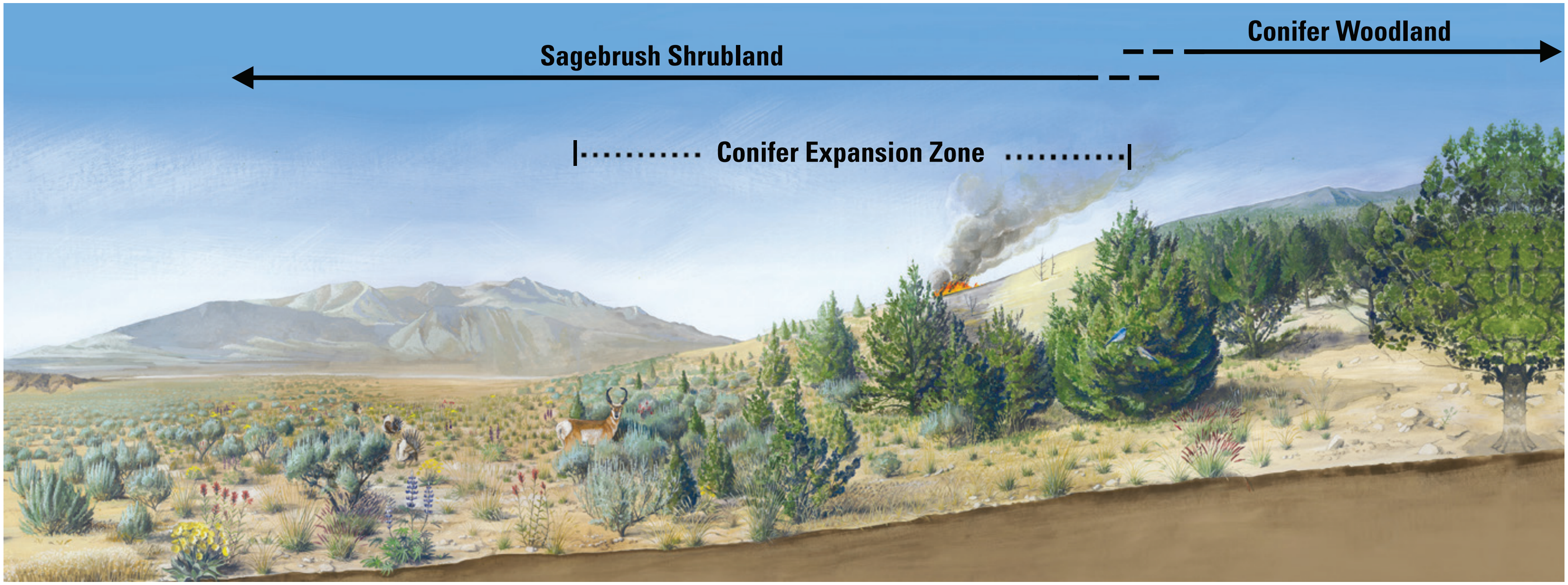

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Figure M1. Illustration depicting the shrubland-to-woodland continuum. Adapted from Johnson and others (2019a, b).

high-resolution mapping of conifer cover within the occupied range of greater sage-grouse (*Centrocercus urophasianus*) found that over one-quarter of mapped lands already support >1 percent tree cover (Falkowski and others, 2017). Newly-available land cover data showing tree cover change in recent decades (1999–2018) further illustrate the patterns and extent of tree cover increase in the sagebrush biome (fig. M2).

Woodland succession progresses in predictable phases that have been described with associated changes in understory vegetation composition (fig. M4): Phase I—trees are present, but shrubs and herbs are the dominant vegetation influencing ecological processes on the site; Phase II—trees are codominant with shrubs and herbaceous vegetation, and all three vegetation layers influence ecological processes; and Phase III—trees are the dominant vegetation on the site and the primary plant layer influencing ecological processes on the site (from Miller and others, 2005). A study in the Great Basin found most sites were still in Phases I and II, but 75 percent of affected shrublands were expected to transition to Phase III woodlands in the next 30–50 years (Miller, R.F., and others, 2008).

Remotely sensed data provides insight into the extent and nature of recent conifer expansion and infill. Across the Great Basin, between 2000 and 2016, the amount of area considered forested by pinyon-juniper (>10 percent tree cover) increased >4,600 square kilometers [1.1 million acres] at an overall rate of 0.46 percent per year (Filippelli and others, 2020). Widespread infilling also occurred over this time period with 80 percent of documented increases in pinyon-juniper aboveground biomass because of infilling of existing woodlands (Filippelli and

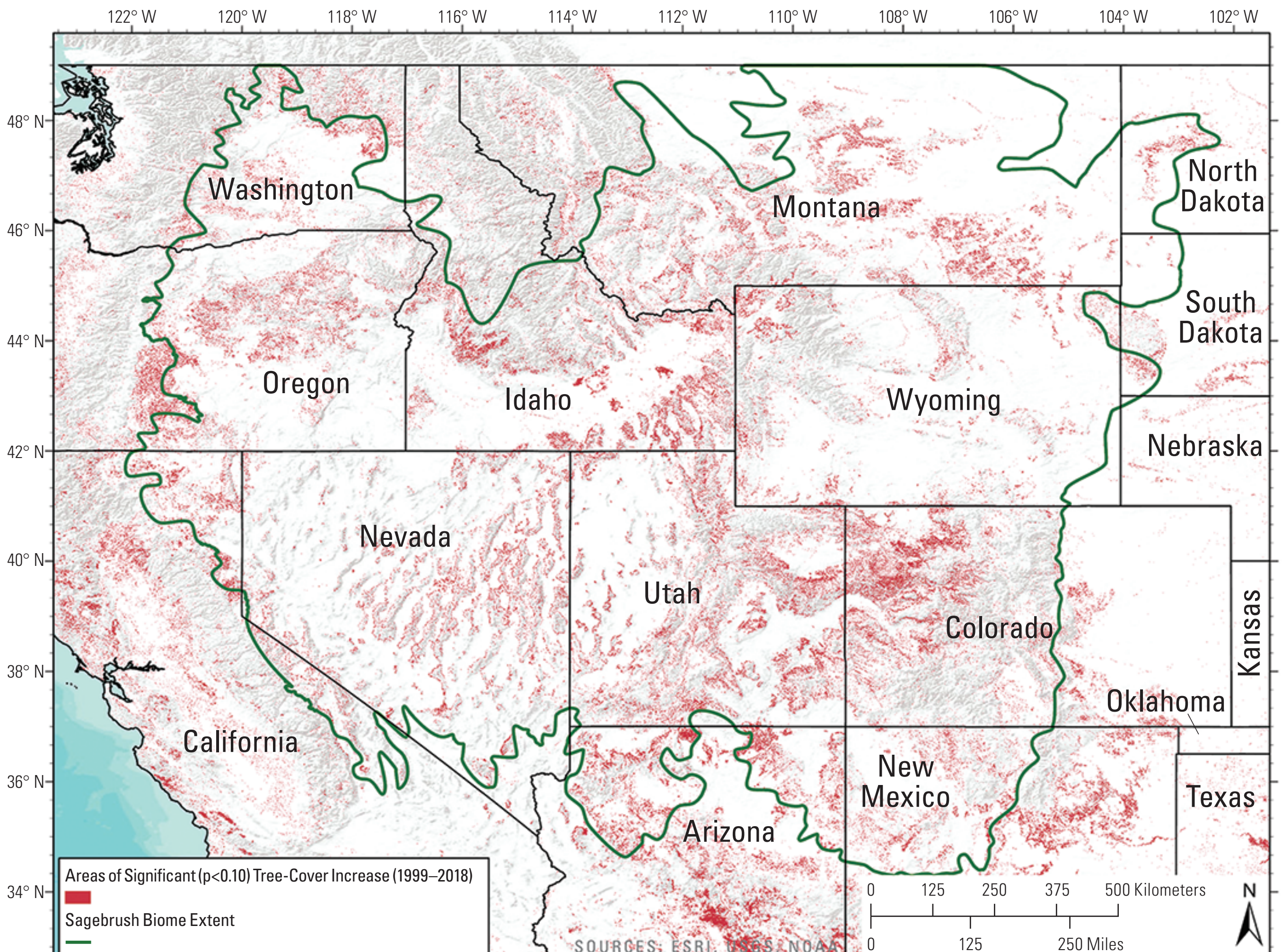

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Figure M2. Rangelands experiencing a significant (p<0.10) increase in tree cover in the western United States (1999–2018). Tree cover includes all tree species, not just conifers. Tree cover from Jones, M.O., and others (2018) assessed for rangelands only, as defined by Reeves and Mitchell (2011). p, probability; <, less than.

others, 2020). Mapping of conifer cover within the occupied range of greater sage-grouse range revealed over one-quarter of mapped lands already support >1 percent tree cover (Falkowski and others, 2017). Newly available land-cover data showing changes to tree cover in recent decades (1999–2018) across rangelands in the sagebrush biome further illustrate the patterns and extent of tree-cover increase (fig. M2; Jones, M.O., and others, 2018).

Causes of recent increases in tree expansion and infill are not fully understood but are often attributed to climate, grazing, and reduced fire occurrence (Miller and others, 2019). However, there is no scientific consensus on the relative importance of these factors (Baker, 2011; Miller and others, 2011). Because a continuum of vegetation types exist across the biome—from sagebrush shrublands to pinyon-juniper woodlands and conifer forests—understanding ecological site and stand characteristics is critical when evaluating conifer cover changes and determining appropriate management responses (Floyd and Romme, 2012; Miller and others, 2014a).

Impact on Sagebrush Communities, Ecosystem Processes, and Wildlife Communities

Where conifer and sagebrush communities interface, an increasing dominance of trees results in the decline of perennial grasses (Tausch and West, 1995; Schaefer and others, 2003; Roundy and others, 2014a), perennial forbs (Bates, 2005; Dhaemers, 2006), and herbaceous productivity and species richness (Miller and others, 2000). Declines occur particularly on warm and dry sites and on sites with shallow, root-restrictive layers in the soil profile (Miller and others, 2005). Increasing woodland cover can affect snow distribution and soil water availability, which in turn shortens the growing season and the duration of water availability (Bates and others, 2000; Roundy and others, 2014a; Kormos and others, 2017). Conversion of shrubland to woodland has also been shown to influence infiltration, runoff, erosion, and sediment loads (Pierson and others, 2007, 2010; Petersen and Stringham 2009; Miller and others, 2013). Susceptibility to erosion

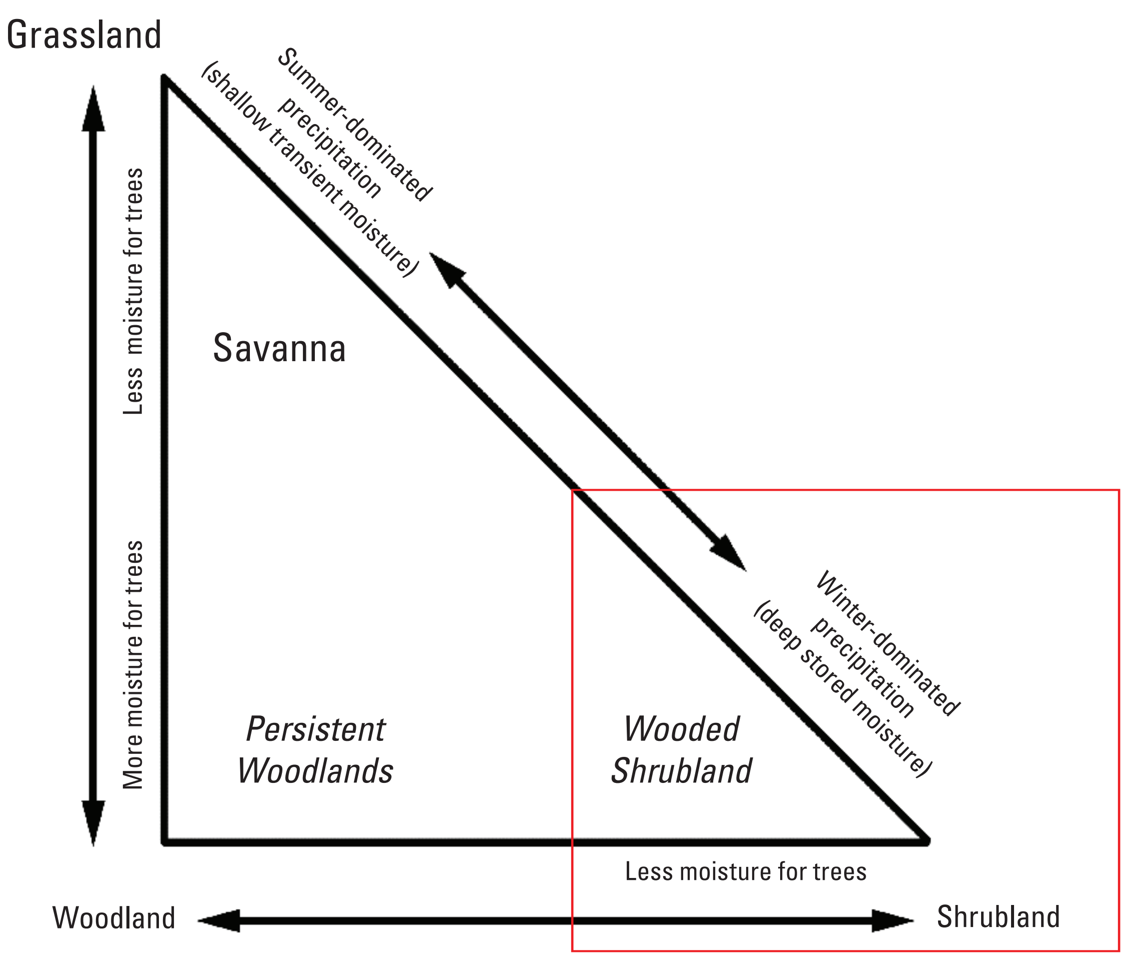

Figure M3. General framework for the pinyon (*Pinus* spp.)-juniper (*Juniperus* spp.) woodland-to-shrubland continuum along a gradient of soil moisture and seasonality of precipitation (adapted from Romme and others, 2009). Expansion of woodlands into former sagebrush (*Artemisia* spp.) shrublands (red box) is the primary focus of this strategy.

Phase I:

- Shrub and herbaceous dominance
- Active tree recruitment
- Low cone production

Phase II:

- Tree, shrub, and herbaceous codominance
- Active tree recruitment
- Cone production moderate to high
- Shrubs intact to thinning

Phase III:

- Tree dominance
- Shrubs >75% absent
- Herbaceous intact (cool-moist sites) to depleted (warm-dry sites) with increased loss on sites with shallow soils
- Limited tree recruitment
- Cone production low to none

Figure M4. Phases of woodland succession and observable field characteristics (adapted from Maestas and others, 2016). >, greater than; %, percent.

following tree increases varies with ecological site potential, as determined by climate, geomorphology, soil erodibility, and ground cover (Davenport and others, 1998).

The carbon cycle changes with increasing tree cover in shrub- and grass-dominated ecosystems. In sagebrush ecosystems, most carbon is stored below ground in the roots (Rau and others, 2011). Conifer expansion into sagebrush ecosystems increases aboveground carbon storage owing to the large increase in biomass, but effects on below-ground carbon storage are poorly understood (Rau and others, 2011). Because there is a larger part of the carbon pool above ground, it is susceptible to volatilization during high-intensity fires (Rau and others, 2009, 2011).

These alterations can reduce sagebrush ecosystem resilience to disturbances and resistance to invasive plants and increase susceptibility to shifts to novel ecosystem states (Chambers and others, 2007, 2014c; Miller and others, 2013). Increases in woody fuel loads because of conifer expansion and infilling heighten the risk of high-severity crown fires, especially during extreme fire weather. High-severity fires can increase the potential for ecosystem conversion to an alternative state dominated by invasive annual grasses (that is, cheatgrass [*Bromus tectorum*] and medusahead rye [*Taeniatherum caput-medusae*]) in warmer and drier areas with insufficient perennial herbaceous species to promote recovery (Miller and others, 2014a; Chambers and others, 2014c). Excessive soil loss on steeper slopes can also result in conversion to an eroded state that is largely irreversible (Chambers and others, 2014c). These state shifts can reduce ecosystem function at landscape scales by fragmenting intact sagebrush shrublands and impairing movements and reproductive processes necessary to sustain plants and wildlife.

Both sagebrush- and woodland-dependent wildlife are affected by increases in conifer cover in the sagebrush biome. Increases in conifer canopy cover result in nonlinear declines in sagebrush cover (Miller and others, 2000; Roundy and others, 2014a), which directly reduces the amount of available food and cover for sagebrush-dependent species. Even before direct habitat loss occurs, sage-grouse avoid or are negatively associated with conifer cover during all life stages (that is, nesting, brood-rearing, and wintering; Doherty and others, 2008, 2010a, 2016; Atamian and others, 2010; Casazza and others, 2011; Dinkins and others, 2014a; Walker and others, 2016; Severson and others, 2017a). Local sage-grouse distribution and demographic rates are impacted with low amounts of conifer present (approximately 1.5–4 percent canopy cover; Baruch-Mordo and others, 2013; Coates and others, 2017a).

No leks remained active when conifer canopy exceeded 4 percent within 1 kilometer (km; 0.6 mile [mi]) of the lek in an Oregon study (Baruch-Mordo and others, 2013). Also, most active leks averaged <1 percent conifer cover within 5 km (3.1 mi) in the western part of the range (Knick and others, 2013). Sage-grouse movement across conifer-expansion areas may be more rapid than across areas without conifer expansion. This may result in lower survival rates among sage-grouse because of potential increased exposure to predators (Prochazka and others, 2017). As a result, the perceived increased risk of predation may cause sage-grouse to avoid habitats with conifer expansion. Higher-elevation sites with early-phase woodland expansion (>2 percent conifer cover) that provide desirable food sources may function as ecological traps, likely because of increased predation from raptors (Coates and others, 2017a).

Phase III encroachment of Douglas-fir (*Pseudotsuga menziesii*) at high-elevation mountain big sagebrush sites in Montana and Idaho reduced availability of forage (big sagebrush [*A. tridentata*] and forbs and grasses), decreased cover quality, and increased predation risk for pygmy rabbits (*Brachyiagus idahoensis*) relative to reference plots (Woods and others, 2013). Shifts in small mammal species composition have been documented, including a decrease in sagebrush specialists (for example, Great Basin pocket mouse [*Perognathus mollipilosus*]) and an increase in woodland specialists (for example, pinyon mouse [*Peromyscus truei*]), in association with increasing conifer woodlands (Rickart and others, 2008).

Long-term trends in Breeding Bird Survey data across the region show similar patterns for songbirds, including decreases in sagebrush species and increases in woodland species, with the exception of pinyon jay (*Gymnorhinus cyanocephalus*; Sauer and others, 2017). Pinyon jay declines may reflect, in part, changes in habitat structure and quality and pinyon pine productivity (Boone and others, 2018) and mortality because of drought (Fair and others, 2018) in persistent pinyon-juniper woodlands and savannas. Big game, such as mule deer (*Odocoileus hemionus*), are likely affected by changes in forage availability and quality that occur as woodland succession advances. Experimental and observational research has shown that nutrition on winter range can limit mule deer survival and population growth (Baker and Hobbs, 1985; Peterson and Messmer, 2007; Bishop and others, 2009). Suppression of forage by trees may explain why the amount of pinyon-juniper in the annual mule deer home range was negatively related to—and explained 26 percent of variation in—ingesta-free body fat in female mule deer in New Mexico (Bender and others, 2007).

Impact on Human Resource Needs and Values

Domestic livestock grazing is the most widespread, contemporary human use of sagebrush ecosystems impacted by woodland expansion. Perennial herbaceous cover is halved when pinyon-juniper cover reaches 40 percent (fig. M5; Roundy and others, 2014a), directly reducing available forage for grazing. In a model of juniper encroachment impacts on ranch economics, McClain (2013) showed that transitioning from Phase I to Phase III reduced available forage, thereby limiting the number of livestock that could be sustained and reducing ranch income by a third. Removing juniper from shrublands increased the livestock carrying capacity by nearly 10 times, which provides added management flexibility as the forage base improves (Bates and others, 2005). Observational studies suggest potential interactive effects among livestock grazing, tree expansion, and livestock forage. In those studies that compared adjacent grazed and historically ungrazed areas, pinyon-juniper densities, canopy cover, or basal area were greater in the grazed than ungrazed pastures (Madany and West, 1983; Guenther and others, 2004; Soulé and others, 2004; Shinneman and Baker, 2009). Impacts of increasing conifer cover on other human resource needs and values, such as use by indigenous peoples and recreation, are likely occurring but not well studied.

Current Efforts to Address Conifer Expansion

Conifer removal has been occurring in the sagebrush biome for decades, but contemporary management differs in its primary objectives, approach, and scale. Early conifer removal efforts (for example, 1950s–70s) were largely done to increase forage production, improve watershed conditions, and enhance deer winter range (Johnson, 1967; Terrel and Spillett, 1975). Because little distinction was made between ecological sites that supported presettlement or newly expanded woodlands, and soil and understory vegetation conditions were seldom evaluated prior to treatment, the response to tree removal was not always positive (O'Rourke and Ogden, 1969; Clary, 1971, 1974). Current primary objectives for most treatments focus on fuels reduction, shrub- and grassland-dependent wildlife habitat, watershed function, and sagebrush ecosystem restoration (Miller and others, 2019). Treatment planning has become more nuanced and now often incorporates ecological site potential, ecosystem resilience to disturbance and resistance to invasive plants, and phase of woodland succession into management decisions (Tausch and others, 2009; Miller and others, 2014a).

In recent years, sage-grouse have become a primary driver of landscape-scale conifer removal in the sagebrush biome with the threat of sage-grouse species being listed under the Endangered Species Act of 1973 (16 U.S.C. 1531 et seq.) and heightened awareness of conifer expansion effects on sagebrush habitats (U.S. Department of the Interior, 2015c; Miller, R.F., and others, 2017). Research indicating sage-grouse sensitivity to very low levels of conifer (approximately 4 percent; Baruch-Mordo and others, 2013) spurred a shift in treatment approach to prioritizing tree removal in expansion areas that are still in early phases of succession (Phases I–II) over areas of dense woodlands (Phase III; fig. M6). Additional research on understory response to woodland succession and treatment further emphasized the benefits of targeting areas of early-phase tree expansion for preserving ecosystem resilience and resistance (Roundy and others, 2014a; Chambers and others, 2014c).

Owing to sage-grouse space requirements and limited restoration resources, clustered treatments across ownership boundaries that expand upon large sage-grouse strongholds have become preferred over small, scattered treatments that

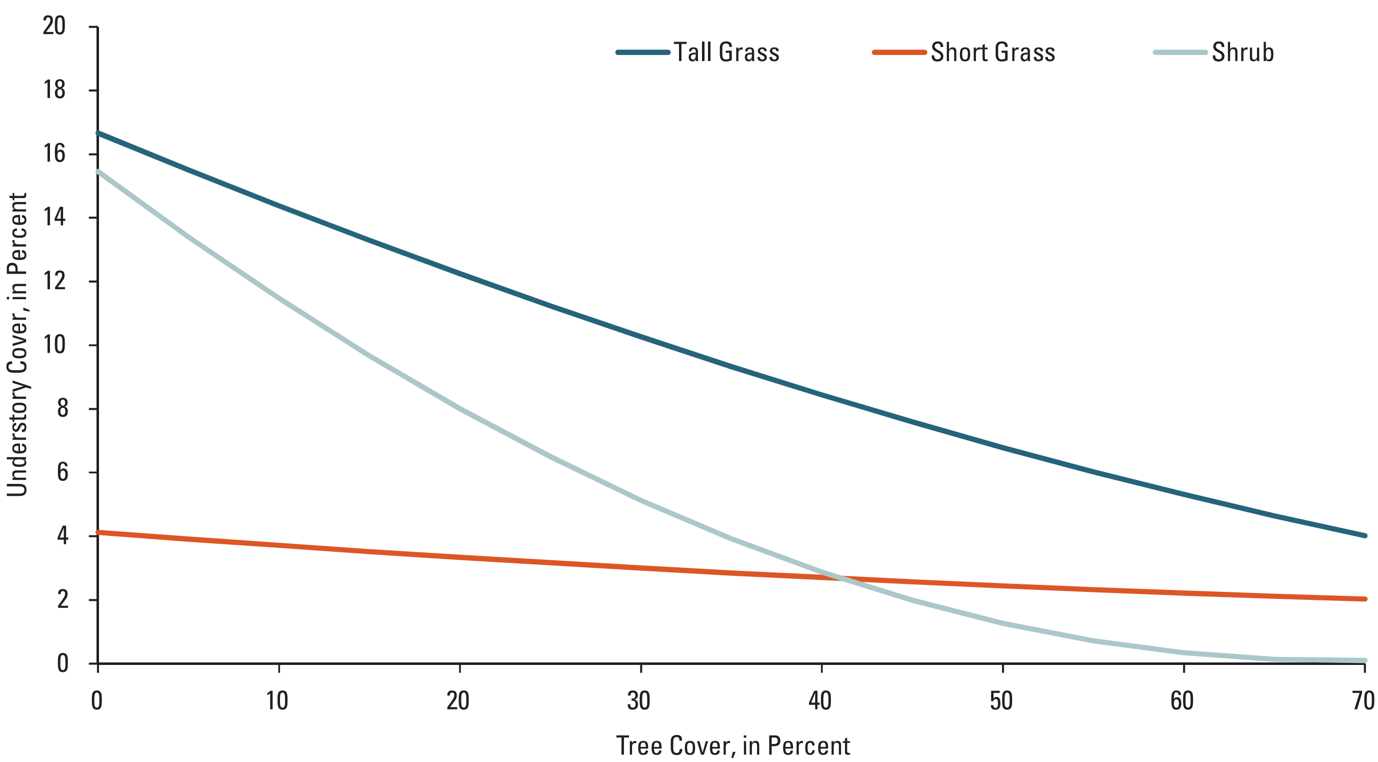

Figure M5. Effects of increasing tree cover on understory cover of shrubs and grasses on 11 sites measured across the Great Basin (Roundy and others, 2014a; adapted from Maestas and others, 2016).

may not yield habitat benefits (Severson and others, 2017b). Use of selective mechanical techniques for tree removal (for example, chainsaw cutting, mastication) has become more prevalent as a means to preserve understory shrubs and herbaceous vegetation and reduce opportunities for invasive annual grasses, although prescribed fire remains an important tool for long-term management under certain conditions (Boyd and others, 2017b).

Extensive efforts to address conifer expansion are ongoing across the sagebrush biome for a variety of land-management objectives. Some larger regional efforts highlighted here provide a glimpse of ongoing management. Since 2010, a diverse coalition of partners have greatly accelerated conifer removal efforts to improve sage-grouse habitat (U.S. Department of the Interior, 2015c). The U.S. Department of Agriculture, Natural Resources Conservation Service launched the Sage Grouse Initiative (SGI) to accelerate voluntary and incentive-based species recovery and proactive ecosystem restoration (Natural Resources Conservation Service, 2015). Private landowners through SGI treated over 250,000 hectares (ha; 617,000 acres) of conifer expansion between 2010 and 2017.

States, other Federal agencies, and private organizations are also involved in woodland management for sage-grouse, fuels reduction, and watershed improvement. For example, from 2005 to 2018, the State of Utah's Watershed Restoration Initiative partnership completed 459,120 hectares (ha; 1,134,472 acres) of pinyon-juniper removal, averaging roughly 35,000 ha (87,000 acres) of treatment per year (fig. M7; Utah Watershed Restoration Initiative, 2019). The U.S. Department of the Interior, Bureau of Land Management (BLM) has also accelerated treatments addressing conifer expansion on public lands mainly because of efforts related to sage-grouse habitat improvement and fire-risk reduction. From 2013 to 2017, the annual rate of conifer removal increased nearly fivefold (2013: 21,606 ha [53,390 acres]; 2017: 101,636 ha [251,137 acres]) for a total of 284,266 ha (702,412 acres) of conifer treated in the sagebrush biome. Conifer treatment continues to be a priority for BLM in the region.

A 2019 study used remote sensing to assess conifer reductions across a large part of the sage-grouse range. The study encompassed >45.7 million ha (>113 million acres) of sagebrush ecosystems over a 4–6-year period coinciding with large-scale treatment efforts (fig. M8; Reinhardt and others, 2020). Of the total estimated area experiencing conifer reduction during this timeframe, 87 percent occurred in three States in the Great Basin: Nevada, Oregon, and Utah. Over half (53 percent) of the conifer reduction occurred in Priority Areas for Conservation (PACs) or in sage-grouse strongholds where managers have been working to maintain large, intact sagebrush-dominated landscapes (U.S. Fish and Wildlife Service, 2013). Mapping confirmed that a diverse array of landowners and managers are contributing to conifer reduction, with the majority of treatments occurring on BLM-administered lands. Accelerated restoration efforts in recent years have raised some concerns about loss of tree cover. However, despite recent concerted efforts, conifer reductions owing to management and wildfire have occurred on just 1.6 percent of the total area supporting trees, providing critical context for local and regional discussions (Reinhardt and others, 2020). In fact, current conifer-reduction efforts may just be keeping pace with the estimated expansion rates of approximately 0.4 to 1.5 percent per year (Sankey and Germino, 2008).

Climate change and wildfire will likely continue affecting conifer expansion in some areas. The total area of pinyon-juniper land cover types that burned in the sagebrush biome increased significantly from 1984 through 2013, except in the Central Basin and Range (Board and others, 2018). More than one-third of the total conifer reduction in studied sage-grouse range was attributed to wildfire (fig. M8; Reinhardt and others, 2020). Climate change may also affect tree expansion as localized areas of die-off have resulted following recent drought in the Central Basin and Range (Greenwood and Weisberg, 2008; Flake, 2016; Flake and Weisberg, 2019) and the Southwest (Fair and others, 2018). Pinyon-juniper woodlands are projected to contract in warmer and drier parts of the biome (Rehfeldt and others, 2012). Conversely, increasing atmospheric carbon dioxide may accelerate growth rates of western juniper (*J. occidentalis*) where soil water and nutrients are not limiting, potentially increasing rate of woodland expansion in some areas (Knapp and others, 2001).

Efficacy of Tree Removal at Restoring Ecosystem Function and Plant and Animal Communities

The efficacy of conifer removal for restoring and maintaining sagebrush ecosystem function, and plant and animal communities, has been well-documented in recent years (see special issues summarized in McIver and others, 2014; Miller, R.H., and others, 2017; Miller and others, 2019). As with all restoration treatments, the degree of efficacy varies depending on treatment method (for example, mechanical, fire), pretreatment site type and ecological conditions, spatial location of treatment, follow up treatments (for example, seeding, weed control), and posttreatment management. However, some generalized outcomes emerging from the literature are summarized in this chapter.

Conifer removal generally results in more herbaceous cover and biomass; twofold to twentyfold increases have been recorded after tree removal (Young and others, 1985; Clary, 1987; Vaitkus and Eddleman, 1987; Rose and Eddleman, 1994; Bates and others, 2000; Stephens and others, 2016; Bates and others, 2017). Perennial herbaceous vegetation is critical to site resilience to disturbance and resistance to invasive annuals (Chambers and others, 2014c; Miller and others, 2014a; Roundy and others, 2014a). However, both perennial herbaceous vegetation and cheatgrass can increase in response to rises in available nutrients and water following conifer removal. Increases in cheatgrass can be

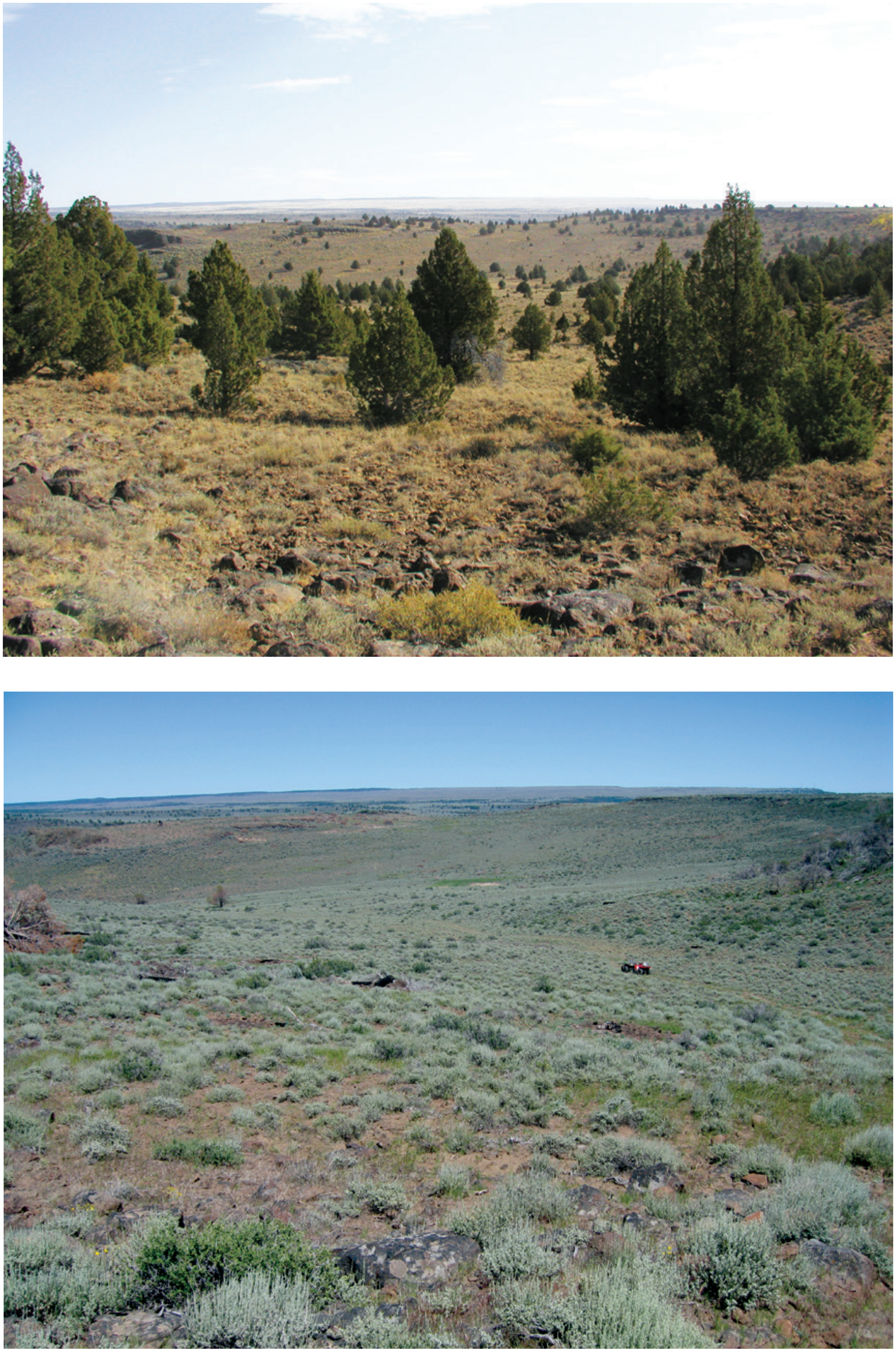

Figure M6. Example of conifer removal in the sagebrush (*Artemisia* spp.) ecosystem in Oregon. The South Warner Project Area before (2008, top) and after (2015, bottom) hand felling of Phase II stands of juniper (*Juniperus* spp.) trees in 2013 (photograph by Todd Forbes, U.S. Department of the Interior, Bureau of Land Management).

especially problematic on relatively warm and dry sites with low initial cover of perennial herbaceous species. In areas susceptible to invasion, treatments should be located where sufficient perennial species occur for site recovery. Alternately, to prevent invasion, land managers should plan follow up treatments such as invasive plant control and seeding (Chambers and others, 2014c; Miller and others, 2014a; Roundy and others, 2014a).

Treatment effects depend on site type; the initial abundance and composition of trees, shrubs, and herbaceous plants; and the type of treatment. Pretreatment tree cover is usually negatively related to pretreatment herbaceous cover, and this relationship influences the potential for posttreatment increases in perennial herbaceous species (Roundy and others, 2014a; Chambers and others, 2014c; Williams and others, 2017). On sites with low to moderate resistance to invasive annual grasses, increases in cheatgrass and other annual exotics are typically greater on sites that have more pretreatment tree cover (Chambers and others, 2014c). On these sites, perennial native herbaceous cover of at least 20 percent appears necessary to prevent a large increase in cheatgrass and other annual invasive plants after treatment (Chambers and others, 2014c). On cooler and moister sites with relatively high resistance, perennial native herbaceous cover may be less important for preventing dominance by annual invaders owing to lower climate suitability (Chambers and others, 2007). However, adequate cover of perennial herbaceous species and root-sprouting shrubs is still necessary for soil stabilization and overall site recovery (Miller and others, 2014a). Sites differ in topography, soil characteristics, and productivity as well as resistance to invaders, and all of these factors should be considered when selecting sites for treatment and evaluating indicators of potential site recovery.

Treatments performed on sites in early stages of woodland succession (Phases I and II) are typically more effective at maintaining the desired sagebrush ecosystem state,

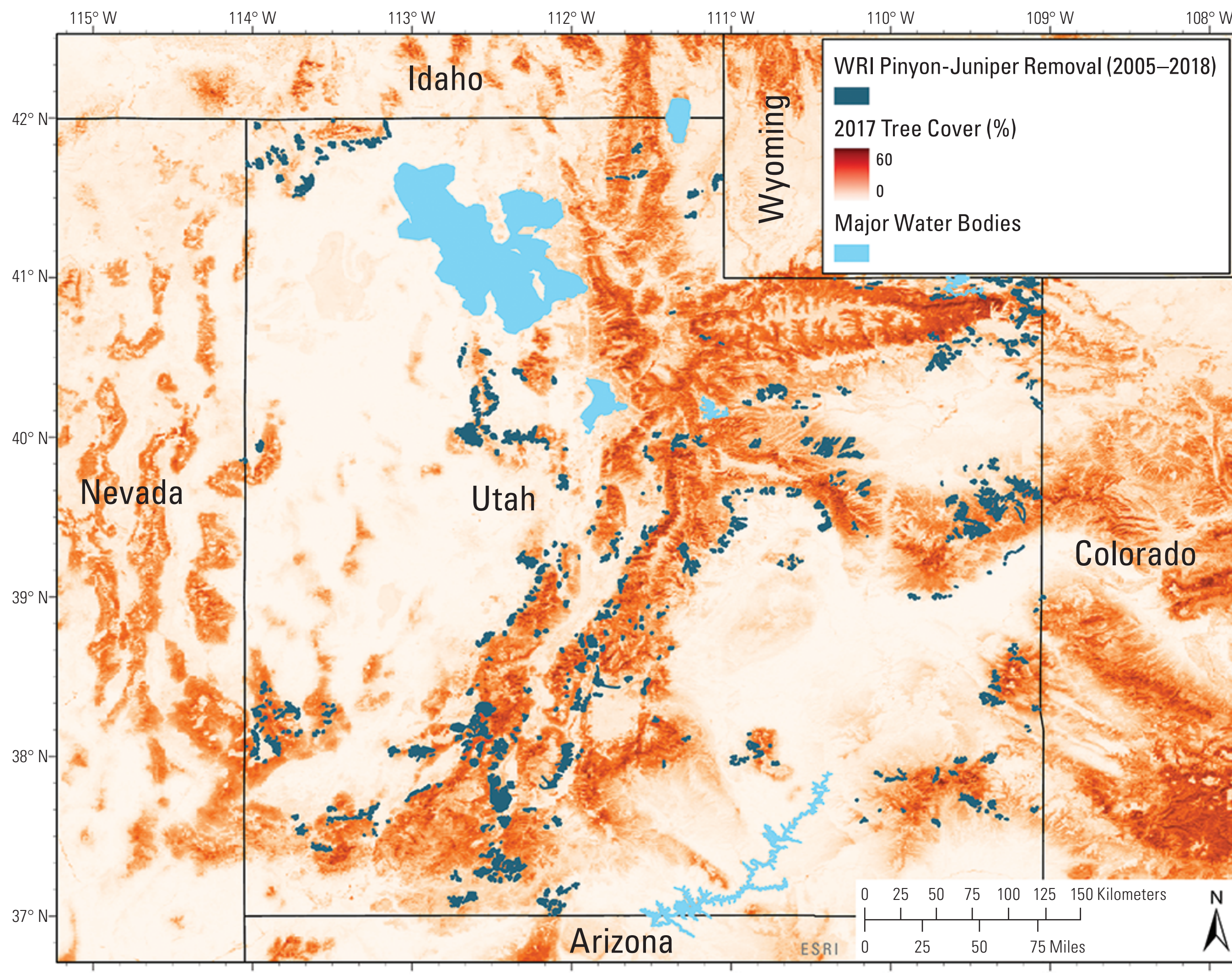

Figure M7. Location of pinyon pine (*Pinus* spp.) and juniper (*Juniperus* spp.) removal conducted through Utah's Watershed Restoration Initiative (WRI) from 2005 to 2018 overlaid on tree cover (from Jones, M.O., and others, 2018). Most pinyon-juniper removal is located in conifer expansion areas along the shrubland-woodland ecotone. %, percent.

but plant composition and structure as well as the treatment method may affect outcomes. Mechanical treatments (for example, cutting, mastication) typically retain most understory shrubs and perennial herbaceous vegetation, whereas prescribed fire often removes nonsprouting shrubs and may result in initial decreases in perennial herbaceous vegetation depending on fire severity (Roundy and others, 2014a; Williams and others, 2017). Higher pretreatment abundance of woody fuels can increase fire severity, which negatively impacts the abundance of perennial grasses and the ability to resist invasive annual grasses posttreatment (Williams and others, 2017). For example, prescribed fire in Phase III plant communities can create a high risk of postburn annual grass invasion (Bates and others, 2013) necessitating follow up weed control and seeding.

On warm and dry sites with lower resilience and resistance, mechanical treatment of Phase I and II conifer expansion is recommended to increase the probability of recovery to a desirable state (Chambers and others, 2014c; Williams and others, 2017). On cool and moist sites with higher resilience and resistance, prescribed fire can be an effective means to restore sagebrush shrublands and provide a longer treatment lifespan before the return of trees, depending on treatment objectives and site conditions (Bates and others, 2017; Boyd and others, 2017b). Regardless of the method used for conifer reduction, posttreatment abundance of deep-rooted perennial bunchgrasses and exotic annual grass response will be important determinants of recovery (Condon and others, 2011).

When strategically targeted and properly designed, conifer removal is one of the few restoration practices with documented efficacy for benefiting sagebrush-dependent birds. In a before-after, control-impact (BACI) study in southern Oregon, nesting habitat suitability for sage-grouse increased after tree removal

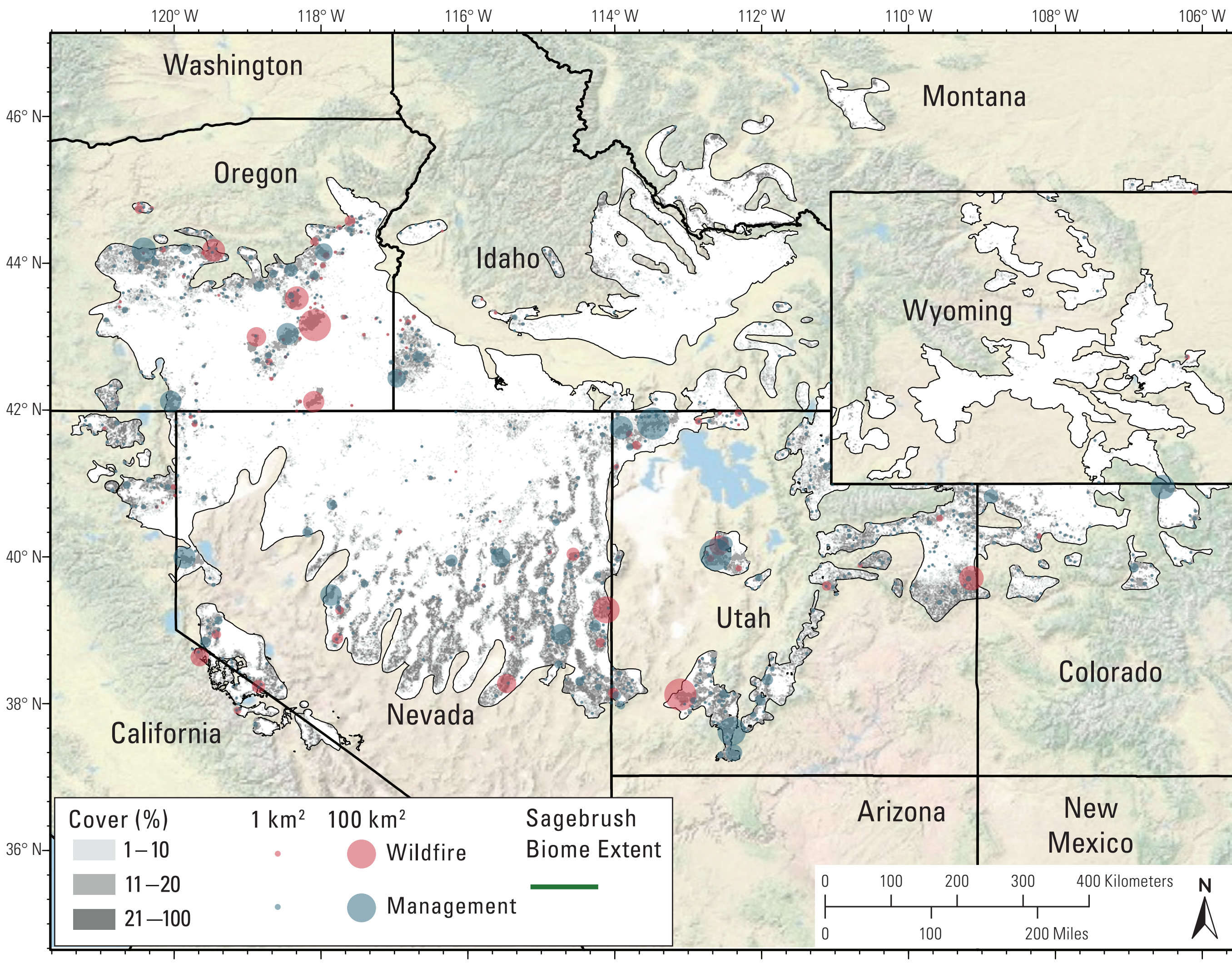

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Figure M8. Locations of predicted conifer reductions owing to management and wildfire in occupied greater sage-grouse (*Centrocercus urophasianus*) range (adapted from Reinhardt and others, 2020). White/grey areas on the image represent mapped tree cover, and the green boundary represents the extent of the sagebrush (*Artemisia* spp.) biome. %, percent; km^2, square kilometer.

(Severson and others, 2017c), and nesting females were quick to use restored habitats (Severson and others, 2017b). Trends in annual female survival (an increase of 6.6 percent) and nest survival (an increase of 18.8 percent) were higher in the juniper-removal treatment area relative to the control landscape (Severson and others, 2017d). Population benefits accrue with time, and integrated population modeling shows that conifer-removal treatments result in an approximately 12 percent increase in population growth rates in treated landscapes relative to untreated control landscapes (fig. M9; Olsen, 2019). In northwest Utah, female sage-grouse using restored habitats were more likely to raise a successful brood (Sandford and others, 2017). Taken together, studies show that conifer removal can increase habitat availability for nesting and brood-rearing sage-grouse with population-level benefits. The potential efficacy of conifer removal for Gunnison sage-grouse (*C. minimus*) was modeled, with results showing large-scale, coordinated conservation efforts to remove conifers, while ensuring treatment areas regenerate into sagebrush-dominated cover, could increase breeding habitat by 46–69 percent (Doherty and others, 2018).

Shrub and grassland songbirds also benefit from treatments. Brewer's sparrow (*Spizella breweri*) increased following tree removal on the Colorado Plateau (Crow and van Riper, 2010). Studies in other locations suggest that removing juniper benefits the species (O'Meara and others, 1981; Noson and others, 2006; Reinkensmeyer and others, 2007; Knick and others, 2014a). In the Great Basin, abundances of Brewer's sparrow, green-tailed towhee (*Pipilo chlorurus*), and vesper sparrow (*Pooecetes gramineus*) more than doubled following mechanical conifer removal (Holmes and others, 2017), and 85 percent of conifer removal conducted through SGI coincided with high abundance centers for Brewer's sparrow (Donnelly and others, 2017). Sagebrush sparrow (*Artemisiospiza nevadensis*) density was either increased or not affected by conifer removal (Reinkensmeyer, 2000; Woolley and Heath, 2006; Reinkensmeyer and others, 2007; Knick and others, 2014a). Though sage thrashers (*Oreoscoptes montanus*) may use small trees, pinyon-juniper encroachment into sagebrush stands does not benefit the birds, and the removal of encroaching junipers improves habitat (Reinkensmeyer, 2000; Noson and others, 2006; Reinkensmeyer and others, 2007). Findings illustrate that conifer removal performed for sage-grouse that retained shrub cover can result in immediate benefits for other sagebrush birds of high conservation concern, but treatment technique and location, ecological site type, and pretreatment understory vegetation matter (Knick and others, 2014a; Miller and others, 2014a).

Benefits of conifer removal have also been recorded for sagebrush mammals of management concern. In an 11-year BACI study, tree removal in Phase II pinyon-juniper areas maintained small mammal densities more effectively than untreated control areas (Hamilton and others, 2019). Several studies have experimentally tested for effects of pinyon-juniper treatments (mechanical or chemical thinning) on mule deer habitat use or demography (Bender and others, 2013; Bergman and others, 2014a, b, 2015). In Colorado, overwinter survival of mule deer fawns was 15 percent higher in mechanically treated areas with follow up seeding and weed control than in untreated control areas (Bergman and others, 2014a). Furthermore, mule deer use of mechanically treated areas was positively related to body size and condition of adult females in New Mexico (Bender and others, 2013) and Colorado (Bergman and others, 2014b), though the significance of these effects was marginal.

Depending upon site conditions and conifer phase, conifer removal can extend the duration of soil water availability by up to 26 days (fig. M10; Roundy and others, 2014b), providing added moisture for plants during critical growth periods. Conifer cutting also extended the period of active growth for understory plants by up to 6 weeks because of greater soil water availability (Bates and others, 2000). At the watershed scale, water delivery can be delayed by an average of 9 days in sagebrush shrublands compared with juniper-dominated systems (Kormos and others, 2017). However, a synthesis across the western United States found that the initial ecohydrologic and erosion impacts of tree reduction on pinyon-juniper woodlands by fire, mechanical tree removal, or drought depend largely on the degree to which perturbations alter vegetation and ground cover structure, initial conditions, and inherent site attributes (Williams and others, 2018). Overall, the literature is inconclusive regarding tree reduction impacts on watershed-scale changes in groundwater and streamflow (Williams and others, 2018). Conifer removal may provide the added ecosystem service of improved water capture, storage, and delayed release in some semiarid ecosystems (which will become increasingly important with warming climate conditions), but results are variable and site specific.

Potential Impact of Conifer Removal on Sagebrush Species

Negative effects of conifer removal are typically associated with woodland-affiliated species and not with sagebrush species. A literature review by Bombaci and Pejchar (2016) found no consistent positive or negative trend in the overall effects of pinyon-juniper woodland reduction on wildlife. However, trends were apparent when analyzed by taxonomic and functional groups and treatment types (Bombaci and Pejchar, 2016). Pinyon-juniper treatments were generally benign or beneficial for sagebrush and shrub-grassland obligate species while generally benign or negative for woodland and woodland-shrubland species. Research was limited in some cases and especially so for nontarget species like invertebrates and reptiles (Bombaci and Pejchar, 2016). Few studies showing positive responses to conifer reduction by sagebrush and shrub-grassland species were found, but the authors noted that most studies were short-term, and species may not respond until several years' posttreatment (Bombaci and Pejchar, 2016).

Detrimental impacts of treatments on sagebrush wildlife species are plausible depending upon factors like technique, location, size, and time since treatment. For example, prescribed burning to control expanding conifers may have negative impacts on sagebrush-obligate species in the short-term if sagebrush cover is lost and trees are not sufficiently removed (Knick and others, 2014a). Conifer removal conducted in areas with poor herbaceous vegetation and invasive annual grasses can result in state shifts to annual grass dominance if follow up weed control and seeding are not implemented. Conifer thinning, as opposed to complete removal, may create ecological traps for sagebrush obligates, like sage-grouse, who might use areas of low conifer cover but suffer lower survival (Coates and others, 2017a). Mechanical conifer removal may also elevate wildfire hazard, and subsequent fire severity, in areas where slash is not properly addressed posttreatment. Some migratory raptors use conifer expansion areas in sagebrush shrublands for nesting habitat. Removal of nest trees can displace individual birds, although the population-level consequences are not well understood.

Habitat structure and composition for both sagebrush and woodland wildlife species have been affected by tree expansion and infill over the last 200 years, and wildlife habitat availability and use have likely been impacted across the woodland-to-shrubland spectrum. Adaptive habitat use may lead to perceived conflicts where restoration of sagebrush ecosystems directly affects wildlife using conifer expansion areas. For example, species that relied on historically open-stand persistent woodlands that have now transitioned to dense forests may have shifted habitat use to tree-encroached sagebrush shrublands.

Big game, such as mule deer, use pinyon-juniper woodlands, savannas, and conifer-invaded shrublands seasonally in some areas (for example, historical winter ranges), and pinyon-juniper has been suggested to provide important thermal and hiding cover. Concern over potential conifer removal impacts on deer habitat use has been raised (Coe and others, 2018), but mule deer use of pinyon-juniper can be a misleading indicator of habitat quality (Bergman and others, 2015; Maestas and others, 2019), and experimental evidence supporting the thermal and hiding cover hypothesis is lacking. In contrast, demographic benefits of pinyon-juniper removal for wintering mule deer have been rigorously tested and documented (Bergman and others, 2014a).

Gray flycatchers (*Empidonax wrightii*) occupy the shrubland/woodland ecotone and have increased significantly in recent years as pinyon-juniper has expanded into sagebrush and other shrublands (Sauer and others, 2017). Not surprisingly, chaining of pinyon-juniper woodlands and hand removal of juniper that had expanded into sagebrush reduced the breeding density of gray flycatchers (O'Meara and others, 1981; Holmes and others, 2017). The pinyon jay is closely associated with pinyon-juniper woodlands and has experienced long-term (1968–2017) population declines. Pinyon jay declines are perplexing given they have occurred during a period of preferred habitat expansion, suggesting changes in woodland extent may

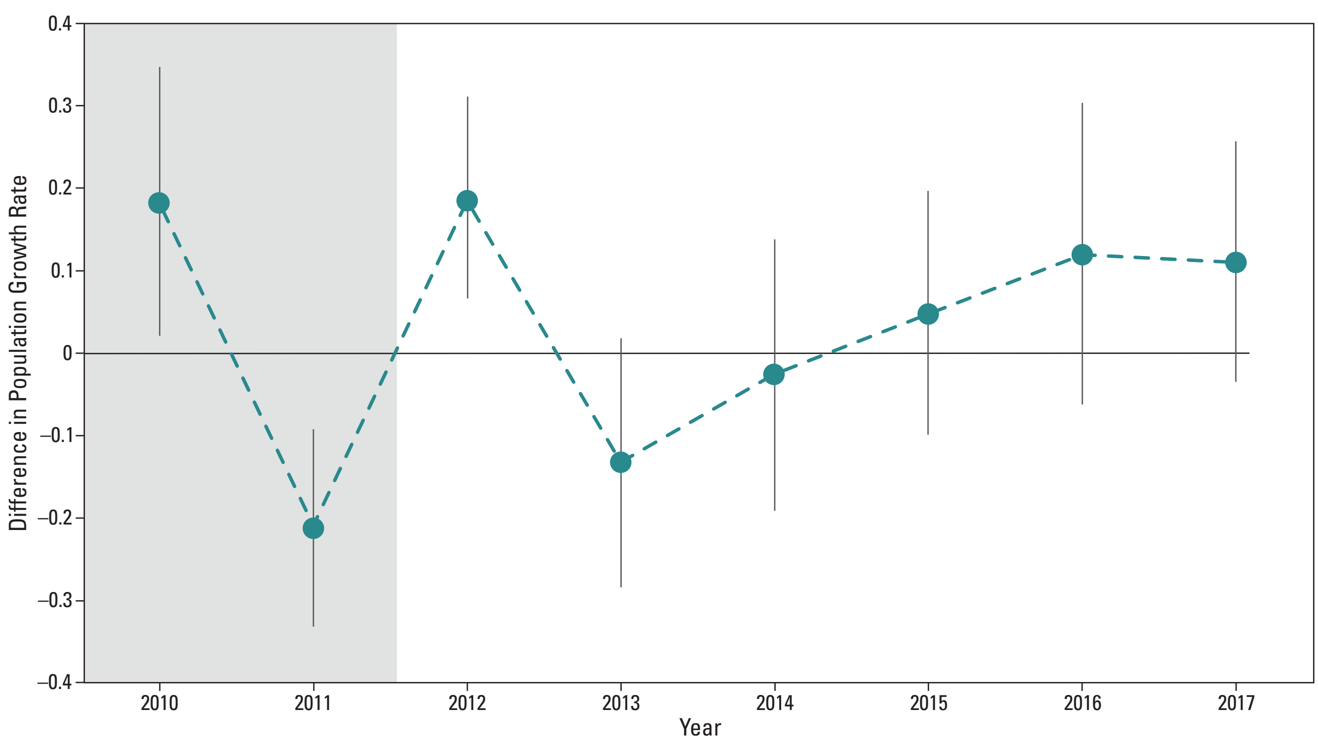

Figure M9. Population growth rates of greater sage-grouse (*Centrocercus urophasianus*) prior to and following landscape-scale conifer removal (adapted from Olsen, 2019). Graph represents the difference in population growth rates in the treatment and control study areas in the Warner Mountains, Oregon. Gray box indicates pretreatment period. Positive growth rates represent years the treatment area performed better than the control (vertical lines are 95-percent credible intervals).

not be the primary driver (Boone and others, 2018). Several alternative hypotheses for these population declines have been put forth including changes in woodland habitat structure and quality (for example, canopy closure), landscape-scale structural changes, pine productivity, and climate change (Boone and others, 2018).

Regardless, the lack of information on pinyon-jay declines has spurred speculation about unintended impacts of conifer management for sagebrush-associated species (Boone and others, 2018; Johnson and others, 2018; Magee and others, 2019). Local decreases in pinyon jay habitat occupancy have been documented in response to fuels treatments in persistent pinyon-juniper woodlands and savannas, but higher occupancy in treatments at landscape scales have been documented (Magee and others, 2019). Pinyon jays have large home ranges (typically 3,500 ha [8,645 acres] to 6,400 ha [15,800 acres]; Marzluff and Balda, 1992; Johnson and others, 2016), and in winter, flocks occasionally move hundreds of miles from their home ranges when food resources are limited (Johnson and Balda, 2020). Pinyon jay habitat use is highly dynamic across years in response to availability of pinyon pine nut production (Somershoe and others, 2020). Thus, the location of treatments along the woodland-to-shrubland spectrum relative to pinyon jay occurrence and breeding colony location likely influences the degree of potential effects. Ultimately, landscape scale studies that identify pinyon jay habitat requirements within and across seasons will allow for improved design of treatments so that sagebush obligates benefit while not impacting pinyon jays.

Conifer removal is often cast as creating winners and losers among wildlife species (Bombaci and Pejchar, 2016), but a more nuanced approach that considers ecological site potential and the full woodland-to-shrubland spectrum (see fig. M1) would allow for holistic multispecies management. Existing wildlife literature evaluating pinyon-juniper management rarely distinguishes between tree removal performed in persistent woodlands and savannas from that in historical shrublands and grasslands, yet ecological sites and associated state-and-transition models are widely used in rangeland ecology as a basis for management decisions (Caudle and others, 2013; Miller and others, 2015). Carefully crafted management prescriptions based on ecological site potential and historical stand conditions and dynamics (Floyd and Romme, 2012) are likely needed to address all species' habitat requirements at a whole watershed scale in the appropriate places on the landscape. Increasing availability of spatially explicit data now allows for optimization in landscape restoration planning that can further balance multispecies needs (Reinhardt and others, 2017; Ricca and others, 2018).

Acknowledgments

This chapter was made possible through staff support provided by the U.S. Department of Agriculture, Natural Resources Conservation Service, and Working Lands for Wildlife partnership. We thank Matthew Jones, Brady Allred, Jason Reinhardt, and Andrew Olsen for their data contributions and figures.

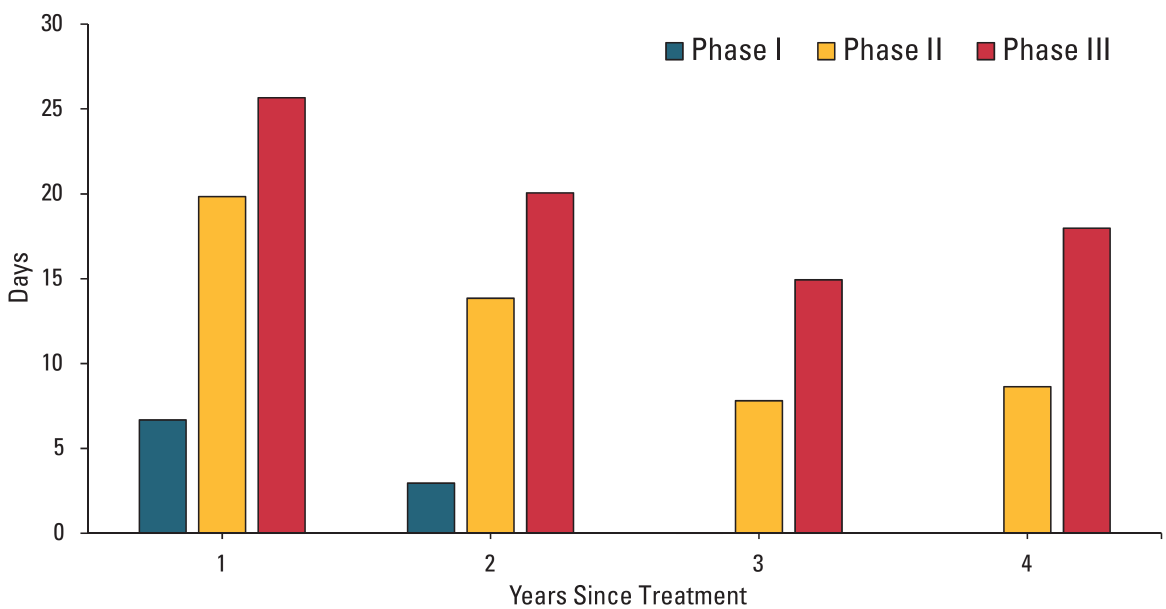

Figure M10. Additional days of soil-water availability following removal of encroaching conifer in Phase I, II, or III of conifer expansion. Tree removal decreases canopy interception of precipitation and tree water use, which results in additional days of soil water availability compared to untreated areas. Additional days of soil-water availability decline with increasing plant cover over time but remain significant for Phase II and III expansion (Roundy and others, 2014b). Figure adapted from Maestas and others (2016).

Chapter N. Free-Roaming Equids

By Terry A. Messmer,[1] San J. Stiver,[2] Mike Cox,[3] and Brian A. Rutledge[4]

Executive Summary

The Wild Free-Roaming Horses and Burros Act of 1971 (16 U.S.C. ch. 30 1331 et seq.) gave the U.S. Department of the Interior, Bureau of Land Management and the U.S. Department of Agriculture, Forest Service the statutory obligation to manage and protect free-roaming equids (that is, feral horses and burros, described as "wild" in the Act) in designated management areas within the sagebrush (*Artemisia* spp.) biome. The intent of the Wild Free-Roaming Horses and Burros Act was to ensure healthy populations of free-roaming equids—defined by this law as wild horses (*Equus caballus*) and burros (*E. asinus*)—on designated Federal lands, in ecological balance with other multiple-uses. The Wild Free-Roaming Horses and Burros Act, as amended by the Public Rangelands Improvement Act of 1978 (43 U.S.C. ch. 37 1901 et seq.) required the Bureau of Land Management to "determine appropriate management level for wild horse and burros on [designated] public lands." As of March 1, 2019, the appropriate management levels for Bureau of Land Management-administered herds was 26,690, with an estimated 88,090 wild horse and burros actually inhabiting designated herd management areas at that time.

Overabundant free-roaming equids are impacting the overall health of western rangelands by degrading ecosystem function and reducing the forage and water available for domestic and native wildlife species. The Wild Free-Roaming Horses and Burros Act identified tools that the Bureau of Land Management and Forest Service can use to manage wild horses and burros—including sale without limitation and euthanasia—both of which are currently restricted. Without active management to reduce growth rates, wild horse and burro populations could more than double in 4 years, exceeding the carrying capacity of the rangelands which they occupy. This will result in increased equid mortality from starvation, and the impact on native wildlife and domestic livestock will be significant. The Bureau of Land Management and Forest Service have retained the ability to gather wild horses and burros in areas where the populations are impacting the rangeland and the health of the animals is compromised. However, if gathered animals are not adopted or sold under applicable legal limitations, the agencies must care for them for the remainder of their lives. In fiscal year 2018, the Bureau of Land Management spent $49.8 million—61 percent of its $81.2 million wild horse and burro program budget—to care for animals in holding facilities.

Introduction

The feral horse (*Equus caballus*; wild horse hereafter), was common in North America through the Pleistocene epoch but went extinct in North America around 10,000 years ago, along with other native megafauna (Webb, 1984; MacFadden 2005; Luís and others, 2006). The horse continued to evolve in Eurasia where it was domesticated about 5,000 years ago (Levine, 1999; Garrott, 2018). Feral burros (*E. asinus*; wild burro hereafter) also evolved in Eurasia (Geigl and others, 2016). Horses and burros were re-introduced to North America by European colonists (Haines, 1938; Dobie, 1952; Bureau of Land Management, 2017). Some of the horses and burros brought back to North America by Spanish explorers escaped or were intentionally released to the wild. These early populations, derived from Spanish bloodlines, were augmented and largely superseded with intentional and unintentional releases of domesticated horses by the military and others through the mid-20th century (Dobie, 1952; Young and Sparks, 2002; Bureau of Land Management, 2017). All current free-roaming equid populations are descendants of horses or burros reintroduced to North America since 1493 (Mitchell, 2015). Ecologically, all free-roaming equids in North America are feral species (The Wildlife Society, 2016).

Legal Status of Horses and Burros on Public Lands

The Wild Free-Roaming Horses and Burros Act (WFRHBA) of 1971 (16 U.S.C. ch. 30 1331 et seq.) gave the U.S. Department of the Interior (DOI), Bureau of Land Management (BLM) and the U.S. Department of Agriculture (USDA), Forest Service (Forest Service) the statutory obligation to manage and protect free-roaming equids (wild horses and burros [WHBs]) in designated management areas (table N1). The intent of the WFRHBA was to ensure healthy populations of free-roaming equids on certain Federal lands, in ecological balance with other multiple uses on designated public

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lands. Section 3 of the WFRHBA states that the Secretary of the Interior shall consult with appropriate State wildlife agencies to facilitate achieving the natural ecological balance of all wildlife species, particularly endangered species. It also prohibits the exploitation or destruction of the animals by private citizens (Norris, 2018).

While all free-roaming equids in the United States may be considered feral (The Wildlife Society, 2016), only the subset designated by the WFRHBA have the legal protection of wild horses and burros. These include descendants of unclaimed, unbranded, free-roaming horses and burros that were present on BLM and Forest Service lands in 1971. The WFRHBA definition of WHBs, does not apply to free-roaming equids that may inhabit National Parks, National Wildlife Refuges, Tribal, State, or private lands. The WFRHBA also identified the tools the BLM and Forest Service could use to manage WHB populations (National Research Council, 2013; Hendrickson, 2018). Currently, the sale of WHBs without limitation and the use of euthanasia (Norris, 2018) are unavailable for use by the BLM or Forest Service because of congressional appropriation riders and litigation (Norris, 2018).

The Taylor Grazing Act of 1934 (43 U.S.C. 315) established a Federal system of grazing allotments across public lands to manage the number of domesticated animals permitted to graze within their geographic boundaries (Banner and others, 2009). Permittees graze their livestock on the allotments that the Federal land agencies lease them through permits that are structured to last 10 years (Bureau of Land Management, 2011). The allotment system evolved as a tool to regulate grazing practices that were damaging public lands (Holechek, 1981; Cawley and Freemuth, 1997; Banner and others, 2009). Grazing of public lands by WHBs is not subject to the Taylor Grazing Act because the WFRHBA defined them as a unique public resource. To address permittee concerns about free-roaming equids competing with their livestock for forage, Congress, through the WFRHBA, legally allocated forage resources for WHBs.

Administrative Structure

The WFRHBA, as amended by the Public Rangelands Improvement Act (PRIA) of 1978 (43 U.S.C. ch. 37 1901 et seq.), required the BLM to "determine appropriate management levels (AMLs) for WHBs on [designated] public lands." These designated public lands are referred to as herd management areas (HMAs) on BLM lands and territories on Forest Service lands. The Forest Service designates wild horse territories (WHT), wild burro territories (WBT), and wild horse and wild burro territories (WHBT; Griffin and others, 2019). The PRIA gave BLM the statutory responsibility for deciding how the HMA AMLs should be achieved along with the agency's multiple-use mandate, which includes wildlife, livestock, wilderness, and recreation considerations (Danvir, 2018; Norris, 2018). Section 3 of the WFRHBA also states that "any adjustments in forage allocations on any such lands shall take into consideration the needs of wildlife species which inhabit such lands."

The AML set by the BLM and Forest Service in 1978 was 26,715 WHBs inhabiting 119,000 square kilometers (km^2; 45,945 square miles [mi^2]) in designated HMAs on public land across 10 western States. The BLM was also given the authority under PRIA to change AMLs to reflect range conditions. As of March 1, 2019, AML for BLM-administered WHB herds, representing the sum of 177 local HMA decisions, was 26,690. As of March 1, 2019, the BLM estimated that there were more than 88,000 WHBs inhabiting designated HMAs, surrounding herd areas (HAs), and other private and public lands (figs. N1 and N2; Bureau of Land Management, 2020). This estimate does not include an estimated 14,000 to 18,000 new foals and does not reflect removals since that date. To achieve AML in 2019, the BLM would need to remove more than 65,000 animals from designated HMAs.

The Forest Service manages approximately 7,100 wild horses and 900 wild burros on 53 territories across approximately 10,117 km^2 (3,906 mi^2) of National Forest System lands in 19 national forests within 5 Forest Service regions and 9 States (https://www.fs.fed.us/wild-horse-burro/index.shtml). Thirty-four of these areas are active (horses or burros present); WHT (27), WBT (4), or WHBTs (3) in Arizona, California, Montana, Nevada, New Mexico, Oregon, and Utah. Twenty Forest Service territories are part of joint management areas that are managed in cooperation with the BLM.

Impacts of Free-Roaming Equids

Estimating the ecological costs of free-roaming equids on western public lands remains problematic because the impacts are dispersed, and there are thousands of additional free-roaming equids that range across private, county, State, and Tribal lands in the West (Beever and others, 2018) beyond those managed by Federal agencies. Overabundant free-roaming equids are impacting the overall health of western rangelands by degrading ecosystem functions and reducing the forage and water available for domestic and native wildlife species (Beever and Aldridge, 2011; The Wildlife Society, 2016; Danvir, 2018; Jakus, 2018; reviewed in Griffin and others, 2019). Free-roaming equids can alter sagebrush-ecosystem processes in a number of ways, including selective consumption of plants, trampling of plants, and compaction of soil which causes increased soil erosion (Dyring, 1990; Beever and Herrick, 2006).

Free-roaming equids also facilitate the spread and establishment of invasive plants such as cheatgrass (*Bromus tectorum*) directly through ingestion, transport, and excretion of viable seeds (King and others, 2019) and indirectly by creating disturbed sites amenable to invasion (Knapp, 1996; Beever and others, 2003, 2008). In the Great Basin, areas without wild horses had higher shrub cover, native plant cover, species richness, overall plant biomass, and lower cover of invasive plant species such as cheatgrass compared to areas with horses (Smith, 1986; Beever and others, 2008; Davies and others, 2014b; Zeigenfuss and others, 2014;

Table N1. Summary of major legislation, policies, and actions regarding wild horse (*Equus caballus*) and burro (*E. asinus*) management by the U.S. Department of the Interior, Bureau of Land Management. (Adapted from National Horse and Burro Rangeland Management Coalition, 2017; Norris, 2018).

Policy	Date	Relevant provisions
Wild Free-Roaming Horses and Burros Act of 1971 (Public Law 92–195)	December 15, 1971	Authorizes and directs the Secretaries of the Interior and Agriculture “to protect and manage wild horses and burros as components of the public lands” that shall be managed in a “manner that is designed to achieve and maintain a thriving natural ecological balance.” In areas found to be overpopulated the destruction of old, sick, or lame animals in the most humane manner possible is permitted and to capture or remove wild horses and burros for private maintenance under humane conditions and care. Limits range of wild horses and burros to areas of public lands where they existed in 1971.
Federal Land Policy and Management Act of 1976 (Public Law 94–579)	October 21, 1976	Directs the Secretary of the Interior to manage Bureau of Land Management (BLM) lands under principles of “multiple use and sustained yield.”
Public Rangelands Improvement Act of 1978 (Public Law 95–514)	October 25, 1978	Directs the Secretaries to “maintain a current inventory of wild horses and burros on given areas of public lands [Herd Management Areas]” to determine “whether and where overpopulation exists.” Directs the Secretaries to “determine appropriate management levels [AML] * * * and determine whether appropriate management levels should be achieved by removal or destruction of excess animals or through other options (such as sterilization or natural controls on population levels).” Directs the Secretaries to destroy “additional excess wild free-roaming horses and burros for which an adoption demand by qualified individuals does not exist * * * in the most humane and cost-efficient manner possible.”
BLM’s Burford Policy	1982	BLM Director Robert Burford places a ban on the destruction of healthy horses.
Interior Appropriations Act Rider	1988–2004	Congress inserts language into the text of Interior Appropriation Bills stating that “appropriations herein made shall not be available for the destruction of healthy, unadopted, wild horses and burros in the care of the Bureau or its contractors.”
Animal Protection Institute of America (APIA) appeals to Interior Board of Land Appeals (109 IBLA 112)	1989–1990	The Interior Board of Land Appeals (IBLA) concludes that under the Wild Free Roaming Horses and Burros Act (WFRHBA) removals must be “properly predicated on a * * * determination that removal is necessary to * * * prevent a deterioration of the range.” Interior Board of Land Appeals interprets AML as “synonymous with restoring the range to a thriving natural ecological balance.” Thus, the number of “excess” animals the Secretary is authorized to remove is that which prevents deterioration of the range—taking into account multiple-use—or that which exceeds a properly established AML.
Fiscal year 2005 Omnibus Appropriations Act (Public Law 108–447)	December 8, 2004	Directs the sale—without limitation—of excess wild horses and burros (or their remains) if “the excess animal is more than 10 years of age; or the excess animal has been offered unsuccessfully for adoption at least 3 times.” Also provides that wild horses and burros or their remains, once sold, are no longer wild horses and burros for purposes of the 1971 Act.
BLM establishes limitations on sale of wild horses and burros	2005–Present	BLM implements internal controls intended to prevent slaughter of sold animals. As part of the sale of any wild horse or burro, buyers must agree not to knowingly sell or transfer ownership of the animals to persons or organizations that intend to resell, trade, or give away animals for processing into commercial products.
Interior Appropriations Act Rider	2010–Present	Congress inserts language into the text of Interior Appropriations prohibiting “the destruction of healthy, unadopted wild horses and burros in the care of the Bureau or its contractors or for the sale of wild horses and burros that results in their destruction for processing into commercial products”
The National Academy of Sciences’ review of BLM wild horse and burro management program	2013	The National Academy of Sciences’ review of BLM wild horse and burro management program Report finds that “continuation of ‘business as usual’ practices will be expensive and unproductive for BLM. Food-limited horse populations would affect forage and water resources for all other animals on shared rangelands and potentially conflict with the multiple-use policy of public rangelands and the legislative mandate to maintain a thriving natural ecological balance.”
National Wild Horse & Burro Advisory Board recommendation	September 2017	“BLM should follow stipulations of the WFRHBA by offering all suitable animals in long and short term holding deemed unadoptable for sale without limitation or humane euthanasia. Those animals deemed unsuitable for sale should then be destroyed in the most humane manner possible.”
BLM adoption incentive	March 2019	Bureau of Land Management sets policy to provide a $500 payment to individuals who adopt an untrained wild horse or burro, with an additional $500 payment 1 year later, when the title for the animal is transferred to the new owner.

Boyd and others, 2017a; Scasta and others, 2018; Griffin and others, 2019). Free-roaming equids in sagebrush landscapes disproportionately use riparian habitats (Crane and others, 1997). Wild burros can have grazing and trampling impacts that are similar to wild horses (Carothers and others, 1976; Hanley and Brady, 1977) and can substantially affect riparian habitats (Tiller, 1997).

Impacts of WHBs on sagebrush ecosystem function and plant composition and structure affect habitat quality for sagebrush-dependent wildlife (Beever and Aldridge, 2011). Horses can also exclude other vertebrates (Hall and others, 2016) and native ungulates such as pronghorn (*Antilocapra americana*; Berger, 1985; Gooch and others, 2017), bighorn sheep (*Ovis canadensis*; Ostermann-Kelm and others, 2008), and elk (*Cervus canadensis*; Perry and others, 2015) from water sources.

Free-roaming equid HMAs overlap the range of several at-risk sagebrush wildlife species, most notably the greater sage-grouse (*Centrocercus urophasianus*). In 2011, approximately 12 percent of the current range of greater sage-grouse was managed for free-roaming equids (Beever and Aldridge, 2011). Approximately 60 percent of an estimated 67,027 wild horses and burros (39,285 animals) occurred within 52,610 km^2 (20,312 mi^2) of greater sage-grouse habitat. On lands administered by the Forest Service, an estimated 3,400 free-roaming equids occur within about 1,800 km^2 (695 mi^2) of greater sage-grouse general and priority habitat. In addition, an estimated 650 wild horses occur within Bi-State sage-grouse habitat in California and Nevada, on about 283 km^2 (109 mi^2) administered by the Forest Service and 333 km^2 (129 mi^2) administered by the BLM. Wild burros

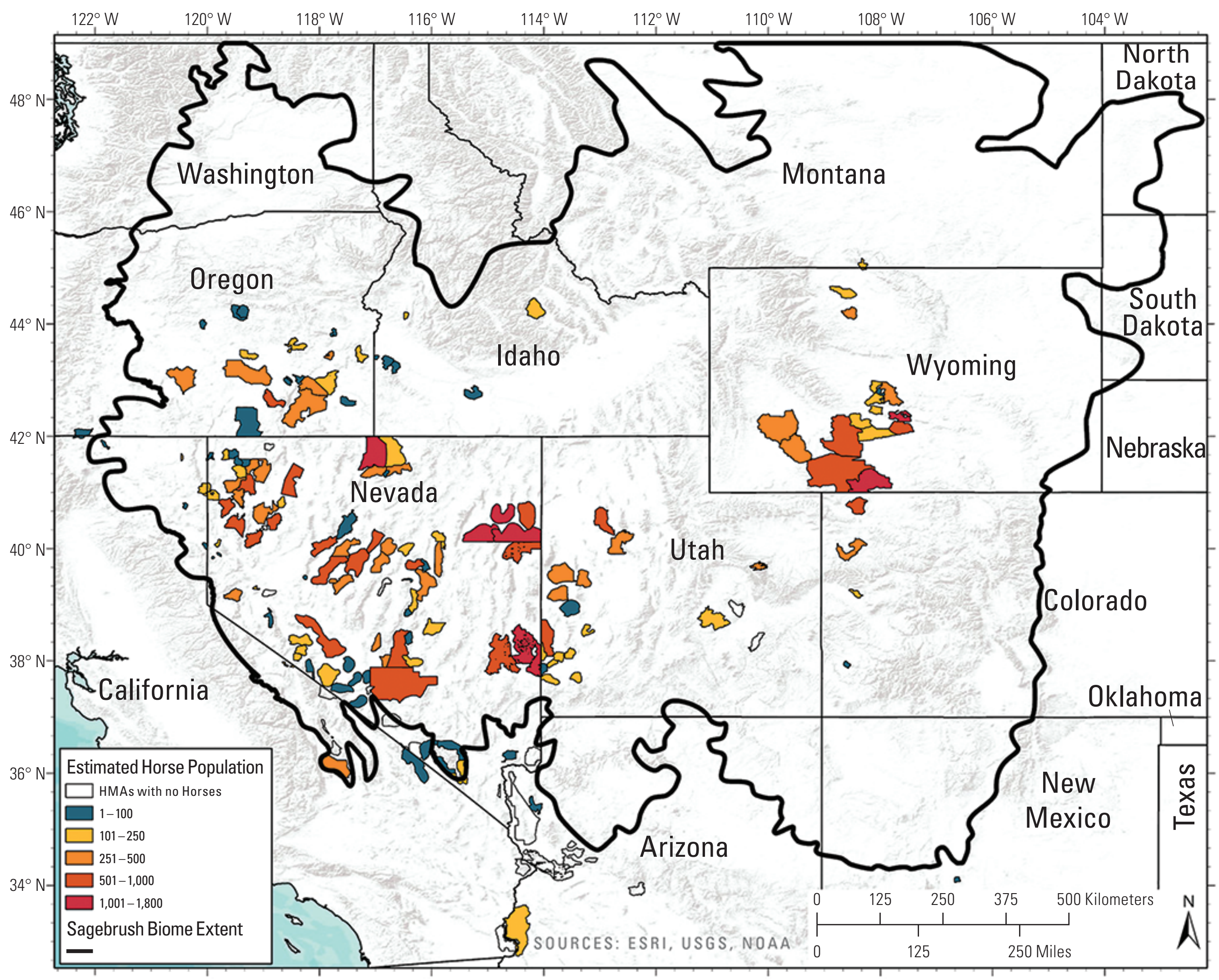

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Figure N1. Estimated populations of wild horses (*Equus caballus*) administered by the U.S. Department of the Interior, Bureau of Land Management by designated herd management areas (HMAs) in the sagebrush (*Artemisia* spp.) biome, March 2019 (Bureau of Land Management, 2019d).

are not nearly as numerous as wild horses in the sagebrush biome. However, the tendency of burros to use low-elevation habitats throughout the year may lead to a high degree of overlap between burros and sage-grouse habitat, where burros and greater sage-grouse co-occur (Beever and Aldridge, 2011).

More WHBs on public lands means increased demands will be placed on rangeland resources, particularly during periods of drought. When public-land resources become depleted, stressed free-roaming equids will search out new sources of forage and water (Hennig and others, 2018). With these movements, they may cross public highways more frequently (increasing motorist safety risks) and use private land where they consume livestock feed and cause damage to private property (Scasta and others, 2018). The presence of WHBs can also negatively impact habitat restoration efforts (Griffin and others, 2019).

Free-Roaming Equid Management

Free-roaming equid populations have relatively high population growth rates (Garrott, 2018). Population growth rates average 15–20 percent per year (Garrott, 2018), and increases as high as 39 percent have been observed (Ransom and others, 2016). Native predators do not regulate WHBs populations (Garrott, 2018); consequently, free-roaming equid management is focused on herd reduction. Free-roaming, unclaimed, stray horses that do not have Federal protections can be removed from the range and treated with fertility-control methods pursuant to applicable State laws.

The BLM and Forest Service gather WHBs from the range in areas where the populations are impacting the rangeland and the health of the animals is compromised

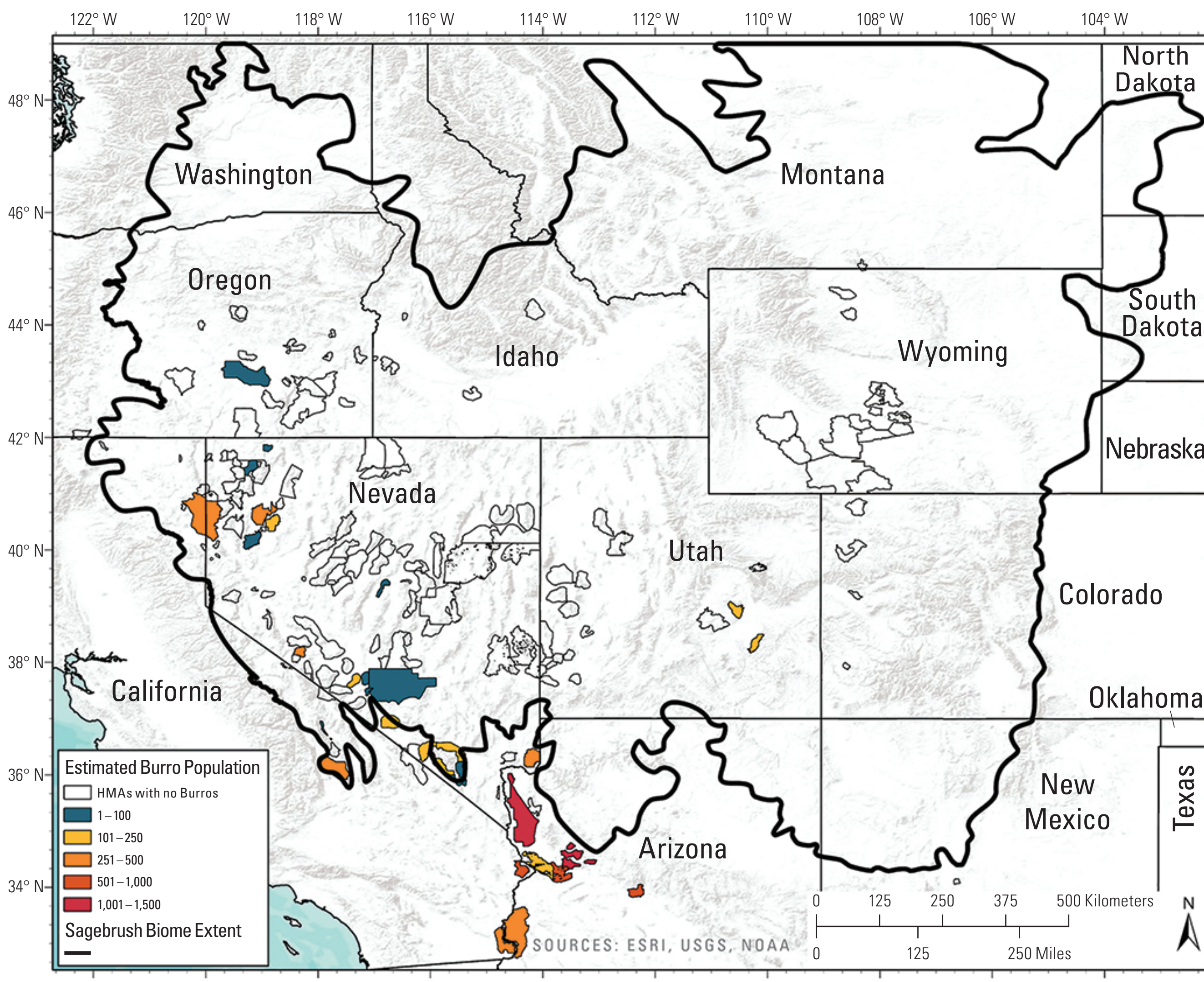

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Figure N2. Estimated populations of wild burros (*Equus asinus*) administered by the U.S. Department of the Interior, Bureau of Land Management by designated herd management areas (HMAs) in the sagebrush (*Artemisia* spp.) biome, March 2019 (Bureau of Land Management, 2019d).

(Scasta and others, 2018). However, if gathered animals are not adopted or sold under applicable legal limitations, the agencies must care for them for the remainder of their lives. In fiscal year (FY) 2018, the BLM spent $49.8 million, 61 percent of its $81.2 million WHB program budget, to care for animals in holding facilities (Bureau of Land Management, 2018b). The BLM has estimated that the costs of caring for one unadopted wild horse over its lifetime will exceed $48,000 if it is exclusively kept in a corral (Bureau of Land Management, 2018b). The lifetime cost is lower for animals that are maintained on long-term pastures. The Forest Service operates two short-term holding facilities including the new Double Devil Wild Horse Corral in California but has no contracts for long-term pastures.

The BLM developed a 5-year gather schedule to achieve AML by 2020 in 22 HMAs that overlapped areas identified as the most important habitats for greater sage-grouse and other sagebrush obligates. However, under budget projections made in FY 2017, the BLM will not have the fiscal capacity to conduct gathers within greater sage-grouse priority habitat management areas (PHMAs) until 2020 or later and has no capacity to manage wild horse populations that overlap with greater sage-grouse general habitat management areas. Given population growth rates under current management realities the WHB population could exceed 179,000 animals in 2023 (Garrott, 2018). Most HMAs and WHB territories are well above AML (figs. N3 and N4).

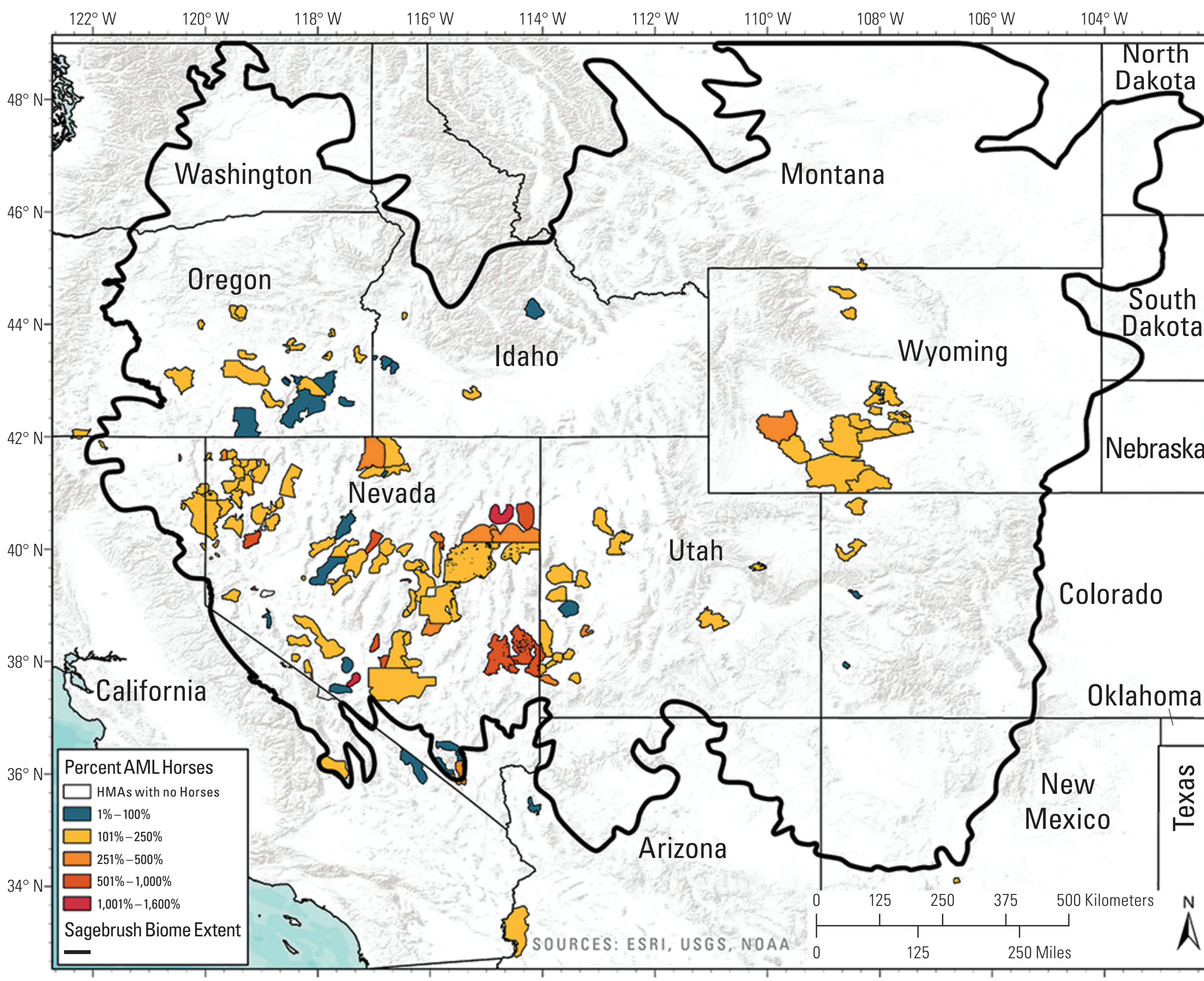

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Figure N3. Estimated U.S. Department of the Interior, Bureau of Land Management-administered wild horse (*Equus caballus*) population size compared to appropriate management levels (AML) in designated herd management areas (HMA) in the sagebrush (*Artemisia* spp.) biome, March 2019 (Bureau of Land Management, 2019d). %, percent.

In a 2018 report to Congress, the BLM outlined four management options, costs, and potential timelines for reducing WHBs to AML, initially within sage-grouse PHMAs (Bureau of Land Management, 2018b):

1. Achieve AML (in priority HMAs) in 8 years, using all the authorities within the WFRHB Act, while substantially decreasing off-range holding costs. Funding needed to implement over this period is about $115 million per year.

2. Achieve AML (in priority HMAs) in 10 years using existing authorities by substantially increasing program funding. Funding needed to implement over this period is about $116 million in FY 2019 increasing to $246 million in FY 2027.

3. Achieve AML (in priority HMAs) in 6 years using existing authorities and creating an adoption incentive program. Funding needed to implement over this period is about $133 million in FY 2019 increasing to $147 million in FY 2023.

4. Achieve AML (in priority HMAs) in 12 years using existing authorities, creating an adoption incentive program, and increasing permanent sterilization. Funding needed to implement over this period is about $135 million in FY 2019, increasing to $143 million in FY 2023.

The only tool used broadly enough to make systemic reductions in WHB populations has been the capture and removal of animals from western rangelands (Bureau of Land Management, 2017; Hendrickson, 2018). All of the

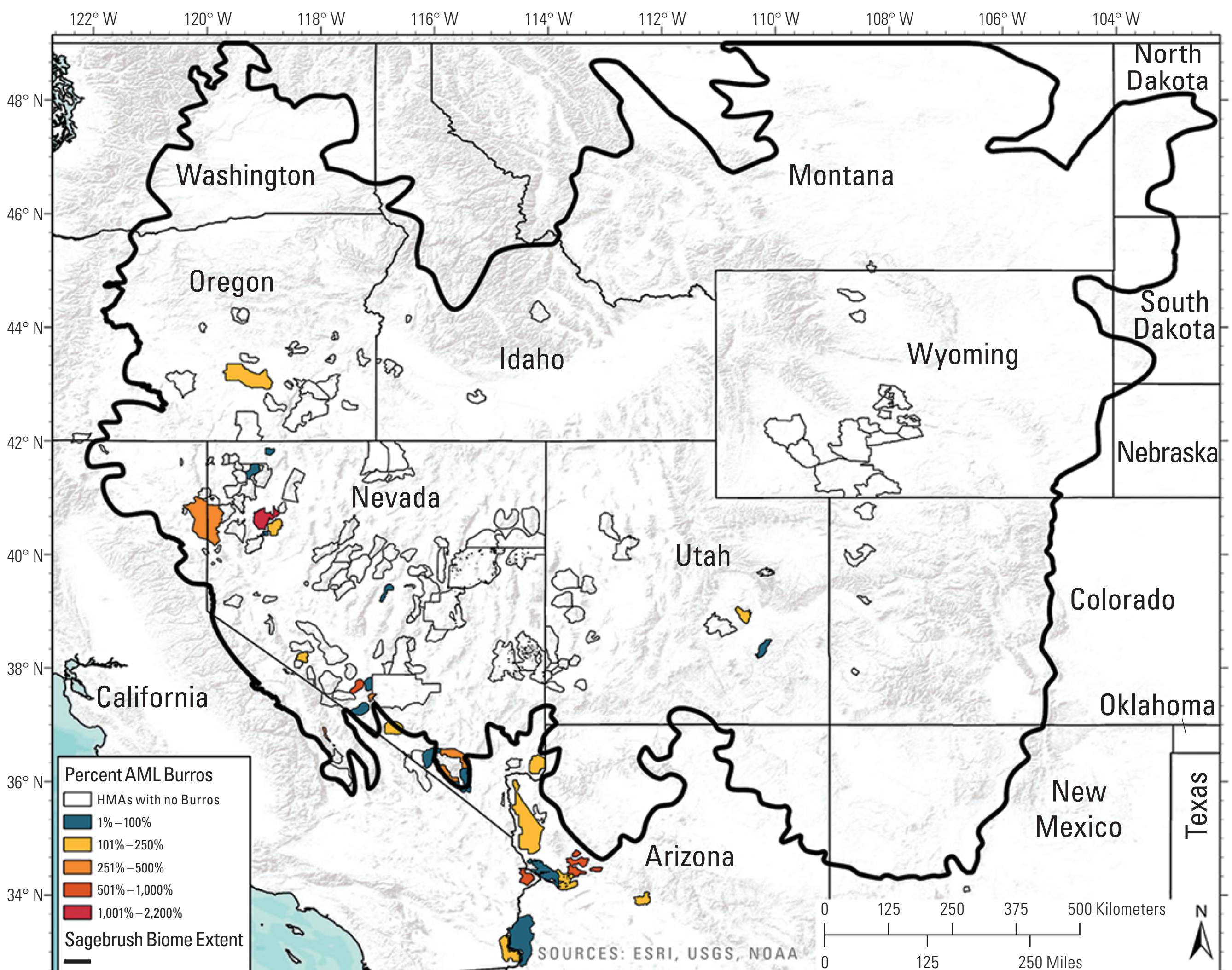

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Figure N4. Estimated U.S. Department of the Interior, Bureau of Land Management-administered wild burro (*Equus asinus*) population size compared to appropriate management levels (AMLs) in designated herd management areas (HMAs) in the sagebrush (*Artemisia* spp.) biome, March 2019 (Bureau of Land Management, 2019d). %, percent.

BLM management options identified in the 2018 Report to Congress will require the increased and widespread gathering and handling of WHBs to implement the proposed population reduction strategies. Implementing any of the options will require different levels of funding, time, agency persistence, National Environmental Policy Act (NEPA) of 1969 (42 U.S.C. 4321 et seq.) analysis, litigation support, and Congressional and stakeholder support. The Science Framework For Conservation and Restoration of the Sagebrush Biome, Part II (Griffin and others, 2019) described approaches that can be used to prioritize areas for gathers and application of other management tools to reduce impacts of free-roaming equids to greater sage-grouse and other sagebrush species based on principles of resilience and resistance to invasive annual grasses (Chambers and others, 2017a).

Free-Roaming Equid Fertility Management

Intensive fertility-control methods have been effective at reducing or maintaining populations to AML only in small closed herd units (that is, herds with no immigration or emigration) with regularly approachable animals. Populations have primarily been regulated through gathers (Kane, 2018). Fertility-control methods (see review by Kane, 2018) have the potential to slow wild horse herd growth rates, but these methods generally have not reduced actual herd sizes. Fertility-control methods cannot maintain wild horse herds near AML unless enough animals are first removed to bring the herd down to AML, and a high fraction of the remaining animals must be treated frequently enough to limit fertility over the long-term. The BLM has concentrated fertility-control efforts on females because the number of mares receiving contraceptives directly reduces population growth. When a herd of wild horses included some spayed mares, population growth rates were reduced approximately in proportion to the fraction of mares spayed (Collins and Kasbohm, 2017).

In recent years, the BLM has administered fertility control vaccines to fewer than 800 mares per year. Current vaccines are effective for only 1–2 years (see sidebar). Because horses in most herds are not approachable, mares need to be captured every time they are given a dose, with net costs of approximately $2,500 per mare.

Stallions can be sterilized by castration (also known as neutering or gelding) or vasectomy. Sterile stallions may reduce mare fertility rates if they prevent fertile stallions from mating, but a single fertile stallion may mate with many mares. Still, fertile mares had reduced fertility even when only approximately 40 percent of stallions were vasectomized (Collins and Kasbohm, 2017).

The BLM faces litigation in opposition to most proposed management actions that include fertility control as a part of WHB management, whether the proposed method includes fertility control vaccines, gelding stallions, or spaying mares (Norris, 2018). With current herd sizes over AML and given the cost to capture and treat a single mare and the limited duration of vaccine effects, reducing herds will be necessary if ecosystem damage is to be reduced. Fertility control methods have the potential to reduce herd growth rates but will be most effective in areas where herds are already at or close to AML.

Human Dimensions and Free-Roaming Equids

The domestication of the horse, the intimate human-horse trust relationship, and the versatile role horses played in the development of human society have demanded a level of care and respect that has few analogues (Levine, 1999; Robinson, 1999; Kelekna, 2009; Scasta, 2019). The strength of the emotional human-horse connection has been the impetus for legislation protecting free-roaming equids, including the Wild Horse Annie Act (18 U.S.C. 47; Smith, A.V., and others, 2016) in the late 1950s to deter “mustanging” (private individuals capturing and raising horses for profit).

Fertility Control Agents

The GonaCon vaccine ($50 per dose) is only roughly 30–40 percent effective for 2 years after the first dose but has an effectiveness of 100-percent reduced fertility rate for 1 year after a booster dose with another 3 years of over 80 percent effectiveness after that (Ransom and others, 2011; Baker and others, 2018). Liquid PZP ($30 per dose) is the most commonly used vaccine; it contracepts 95 percent of treated mares for 1 year, though mares treated four or more times can have longer lasting effects (Nuñez and others, 2017). PZP-22 vaccine pellets ($510 per dose) have a variable 1-year effectiveness of between 30 and 70 percent, with second-year effectiveness dropping to 40 percent or less. In a horse that has already received a dose of PZP-22 vaccine pellets, though, a subsequent liquid PZP booster dose can lead to 2–3 years of contraception with roughly 60–85-percent effectiveness (Rutberg and others, 2017). SpayVac PZP vaccine ($450 per dose) caused long-lasting contraceptive effects from one dose in early trials (Killian and others, 2008) but for unknown reasons performed poorly in the most recent tests in horses.

Management Considerations

The Wildlife Society and the Society for Range Management hosted the "Free-Roaming Equids and Ecosystem Sustainability Summit," on May 29–31, 2019, in Reno, Nevada. The stated purpose of the summit was to develop a stakeholder-based, comprehensive communication strategy and processes to manage free-roaming equids in concert with other public lands multiple-uses to achieve western rangeland ecosystem sustainability. The majority of attendees supported releasing a shared statement that identified areas of potential agreement in future management direction:

- Management of free-roaming horses and burros must be respectful of animal welfare, other public land multiple-uses, and must maintain rangeland health.
- Each area inhabited by free-roaming horses and burros should be managed based on its ecological state, current free-roaming horse and burro populations, and health of land and animals.
- Most HMAs inhabited by WHBs exceed ecological carrying capacity.
- Gathers are the only current means for removing excess WHBs and thus should integrate fertility control options with animal removal.
- Management actions must achieve an ecologically sustainable management level of free-roaming horses and burros through nonlethal means. Investments will be significant initially but will decrease over time as more efficient fertility-control methods become available and as numbers of horses in long-term holding facilities decrease through adoption and natural mortality.
- Free-roaming horse and burro fertility-management research is necessary to develop new techniques.
- The application of existing fertility-control methods should be used based on efficacy specific to the WHBs HMAs. Stakeholders' inability to achieve broad consensus and actions are likely to predicate actions and policies that are unacceptable across the entire spectrum of stakeholders.
- Unified messaging regarding WHB management needs, exponential growth of herds and corresponding ecological damage, and the need for long-term funding is essential.

Working within the above constructs provides challenges to rangeland and wildlife managers in maintaining rangeland ecosystem health. Satisfactory resolution of this complex and often emotional issue will require both biological science and an understanding of human dimensions.

Chapter O. Mining and Energy

By Cameron L. Aldridge,[1] Anna Chalfoun,[1] Patricia A. Deibert,[2] Shawn P. Espinosa,[3] Matthew J. Holloran,[4] and Amanda Withroder[5]

Executive Summary

Mining and energy development are necessary to provide resources to meet human needs, and energy is a current national priority. Many of these essential resources are located in the sagebrush (*Artemisia* spp.) biome, providing a significant economic contribution to individuals and local and State economies. Mining and energy development are regulated by Federal agencies, such as the U.S. Department of the Interior, Bureau of Land Management and Office of Surface Mining and Reclamation, as well as State agencies. Mining and development of energy resources have varying impacts on sagebrush habitats, including direct habitat removal or fragmentation, introduction of invasive plant species, and potential impacts on surface and groundwater. Associated facilities—such as roads, processing facilities, transmission lines, and pipelines—have similar impacts.

Approximately 8 percent of sagebrush habitats across the entire biome are directly affected by oil and gas development, with greater than 20 percent of sagebrush habitats affected in the Rocky Mountain area. Several million additional acres within the sagebrush biome have been impacted by mining activities and alternative energy development, such as wind and solar. Sagebrush-associated wildlife can be impacted by loss and degradation of habitat, as well as by numerous indirect effects such as noise, exposure to contaminants, and disturbance from vehicles and human presence. The actual impact depends on the development location, scale of the project, and how affected habitat is used by wildlife. Numerous Federal and State regulations and policies have provisions to reduce impacts to habitat and wildlife. Restrictions and conservation actions primarily apply to greater sage-grouse (*Centrocercus urophasianus*) habitats. The effectiveness of these measures for conserving sage-grouse or other sagebrush-associated species is uncertain. Cumulative impacts of mining and energy development are poorly understood. Overall, a better and more holistic understanding of how energy development and mining affect the long-term functioning of sagebrush ecosystems, and the persistence of associated wildlife species is needed.

[1]U.S. Geological Survey.

[2]U.S. Fish and Wildlife Service.

[3]Nevada Department of Wildlife.

[4]Operational Conservation.

[5]Wyoming Game and Fish Department.

Introduction

The following description detailing mining and energy development effects on sagebrush (*Artemisia* spp.)-associated wildlife species should be considered incomplete, as impacts from these activities to sagebrush ecosystem processes, individual species, and synergies between them are not fully understood. This chapter focuses on energy and mining impacts and does not fully consider the associated infrastructure such as roads, transmission lines, pipelines, increased human presence, or other indirect effects of associated development, such as incursion of invasive plants (see chap. P, this volume). The impacts of land use development and invasive plant species on sagebrush ecosystems are addressed in separate chapters of this document (see chap. K and chap. P, this volume, respectively, for more detailed information on these topics).

The actual impacts of mining and energy development to a particular species will depend on the location and extent of the disturbance. For example, a large coal mine in habitat adjacent to sagebrush habitat (that is, prairie or mountain shrub communities)—or directly within sagebrush habitat that provides little function for sagebrush wildlife species—may be less impactful than a small gravel pit that occurs within a key migratory corridor or in limiting key or seasonal habitats. Direct impacts to ecosystem functioning and resource conditions will vary based on location and current ecosystem health, and individual species' responses will differ. When determining the actual impact of a mining or energy development activity on the ecosystem or a sagebrush wildlife species, the location and extent of the development and associated structures need to be considered for an accurate assessment, in addition to the context of how those species use the ecosystem and its resources.

Mining

Mining (including exploratory drilling for energy resources) is important to the recovery of energy resources (for example, coal, uranium, and oil) and to the recovery of other minerals and resources used for either industrial or commercial purposes (for example, gravel, lithium, and gold). Mining is an important economic driver in the United States, with an estimated value of nonfuel minerals produced in 2017 at $75.2 billion (U.S. Geological Survey, 2018c). Many States within the sagebrush ecosystem are significant producers of both nonfuel minerals and coal (table O1).

Types of Minerals within the Sagebrush Ecosystem

The three classifications of minerals on federally administered lands within the sagebrush ecosystem are locatable, leasable, and salable. Locatable minerals include mostly metallic mineral deposits, leasables include mostly energy products, and saleables are used primarily for construction purposes (table O2). Regulations regarding extraction and associated reclamation of these three categories vary (as described in the individual sections below), but mining extraction methods—and therefore their potential impacts on sagebrush species—are similar for all three categories. Development activities (for example, leasing) for all Federal mineral rights are handled by the U.S. Department of the Interior, Bureau of Land Management (BLM), but management of the associated mining development and operation activities are subject to regulations of the surface owner.

Mineral extraction on Federal lands and of Federal subsurface resources are subject to the Mining and Minerals Policy Act of 1970 (30 U.S.C. 21a; all Federal lands), the Federal Land Policy and Management Act of 1976 (FLPMA; 43 U.S.C. 1701–1785; BLM), the National Forest Management Act of 1976 (NFMA; (16 U.S.C. 1600–1614; U.S. Department of Agriculture [USDA] Forest Service [Forest Service]) and the National Environmental Policy Act (NEPA) of 1969 (42 U.S.C. 321 et seq.). The Mining and Minerals Policy Act encourages the development of domestic mineral resources and requires mined land reclamation. The FLPMA mandates BLM land management for multiple uses. Specific to mining, the FLPMA includes language on preventing “undue and unnecessary degradation of public lands.” The Forest Service permits mining through NFMA, which requires the agency to make minerals from National Forest System lands available to the national economy, while minimizing any adverse impacts of mining activities on other resources. The NEPA assures that all branches of government give appropriate consideration to the environment prior to undertaking any major Federal action. A key part of the NEPA process is to ensure that decision makers and the public are informed of possible environmental impacts and consequences of a proposed action and any associated minimization measures.

Overview of Impacts of Mining to Sagebrush and Sagebrush Wildlife Species

Energy resources that are obtained via mining include coal, uranium, and lithium. Approximately 90 different nonenergy resources are also mined, including sand, gravel, bentonite, gold, silver, copper, diamonds, gypsum, lime, rare earth elements, and decorative rock (U.S. Geological Survey, 2018c). The density and expanse of mining activities varies depending on the location of the desired resource, ease of access, market commodity prices, and associated regulations governing extraction. Retrieval methods of the desired resource vary by type of mineral, amount, and type of

Table O1. Nonfuel minerals and coal production in 2017 for States within the sagebrush (*Artemisia* spp.) biome. The table reports values for the entire State, across all lands. Some of the reported information may therefore encompass areas outside the sagebrush ecosystem.

State	Nonfuel minerals 2017 production value in millions of dollars	Nonfuel minerals principal produced in each State in order of value[1]	Coal 2017 production in tons by all methods[2]
Arizona	6,610	Copper, sand and gravel, molybdenum, cement, crushed stone	6,221
California	3,520	Sand and gravel, cement, boron, crushed stone, gold	0
Colorado	1,680	Gold, cement, sand and gravel, molybdenum, crushed stone	15,047
Idaho	191	Phosphate, sand and gravel, crushed stone, lead, silver	0
Montana	1,050	Palladium, copper, platinum, sand and gravel, molybdenum	35,232
Nevada	8,680	Gold, copper, crushed stone, sand and gravel, silver	0
New Mexico	1,310	Copper, potash, sand and gravel, crushed stone, cement	13,843
North Dakota	72	Sand and gravel, lime, crushed stone, clay	28,788
Oregon	474	Crushed stone, sand and gravel, cement, diatomite, perlite	0
South Dakota	372	Gold, cement, sand and gravel, crushed stone, lime	0
Utah	2,610	Copper, magnesium, gold, potash, sand and gravel	14,326
Washington	901	Sand and gravel, crushed stone, gold, zinc, cement	0
Wyoming	2,410	Soda ash, helium, bentonite, sand and gravel, cement	316,455
Totals	**29,880**		**429,912**

[1]U.S. Geological Survey Mineral Commodity Summaries Report (U.S. Geological Survey, 2018c).

[2]U.S. Coal Production by State, 2006–2017 (National Mining Association, 2018).

Table O2. List of locatable, leasable, and saleable minerals within the sagebrush (*Artemisia* spp.) biome.

Locatable minerals	Leasable minerals	Saleable minerals
Bentonite	Bitumen	Cinders (scoria)
Copper	Coal	Common clays
Fluorspar	Geothermal steam	Dirt
Gold	Gilsonite (native asphalt)	Gravel
Gypsum	Natural gas	Pumice
Chemical-grade limestone	Oil	Rock
Lithium	Oil shale	Sand
Mica	Phosphate	
Nickel	Potash	
Platinum	Potassium	
Precious gems	Silica (in Nevada only)	
Silver	Sodium	
Uranium	Sulphur	
Zinc	Quicksilver	
Most other metals	Hard rock minerals on acquired lands[2]	
Nonmetallic industrial minerals		
Uncommon varieties of pumice		
Silica, rock, and cinders[1]		

[1]Normally saleable resources that are limited or uncommon in a State may be considered as locatables in limited areas. The designation is determined by Federal certified mineral examiners on a case-by-case basis.

[2]Acquired lands are those lands obtained by the Federal government via direct purchase, condemnation, gift or exchange. They can also include Tribal lands.

overburden (that is, layers of nontarget soil and rock above the desired mineral), amount of the target mineral, and physical conditions (for example, slope).

Mining can occur above ground in open pits (for example, contour and strip mining), underground with surface portals (for example, long wall and deep mining), or through solution where the desired target is dissolved in an injected fluid and subsequently pumped out of the ground. The type of mining extraction and number and type of associated facilities (roads and processing plants, among others) will determine the impacts of mining on sagebrush wildlife species. While most regulations regarding mining require minimizing impacts to wildlife and collecting wildlife data, there is limited information on the actual impacts of mines on sagebrush habitats and sagebrush wildlife species. Most mining studies have been performed at a mine-site scale, only assessing the direct response of wildlife within the mining and associated reclamation footprint. Landscape-scale studies are limited and mostly focused on vegetation changes (Buehler and Percy, 2012). Long-term studies or studies on the impacts of mining on species' vital rates are absent.

Direct impacts of mining on wildlife can include individual mortality when the species is not mobile enough to avoid mining equipment (for example, reptiles, amphibians, and small mammals) or when individuals are struck by vehicles traveling to and from the mining location (Buehler and Percy, 2012). Displacement of individuals or populations from the mine site can occur as a result of habitat loss, noise, ground shock from blasting or crushing activities, and an increase in human presence and activity. Impacts of displacement can include interruption of breeding seasons, lower survival because of increased competition for limited resources, and increased predation rates owing to concentration of prey animals or unfamiliarity with areas of displacement. The long-term effects on survival and reproduction are mostly unknown (Buehler and Percy, 2012).

Surface and subsurface mining results in direct loss of habitat, with the loss typically greater from surface mining versus subsurface activity. Habitat loss from both types of mining can be exacerbated by the storage of overburden in otherwise undisturbed habitat. If the construction of mining infrastructure is necessary, additional direct loss of habitat could result from structures, staging areas, roads, railroad tracks, and powerlines (chap. P, this volume; Monroe and others, 2020). Depending on the mine location, migratory corridors may be disrupted, potentially precluding migratory species from reaching key seasonal habitats or increasing energy costs to reach them (Blum and others, 2015). For endemic species or local populations, habitat loss and fragmentation from mining activities may affect the ability of the species or population to persist in the affected area.

Other indirect impacts from mining can include degradation of water quality and reduction in quantity. Water quality can be impaired by contamination from leaching of waste rock and overburden, chemicals used for mineral extraction, or by fertilizer used for reclamation activities;

Moore and Mills, 1977). Water quantity can be reduced by altered water regimes decreasing surface water availability and by changes in vegetation and topography. Stream crossings or stream relocations may also affect water quality and availability, which could have significant impacts in sagebrush ecosystems where water is limited and disproportionately distributed. Direct mortality to aquatic species and riparian animals may occur during stream alteration. Soil placed in streams may also contain contaminants or other materials that can change the water chemistry, such as acidity and conductivity. Changes in water quality and quantity may result in changes in the number and types of species living in and around the stream. Invertebrates and salamanders are especially sensitive to changes in water flow and chemistry (Moore and Mills, 1977).

Increased noise can mask animal vocalizations, communications that are key to successful mating activities (for example, sage-grouse [*Centrocercus* spp.]; Remington and Braun, 1991; Blickley and others, 2012a; Blickley and Patricelli, 2012; Patricelli and others, 2013); reduce ability to avoid predators; or limit ability to maintain social units. There may be increased predation on sagebrush wildlife species from predators—particularly novel predators—using mining and associated infrastructure for perching and nesting substrates. For example, the common raven (*Corvus corax*), a key predator of sage-grouse nests, selected transmission lines and land-cover types that were associated with human disturbance or fire for nesting (Howe and others, 2014). Fugitive dust or particulate matter that is released into the environment from mining activities, which is common with surface mines, potentially interferes with plant palatability, photosynthesis (and therefore plant productivity), and insect population abundance and occurrence, as well as reducing air quality (Moore and Mills, 1977; Brown and Clayton, 2004). Large surface mines may mitigate this potential negative because of regulatory requirements to control fugitive dust. Sagebrush wildlife species may also be exposed to pathogens introduced from septic and waste disposal systems (Moore and Mills, 1977).

Mining and associated activities create an opportunity for invasion by exotic and noxious grass and weed species that alter habitat suitability and availability for sagebrush and associated species (Moore and Mills, 1977). Invasive plant species, particularly annual grasses, are considered one of the primary threats to the long-term persistence and integrity of the sagebrush ecosystem (U.S. Fish and Wildlife Service, 2013; Chambers and others, 2017a; see also chap. K, this volume). The impacts of invasive plant species on sagebrush wildlife species vary by habitat requirements, but generally result in loss of functional habitat owing to alteration of the sagebrush understory and direct habitat loss if the invasive plant species facilitate wildfire occurrence (for example, Manier and others, 2013).

The following section details the different types of mining that occur within the sagebrush ecosystem. Impacts to sagebrush or sagebrush wildlife species specific to each mining type that vary from those described above are discussed.

Coal Mining

The United States contains the largest coal reserves in the world and is the world's second largest coal producer. More than two-thirds of coal produced in the United States comes from large coal mines in the West, and approximately 80 percent of that coal comes from several large mines in Wyoming (table O1; Naugle, 2011). There are approximately 6 million hectares (ha; 14.8 million acres) of active coal leases in western North America (Naugle, 2011).

Surface mining accounts for the majority (two-thirds) of coal mining operations in the western United States. These operations are concentrated in large coal fields located across Arizona, Colorado, Montana, New Mexico, North Dakota, Utah, and Wyoming. There are approximately 9.4 million ha (23.3 million acres) of mapped coal fields within the western sagebrush biome (fig. O1). Total acres of existing disturbance owing to past or active coal mining are unknown.

The Surface Mining Control and Reclamation Act (SMCRA) of 1977 (30 U.S.C. ch. 25 1201 et seq.) is the primary Federal law that regulates surface coal mining and abandoned mine lands. The SMCRA specifically requires the use of best technology, minimization of disturbance and adverse impacts, resource enhancement, avoidance of disturbance in habitat with unusually high value, and selection of plant species for reclamation based on species' nutritional value, cover, ability to support and enhance habitat, and ability to minimize edge effects. This law also requires life-of-mine planning and reclamation. Reclamation requirements include topsoil salvage and replacement, protection of areas adjacent to mining, surface water runoff control and treatment, road placement and design, and environmental monitoring and reporting. Reclamation plans are developed based on the approved postmine land use and require plans for topsoil replacement and seeding. Seed mixtures must be specified and meet vegetation diversity requirements. Where shrub establishment is prescribed, a shrub density standard must be met. Postmine land use changes are permitted under some regulations (for example, SMCRA; Boyce, 2002), potentially resulting in permanent sagebrush habitat loss.

The U.S. Department of the Interior, Office of Surface Mining Reclamation and Enforcement (OSMRE), established by SMCRA, administers regulatory programs on surface coal mining and the surface effects of underground coal mining operations. Many western States within the sagebrush biome have jurisdiction to regulate both Federal and non-Federal coal operations through cooperative agreements with the Secretary of the Interior which give them primacy, including Colorado, Montana, New Mexico, North Dakota, Utah, and Wyoming (Office of Surface Mining Reclamation and Enforcement, 2018). These six western States have extensive regulatory programs which are required to be no less effective than the OSMRE regulatory program.

The BLM is responsible for the leasing of Federal coal under the Federal Coal Leasing Amendments Act of 1976 (30 U.S.C. 201 et seq.). Interested operators must collect extensive baseline vegetation and wildlife data, which is reviewed by Federal and State agencies. Federal coal leases are screened for unsuitability considerations, namely, habitat for resident wildlife species of high interest to the State that is necessary for maintaining those species. Consultation with the U.S. Fish and Wildlife Service (FWS) must occur if any species listed as threatened or endangered under the Endangered Species Act (ESA) of 1973 (16 U.S.C. 1531 et seq.) may be affected by mining activities. Special wildlife-specific lease stipulations may be applied to Federal coal leases. The leasing process may lead to the exclusion of areas of biological importance and provide for protective stipulations to minimize the impacts of mining on sagebrush-associated species. Permitting is typically a 2-year process with multiple opportunities for public comments and consultation with State and Federal wildlife agencies to make recommendations for reducing or mitigating the impacts of mining on sagebrush-associated species. Permits are renewed every 5 years, allowing for modifications based on required monitoring reports.

Coal exploration involves truck-mounted drill rigs and trucks/other heavy equipment and may include mud pits. Drilling and off-road travel may disturb wildlife and result in vegetation crushing or loss. Actual coal retrieval can be through open pit mines or underground shafts, depending on the depth of the overburden. Underground mining has far fewer impacts on surface resources; however, it is often economically more expensive. After the coal is recovered, it is cleaned, processed, and transported for direct use. Coal mine operations involve large facilities, heavy equipment, the use of explosives, and extensive land-based transportation systems.

The direct and indirect impacts of actual coal mining are similar to those described above. Impacts associated

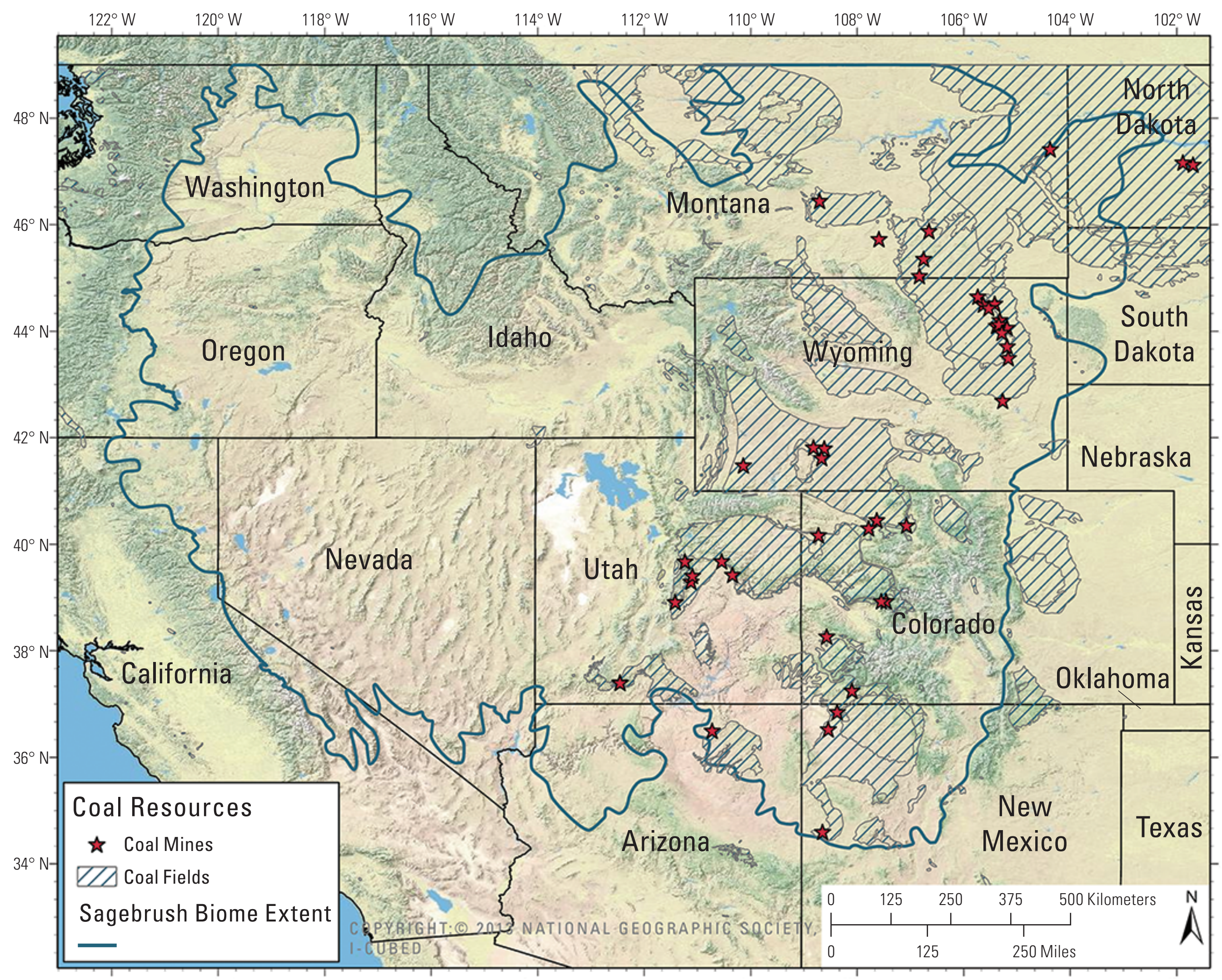

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Figure O1. Coal mines (U.S. Energy Information Administration, 2019e) and coal fields (East, 2013) located within the sagebrush (*Artemisia* spp.) biome as of 2016.

with coal mining are primarily related to the large-scale and long-term alterations (often several decades) of the surface as a result of surface coal mining. As a result of relatively rigorous requirements for monitoring coal mines, as per SMCRA, there is an abundance of wildlife data available for both active and reclaimed mine sites. These data are not typically comprehensively evaluated to draw conclusions regarding species-specific impacts or potential landscape effects. There are few studies (if any) demonstrating the effects of coal mining on sagebrush-associated species. Yet, a study in Colorado established that persistent mining activities have resulted in decreased recruitment to sage-grouse leks (Remington and Braun, 1991), with the long-term population effects and mechanistic causes of decline unknown.

Locatable Minerals

Since 1873, DOI defined locatable minerals as those minerals that are (1) recognized as a mineral by the standard experts, (2) are not subject to disposal under some other land use, and (3) make the land more valuable for mining than agricultural purposes. Locatable mining is specifically regulated by the Mining Law of 1872 (30 U.S.C. 28), but is also subject to controls under FLPMA, NFMA, and NEPA. The Mining Law of 1872 provides opportunities to explore, discover, and purchase certain valuable mineral deposits on Federal lands that are open for mining claim location. Valid and existing rights associated with mining claims cannot be significantly infringed upon, and most mining-related decisions are thus considered nondiscretionary by the Federal land manager. State laws that are consistent with Federal laws can be used to govern locations and recording of mining claims.

Locatable mineral development requires the establishment of a mining claim. Lode claims are associated with mineral veins and have other rock in-place that bears valuable mineral deposits. A lode claim is limited to a maximum of 457 meters (m; 1,500 feet [ft]) in length along the vein with a width of 183 m (600 ft)–104 m (300 ft) on either side of the centerline of the lode). Placer claims cover deposits not subject to lode claims. The maximum size of a placer claim is 8.1 ha (20 acres). However, adjacent placer claims can be made up to a maximum of 65 ha (160 acres).

Exploration activities associated with locatable minerals on public lands does not require permitting by the BLM if the disturbance is less than (<) 2 ha (5 acres) but requires the operator to file a notice of intent. If surface disturbance is greater than (>) 2 ha (5 acres), an exploration plan of operations is required. If exploration results in an economically viable project, a mining plan of operations is developed and submitted to the pertinent Federal land management agency for analysis and approval. Through the NEPA process, design features; seasonal timing restrictions; or other means to avoid, minimize, or compensate for negative impacts may be applied. During the NEPA process, cooperating agencies and the public may provide input that can be used to formulate alternatives; describe potential impacts; and recommend avoidance, minimization, and mitigation strategies. Exploration and development activities must also be permitted by the U.S. Department of State Office of Environmental Quality and Transboundary Issues regardless of disturbance size.

Extraction of many locatable minerals typically requires development of an underground mine or open pit mine (example in fig. O2). Waste material and ore are extracted through the use of drilling, blasting, loading, and transporting via haul trucks. Minerals can be separated from ore through a variety of physical or chemical processes. Most commonly, precious metals are separated through heap leach or milling. Heap-leach production involves placing large quantities of low grade ore on a lined facility and distributing cyanide (for gold or silver) or sulfuric acid (for copper) solutions to the top of the pad where it percolates through the ore and dissolves the desired metal from the ore. Solutions containing the desired metal are collected and processed through a system of collection pipes and lined ponds. Extraction of uranium is similar, but the mineral is dissolved in place, with onsite injection wells and submersible pumps to extract the pregnant solution to the ground surface. The solution is delivered to a processing facility (typically onsite) via a pipeline. The design of an in situ uranium well field varies depending on overburden permeability, deposit type, and ore grade and distribution. Wells (both injection and extraction) are spaced from 20 to 60 m (66–197 ft) apart. Mining operations for uranium occur at a local scale, focused on discrete deposits. Activities associated with drilling, extracting, and processing typically result in the loss of vegetation at the mine location.

Mill production involves passing high-grade ore through one or more mills to create slurry that undergoes additional chemical processing to remove precious metal. Once precious metals are recovered, the remaining slurry is considered waste and is discharged and stored in a tailings storage facility. Additional facilities associated with locatable mineral mines often include structures such as waste rock dumps, rapid infiltration basins, production shafts, quarries, maintenance facilities, office buildings, transmission and distribution lines, roads, pipelines, and ponds. Comprehensive information on the number or surface extent of locatable mineral mines across sagebrush ecosystems is not available (but see fig. O3), though the development of some mineral resources is occurring on a large scale, which is important to the economies of some States in the sagebrush biome (table O1).

Impacts from most locatable mineral mining are captured in the overview of impacts of mining to sagebrush and sagebrush-associated wildlife species section of this chapter. Additional environmental risks with the use of chemicals to retrieve some locatable minerals include potential impacts to groundwater and surface water resulting from contamination of leaching solvents and potential exposure of sagebrush wildlife species to heap piles. In situ uranium mining also has the potential for migration of production liquids from the production aquifer to the surrounding aquifers during operation, including the movement of constituents to groundwater outside the licensed area, excessive consumption of groundwater, residual constituent concentrations in excess of baseline concentrations after the restoration of the production aquifer (although concentrations are monitored), and a mechanical failure of the subsurface well materials releasing production fluids into the overlying aquifers (although this risk is slight as wells are cased).

Mining claims for locatable minerals may also have an increased impact on sagebrush- associated songbirds, small mammals, and reptiles. Boundaries of mining claims must be marked with a minimum of four 0.91 m (3 ft)-tall posts, and the material of choice is typically 4-inch hollow, white PVC pipe because of durability and low costs. Birds, bats, small mammals, and reptiles are attracted to these artificial openings and can become entrapped in the markers. In most instances, they are unable to escape and subsequently die of dehydration or starvation. In 2010, there were 3,388,400 mining claims registered on BLM and Forest Service lands (BLM Public Land Statistics as cited by American Bird Conservancy, 2012) across 12 western States. The highest number of mining claims occur in Nevada (>1,000,000), and an additional 1,200,000 claims occur in California, Colorado, Utah, and Wyoming. Estimated mortality by taxa associated with these pipes is unknown, but up to 32 birds have been found in a single pipe (BLM Public Land Statistics as cited by American Bird Conservancy, 2012). There are no spatial data that allow for an accurate assessment of the overlap of mining claims with sagebrush habitats, so the impact of these pipes to sagebrush-associated wildlife is unknown.

Various regulatory mechanisms have been implemented across several States to minimize wildlife mortality resulting from mining claim pipes. Within the sagebrush ecosystem, Nevada and California specifically prohibit the use of open pipes for use in identifying mining claims. State law in Nevada allows for any uncapped post located on public lands to be knocked down. The BLM issued Instruction Memorandum (IM) 2016-023, “Reducing Preventable Wildlife Mortalities,” directing field staff to identify and, where feasible, modify open pipe structures to preclude wildlife entry. The IM also encourages operators to voluntarily replace existing open pipes with wildlife safe structures.

Voluntary efforts to reduce animal mortality in marking pipes are encouraged by the BLM, State wildlife agencies, and nongovernmental organizations. Nevada has created the Mine Claim Marker Removal Project, focusing on locating and removing hollow mine claim markers across that State and estimating associated wildlife mortality through analysis of the post contents. Local State and Federal agencies, along with a large and diverse suite of nongovernmental organizations, have been actively locating and removing these markers since 2011. This effort is labor intensive, and the actual number of posts and their locations are unknown in Nevada, making this effort unlikely to be 100-percent effective. Similar efforts in other States are not known.

Figure O2. Carlin Trend Gold Mine near Carlin, Nevada, taken on March 12, 2014. Photograph by Matt Maples, Nevada Department of Wildlife, used with permission.

Other Mining Activities

Retrieval of other mineral resources, such as gravel and sand, is similar to the processes described for coal and other surface mining, with similar potential impacts to wildlife. Given the high transportation costs for these materials and the low unit price, the BLM permits development of many local mines near project sites. Bonding may be required for any mineral resource development on public lands to prevent unnecessary or undue degradation and to reclaim the land after mining is complete (Bureau of Land Management, 2019a).

Unlike locatable or energy resources, the Forest Service controls leasing and mining of saleable materials on their lands. For small deposits, surface damage is considered minimal. However, for commercial developments or other extractions where a pit is developed, the Forest Service requires development of a pit plan, including a reclamation plan with bonding. The DEQ regulates most mining operations at the State level.

Most States have a program (for example, land quality division) that regulates surface mining. Depending on the size and type of the proposed mining project, the State DEQ may engage other State resource agencies to provide recommendations for protecting surface resources, including wildlife and habitat, which are incorporated into the permit to mine. Requirements for commercial mine development vary by State, but most have regulations regarding postmining surface stabilization and reclamation.

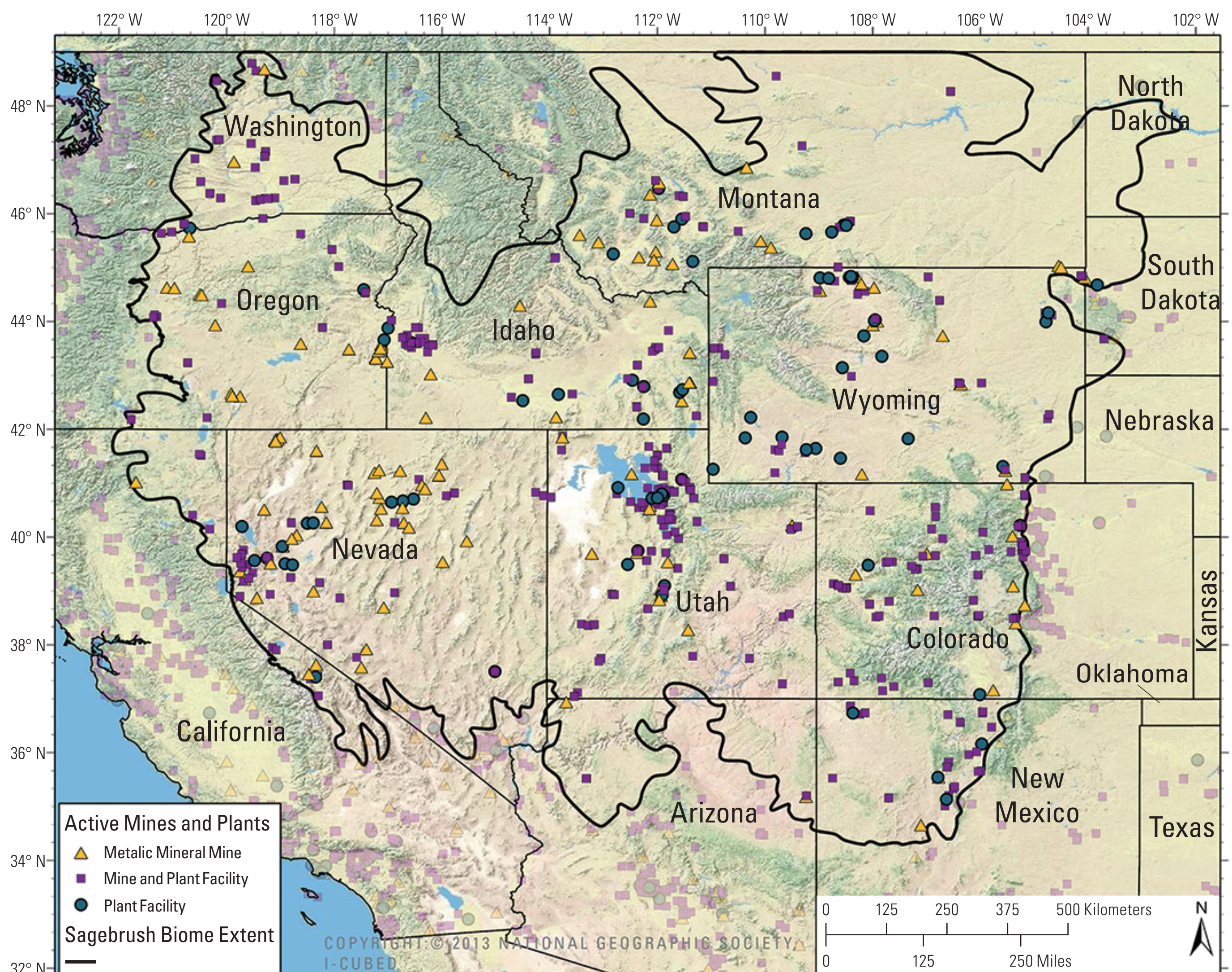

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Figure O3. Active locatable mines and mineral plants within the extent of the sagebrush (*Artemisia* spp.) biome (U.S. Geological Survey, 2018a).

Nonmining Energy Development

Nonmining energy development—such as drilling for oil or geothermal resources—results in different potential impacts to sagebrush and sagebrush-associated species. With the exception of renewable energy, energy sources retrieved primarily through drilling are governed by similar regulations described above for mining as these resources are typically considered leasable minerals (table O2).

Overview of Nonmining Energy Development Impacts to Sagebrush and Sagebrush-Associated Wildlife Species Common to Most Drilling Activities

In general, infrastructure (for example, processing facilities, and roads) and human activities (for example, presence and traffic) associated with oil and gas development (including coal-bed methane, oil shale, and tar sands) have similar impacts to the sagebrush ecosystem and wildlife as described for mining. Construction of oil and gas wells results in the direct loss of sagebrush, but impacts have negative consequences at larger scales than an individual well pad and after drilling is complete. These include habitat fragmentation and alteration because of road and pipeline construction and changes in wildlife behavior (Northrup and Wittemyer, 2012). Infrastructure-supporting drilling activities (for example, well pads, access roads, and staging areas) also remove and fragment sagebrush and associated vegetation, create opportunities for the spread of invasive plant species, provide increased opportunities for some predators (such as common ravens and red fox (*Vulpes vulpes*), increase fugitive dust, and potentially affect water quality. Drilling activities can result in increased noise, potentially interfering with vocalizations of sagebrush wildlife species. Direct mortality may also occur if individuals of a given species are not mobile enough to avoid or move from the location of infrastructure (for example, reptiles, amphibians, small mammals, or less mobile life stages such as juveniles of many taxa) or are struck by vehicles traveling to and from drilling and other appurtenant facilities. Displacement of individuals or populations from a drilling location can occur as a result of habitat loss, noise, and an increase in human presence and activity.

The impacts of drilling activities for energy resources—particularly oil, natural gas, and coal-bed methane—on some sagebrush-associated species are better studied than impacts associated with mining have been. A number of studies indicate that activities associated with oil and gas development have significant effects on greater sage-grouse and can result in reduced fecundity (Lyon and Anderson, 2003; Holloran and others, 2010; Fedy and others, 2014), reduced recruitment (Holloran and others, 2010) and survival (Aldridge and Boyce, 2007; Holloran and others, 2010), direct avoidance across life stages (Carpenter and others, 2010; Holloran and others, 2010), declines in lek attendance and population trends (Doherty and others, 2010b; Harju and others, 2010; Blickley and others, 2012a; Hess and Beck, 2012; Taylor, R.L., and others, 2013; Gregory and Beck, 2014; Green and others, 2017), and localized extirpations of populations (Aldridge and Boyce, 2007; Walker and others, 2007; Duncan, 2010; Harju and others, 2010; Gregory and Beck, 2014; Green and others, 2017).

Associated infrastructure (for example, roads, pipelines, storage facilities, and transmission lines) decreases the effectiveness of habitat for greater sage-grouse (Braun and others, 2002; Lyon and Anderson, 2003; Doherty and others, 2008; Dinkins and others, 2014a, 2014b; LeBeau and others, 2014; Kirol and others, 2015a). Sage-grouse females with successful nests located their nests farther from roads in oil and gas fields than females with unsuccessful nests (Lyon and Anderson, 2003). Transmission towers may provide perches and nesting structures for raptors or common ravens and result in increased densities of these predators (Borell, 1939; Beck and others, 2006; Messmer and others, 2013; Coates and others, 2014b, 2014c; Howe and others, 2014). Proximity to distribution and transmission lines is related to lower adult female survival for greater sage-grouse, which was most likely related to increases in raptors (Dinkins and others, 2014a).

In Nevada, proximity to a transmission line lowered multiple demographic rates for sage-grouse over a 10-year period (Gibson and others, 2018). Also, reservoirs created to hold water produced from energy development increased prevalence of West Nile virus and increased abundance of mesocarnivores, both of which can cause declines in greater sage-grouse populations (Taylor, R.L., and others, 2013). Populations can be affected at large distances from development, with lek attendance decreasing with increasing well density as far as 6.4 kilometers (km; 4 miles [mi]) away (Taylor, R.L., and others, 2013; Green and others, 2017). Population responses may lag development by as much as 4–10 years (Harju and others, 2010; Green and others, 2017).

Many other species—including large mammals—show avoidance of development by altering their home ranges, as well as movements and behaviors, in response to human activity (Sawyer and others, 2009b; Lendrum and others, 2012; Northrup, 2015). Specifically, mule deer (*Odocoileus hemionus*) avoid resources within approximately 4 km (2.5 mi) of well pads, shifting habitat use to less suitable areas in response to increased development (Sawyer and others, 2006). In all of these cases, indirect or functional habitat loss (Aldridge and Boyce, 2007) from energy development can have large impacts on resource availability and fitness for these species.

Disturbance from oil and gas development may also cause local declines in sagebrush-associated avian populations (Gilbert and Chalfoun, 2011) and may cause the birds to modify their behavior in response to increased noise pollution or other disturbances (Pitman and others, 2005; Francis and others, 2011). Nest survival and nest success of sagebrush-obligate songbirds decreases with increasing habitat loss because of energy development (Hethcoat and

Chalfoun, 2015). Habitat changes associated with oil and gas development result in a decrease in local occupancy for sagebrush sparrows (*Artemisiospiza nevadensis*) and sage thrashers (*Oreoscoptes montanus*; Mutter and others, 2015). Densities of Brewer's sparrow (*Spizella breweri*) and sagebrush sparrows decreased with increased well density in Wyoming (Gilbert and Chalfoun, 2011). Additionally, densities of sparrows and sage thrashers decreased near roads associated with development (Ingelfinger and Anderson, 2004). Reclamation within two natural gas fields in western Wyoming resulted in increased nest predation of sagebrush-obligate birds owing to the attraction of rodents to the reseeded areas (Sanders and Chalfoun, 2018). The most effective mitigation for the effects of natural gas development on sagebrush songbirds would be the minimization of sagebrush habitat conversion during new construction and effectively restoring disturbed lands back to habitat that consists of intact big sagebrush (*A. tridentata*) and understories of locally native grasses and forbs.

Development associated with oil and natural gas extraction continues to increase and pose threats to sagebrush habitats across the biome. Current and pending energy developments have been estimated to alter nearly 100 million ha (247 million acres) of wildlife habitat in North America and, as of 2011, it was estimated that over 81 percent of Federal lands were leased for oil and gas development (Naugle, 2011). Roughly 8 percent of sagebrush habitats are directly affected across the biome, with >20 percent of sagebrush habitats affected by oil and gas development in the Rocky Mountain area (Chambers and others, 2017a). This does not account for disturbances associated with infrastructure and development such as roads, powerlines, facilities and pipelines, or the functional loss of habitat (Aldridge and Boyce, 2007). Collectively, these disturbances can disrupt ecosystem processes (Francis and others, 2012) and thus can disrupt the overall functioning of the sagebrush ecosystem, yet such effects on ecosystem processes are not well studied.

Renewable Energy Development

Concerns over the continued use of nonrenewable energy resources have generated interest in the development of renewable energy, particularly wind, solar, geothermal, and biofuels from renewable resources. Most renewable energy development does not result in the degradation of air and water quality or contribute to greenhouse gas emissions, making it an attractive alternative to other energy sources (Johnson and Stephens, 2011). In 2008, the U.S. Department of Energy (DOE) released a report that envisioned wind energy would provide 20 percent of the electricity generated in the United States (U.S. Department of Energy, 2008). Similarly, the DOE released the SunShot Vision Study in 2012, seeking to reduce the costs of solar energy development with a goal of meeting 14 percent of electricity needs by 2030 and 27 percent by 2050 (U.S. Department of Energy, 2012). Both documents support the intent to quickly grow renewable energy development within the United States.

Biofuels—primarily ethanol production from corn (*Zea mays*) or other grains and potentially grasses (Johnson and Stephens, 2011)—can result in the conversion of sagebrush ecosystems to grow these resources. The potential impact of continued conversion on greater sage-grouse was examined by the FWS by comparing cropland suitability models (based on soil and climate data) with sage-grouse population indices. The overlap of potential future conversion with greater sage-grouse was low, with the greatest conversion potential in the northeast and northwest distribution of the species' range (U.S. Department of the Interior, 2015c). Although the analyses did not encompass the entire sagebrush ecosystem, it is likely that agricultural conversion for the purposes of producing biofuels is low.

Wind Energy

Construction of wind farms requires clearing areas to create access roads and turbine pads. Additional roads may be cleared between turbine pads to allow access by cranes needed to erect turbines. Collector electrical cables and circuits are buried, typically along access roads. All these activities require clearing of topsoil, compaction of subsoils, and gravel deposition for the roads, resulting in the direct loss and fragmentation of sagebrush habitats. While reclamation is typically required, turbine pads and access roads are maintained to be free of vegetation to identify where underground structures are located and for maintenance purposes. Wind farms vary in size based on the number of turbines needed (or feasible) within the wind farm boundaries, size of turbines, topography, wind conditions, and distance necessary between turbines to minimize the impacts of wake created by each individual turbine on the adjacent turbines. Compared to most other forms of energy production, wind-energy development has a larger terrestrial footprint per unit of energy (Kiesecker and others, 2011).

To meet the DOE goal of providing 20 percent of electricity by wind by 2030, 241 gigawatts of onshore wind will be needed (estimate based off 2011 technology). That level of development will require 5 million ha (12.3 million acres) of land (an area roughly the size of Florida), along with 18,000 km (11,187 mi) of new transmission lines (Kiesecker and others, 2011). Many States do not have sufficient wind resources to meet DOE goals, so wind-energy development is concentrated in those that do, including States in the sagebrush biome (California, Idaho, Montana, Oregon, South Dakota, Washington, and Wyoming; Kiesecker and others, 2011), with Montana and Wyoming having the highest potential (Copeland and others, 2011).

The BLM has authority to manage facilities for generation, transmission, and distribution of electric energy under FLPMA (U.S. Department of the Interior, 2016).

The FLPMA also allows for the development of facilities of transmission and distribution of electricity generated by wind and included provisions for granting rights-of-way for access to wind development on private lands. The BLM has identified designated leasing areas and development-focus areas (for both wind and solar energy), but development may also occur outside these areas. The BLM manages more than 8 million ha (20 million acres) of public lands with wind-energy potential in 11 western States, including most States within the sagebrush ecosystem. Wind energy on Forest Service lands are governed by special use regulations (U.S. Department of Agriculture, 2011). Both the Forest Service and BLM have specific guidance on wildlife monitoring on wind-energy sites before, during, and after construction. The BLM can apply payment and bonding requirements to wind (and solar) facilities (U.S. Department of the Interior, 2016).

Wind-facility siting authorities are either under local government control or are a mixture of State and local government control. Siting of wind facilities in local control States (Arizona, Idaho, Montana, and Utah) falls entirely to local authorities, such as counties and public utility commissions. In mixed States (California, Colorado, Nevada, Oregon, Washington, and Wyoming), wind facility siting requires approval by State or local government bodies and can also include public utility commissions or siting councils (National Conference of State Legislatures, 2016). State and local considerations for wind-facility siting primarily focus on setbacks for safety, aesthetic, and other social reasons (such as minimizing noise for private property owners) but can also consider impacts to natural resources, such as wildlife.

Direct impacts from wind-facility development include those associated with other types of energy development (loss of animals that cannot avoid construction equipment) and mortality as a result of collision with wind turbine blades. Raptor fatality rates from collisions with turbines were high on farms with early technology turbines, where turbines have lattice support lines, and electrical lines are not buried. These features create perching areas and increase the potential of raptors using an area for foraging (Johnson and Stephens, 2011). Most common raptor fatalities on these older farms included golden eagles (*Aquila chrysaetos*), American kestrels (*Falco sparverius*), red-tailed hawks (*Buteo jamaicensis*), and burrowing owls (*Athene cuncularia*; Johnson and Stephens, 2011). More recent developments have tubular steel towers with buried electrical line, reducing perching opportunities, and therefore fewer raptor mortalities.

Raptor mortality rates summarized in 2011 averaged 0.19 per megawatt (MW) per year for all wind developments (including nonsagebrush States). Monitoring studies on 21 modern facilities in western North America reported 1,247 bird fatalities: 19 percent raptors, 59 percent passerines (primarily horned larks [*Eremophila alpestris*], western meadowlarks [*Sturnella neglecta*], and kinglets [*Regulus* spp.]), and 10 percent upland game birds (primarily nonnative species such as ring-necked pheasants [*Phasianus colchicus*] and chukars [*Alectoris chukar*]). Waterbirds, doves, shorebirds, nighthawks, swifts, and pigeons made up the rest of the mortalities (for more detail see Johnson and Stephens, 2011). Not all birds killed by wind turbines are found because of being taken by scavengers and difficulty in detection.

Bat mortality is also associated with wind turbines, with fatalities from 21 facilities in western North America averaging 2.13 per MW per year. Bat mortalities result from collision, but the majority are thought to be from barotraumas presumably caused by rapid air pressure reduction near moving turbine blades (Johnson and Stephens, 2011). Most reported bat mortalities are tree-roosting species (such as hoary bats [*Lasiurus cinereus*]), but species that forage in open areas are also impacted (for example, little brown bat [*Myotis lucifugus*]; Johnson and Stephens, 2011). Collision mortality of both bats and birds is well-documented at most wind-energy facilities, but population-level effects have not been well studied (Johnson and Stephens, 2011). Frick and others (2017) found that migratory bat species such as hoary bats could face substantial reduction in population size owing to wind-energy development, and concluded that if populations are to be sustained, conservation measures to reduce turbine collisions should be implemented quickly.

The indirect influence of wind-energy development on sagebrush-associated species is not yet well understood, because of limited study. Greater sage-grouse in southeastern Wyoming decreased their summer and brood-rearing habitat selection as the percentage of surface disturbance associated with a new wind-energy facility increased (LeBeau and others, 2017a). Moreover, sage-grouse nest and brood failures increased with proximity to wind-energy infrastructure (LeBeau and others, 2014). Sage-grouse female survival did not vary in relation to wind-energy infrastructure (LeBeau and others, 2014). However, lek counts were not affected until 3-years postdevelopment. At that point, counts decreased by 56 percent at leks near (<1.5 km; 0.9 mi) the wind farm compared to those farther away (>1.5 km; 0.9 mi; LeBeau and others, 2017b). Such lag effects have also been observed in response to oil and gas development, where declines were not observed until 4 years after construction (Naugle and others, 2011). Other grouse species have declined in areas of wind-power development (Johnson and Stephens, 2011).

For passerine birds, no systematic study of wind-farm effects has been conducted within sagebrush, though one study in southeastern Wyoming in mixed-grass prairie suggested mixed effects of turbine density and proximity on two species of grassland birds (horned lark and thick-billed longspur [*Rhynchophanes mccownii*]; Mahoney and Chalfoun, 2016). The size of horned lark nestlings decreased slightly with surrounding turbine density (Mahoney and Chalfoun, 2016). Passerines in grassland ecosystems were also displaced by wind farms, as evidenced by higher densities of birds at distances of 50–200 m (164–656 ft) from turbines (Johnson and Stephens, 2011). Shorebirds and waterfowl appear to be similarly affected (Johnson and Stephens, 2011). There are few studies examining displacement of raptors from wind facilities. In all but one, raptors do not appear to avoid wind facilities for either moving or nesting (Johnson and Stephens, 2011). Pronghorn (*Antilocapra americana*) and elk (*Cervus canadensis*) did not appear to avoid wind development on two facilities where data were collected. However, long-term data are lacking (Johnson and Stephens, 2011). Effects on other taxonomic groups such as small mammals and herpetofauna are unknown.

Geothermal

Geothermal energy is replenished by heat sources deep in the Earth and generates electricity with minimal carbon emissions and is abundant in the western United States. Geothermal steam is identified as a leaseable mineral (table O1) and falls under the same Federal regulations as oil and gas. The BLM has authority to manage geothermal leasing on approximately 9.7 million ha (240 million acres) of public lands (including 42 million ha [104 million acres] of Forest Service lands). To develop a geothermal project on Federal land, a lease allowing access to and use of the federally owned resource is required. The lease does not grant the associated water rights, as those are typically authorized through a State agency. The BLM issues geothermal leases through either a competitive auction of lands thought to have sufficient geothermal resources to warrant exploration—which are nominated by the general public—or through a noncompetitive process. Competitive auctions are held at least every 2 years in States where nominated geothermal leases are pending and may be held in conjunction with BLM-managed oil, gas, and coal lease auctions. Leasing may also occur through a noncompetitive mechanism if lands did not receive a bid in previous competitive auctions. Lands are nominated solely for direct use of geothermal heat (that is, not for electricity generation), or lands are subject to mining claims or operations. Noncompetitive leases typically are for areas with less known geothermal resource attributes than those of competitively leased lands.

Geothermal resources can be retrieved as dry steam, flash steam, or through a binary cycle. All methods require drilling production wells to access the steam or hot water and injection wells to return the cooled water to the subsurface reservoir (Union of Concerned Scientists, 2014). Surface facilities are similar among all geothermal developments, with buildings housing turbines, heat exchangers, and generators, above ground pipelines for water delivery (from production wells and to injection wells), and transmission lines for energy delivery to consumers (U.S. Department of Energy, 2019).

There is only one dry steam plant in the continental United States located at “The Geysers” in northern California. This source uses steam piped directly from underground wells to the power plant where it is directed into a turbine/generator unit. Flash steam power plants use geothermal reservoirs of very hot water (>182 degrees Celsius [°C; >360 degrees Fahrenheit {°F}]) and are the most common. The hot water flows up through wells under its own pressure whereupon the hot water boils into steam. The steam is then used to power a turbine/generator. Binary steam plants operate on water between 107 and 182 °C (225 and 360 °F). The heat from the hot water is used to boil a working fluid that has a lower boiling point. The fluid is vaporized in a heat exchanger which turns a turbine.

Geothermal development is increasing owing to the National Energy Policy of 2001, which encouraged development of alternative energy sources. The Energy Policy Act of 2005 (42 U.S.C. 211) directed the DOI to approve 10,000 MW of nonhydrorenewable electrical generation within 10 years of the date of enactment. Area leased per year on BLM-managed lands for geothermal energy has increased since 2001, with the highest total leased areas in the Northern Great Basin, Snake River Plain, and Central Great Basin (see Knick and others, 2011, table 12.18). Conventional geothermal resources in the sagebrush biome are located in Arizona, California, Colorado, Idaho, Montana, Nevada, New Mexico, Oregon, Utah, Washington, and Wyoming (fig. O4). These States contained all 241 identified moderate-temperature (ranges from 90 to 150 °C [194–302 °F]) and high-temperature (>150 °C [302 °F]) geothermal systems located on private or accessible public lands (Williams and others, 2008). The majority of installed and used power production capacity in the sagebrush biome is produced from geothermal plants located in California, Idaho, Nevada, Oregon, and Utah (Matek, 2016). As of March 2018, the BLM had approved 50 geothermal projects with 1,648 MW of total installed capacity, enough to power 500,000 homes. Since the current competitive auction system was implemented in 2007, 11 lease auctions have been held in various western States.

The associated infrastructure is very similar to other energy development facilities, and includes transmission lines, improved roads, fencing, and storage facilities. In addition, the noise generated from geothermal facilities can be substantial and otherwise alters the normal ambient levels within the surrounding soundscape. Such noise levels can mask sage-grouse vocalizations during the mating period (Blickley and Patricelli, 2012), increase stress hormone levels that may lead sage-grouse to avoid otherwise suitable habitat (Blickley and others, 2012b), and ultimately reduce lek attendance (Blickley and others, 2012a). Further research is needed with respect to the overall impacts of geothermal development on sagebrush-obligate species; however, ongoing research is currently taking place in Nevada on the effects of geothermal development on greater sage-grouse.

Solar Energy Development

Similar to wind-energy development, BLM has authority to manage solar facilities under FLPMA and the associated right-of-way regulations (U.S. Department of the Interior, 2016). Leases have a 30-year life. Additionally, the BLM manages this resource in Arizona, California, Colorado, Nevada, New Mexico, and Utah under the Solar Energy Program, which identifies areas that can be leased for development and exclusion areas. The Solar Energy Program established a competitive leasing program for solar energy zones, although leasing may also occur outside these areas (U.S. Department of the Interior, 2016). The BLM can apply payment and bonding requirements to solar facilities. Solar energy development on Forest Service lands requires a special-use permit, and potentially a transmission right of way to connect the solar project to the energy grid, which are governed by special-use regulations (U.S. Department of Agriculture, 2011). In addition to compliance with their organic acts, both the Forest Service and BLM have specific guidance on wildlife monitoring on energy sites before, during, and after construction.

Solar power generation facilities that are likely to be developed for utility-scale capture of solar energy (that is, greater than or equal to 20 MW electricity that will be delivered into the electricity transmission grid) in the United States over the next 20 years include concentrating solar power; parabolic trough, power tower, and dish engine systems, and photovoltaic systems. The main part that all these technologies have in common is a large solar field where solar collectors capture the sun's energy. In the parabolic trough and power tower systems, the energy is concentrated in a heat transfer fluid which is transferred to a power block where steam-powered turbine systems generate electricity using technology similar to that used in fossil fuel–fired power plants. In contrast, the dish engine and photovoltaic systems

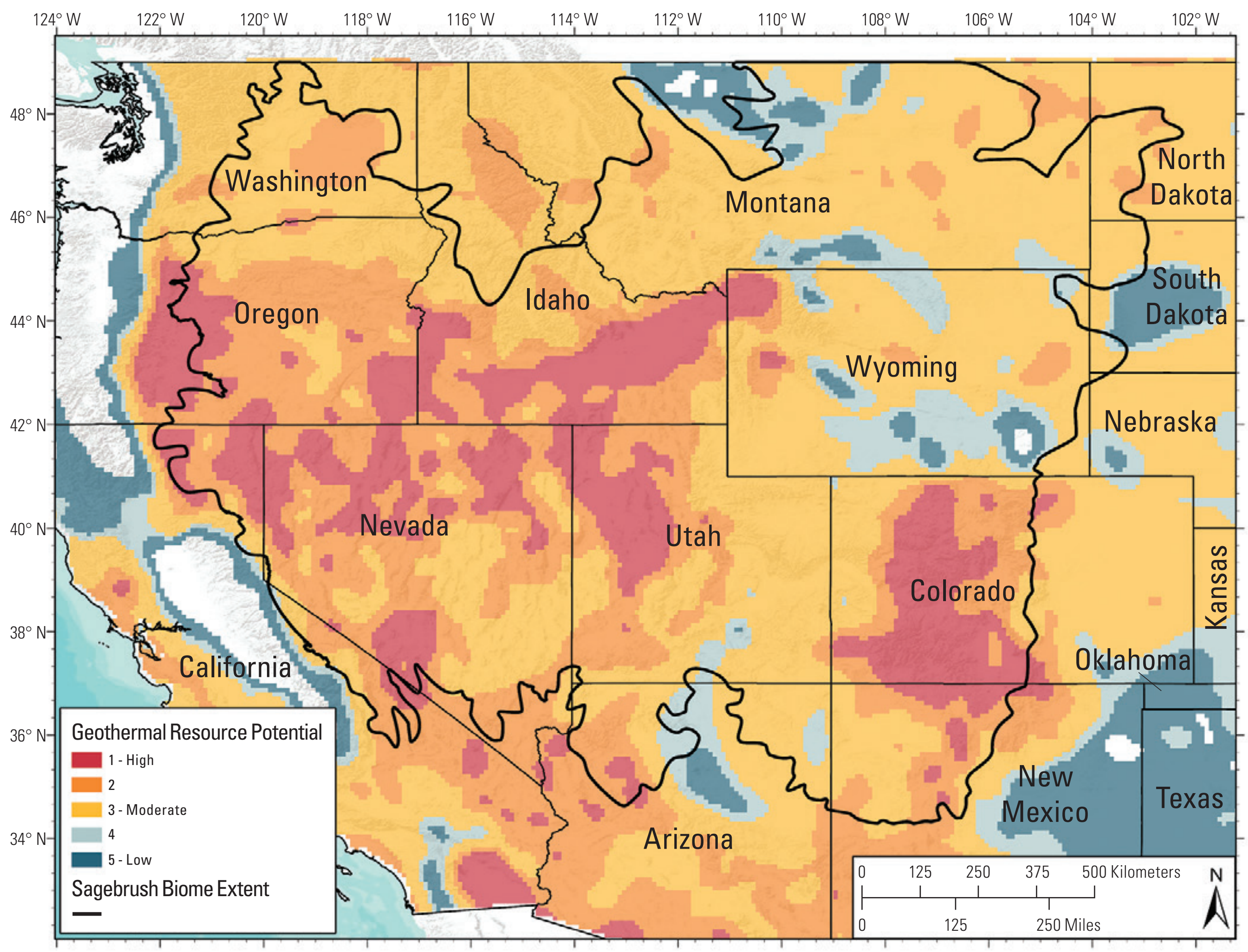

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Figure O4. Geothermal resource potential within the sagebrush (*Artemisia* spp.) biome extent (National Renewable Energy Laboratory, 2019).

are composed of many individual units or modules that generate electricity directly and whose output is combined; these systems do not use a central power block. Solar facilities are likely to have an operational lifetime of at least 30 years.

The primary environmental concerns associated with solar power generation include the large land area required for solar facilities and water consumption. Concentrating solar power systems generally require 2 to 4 ha (5 to 10 acres) to produce 1 MW, and photovoltaic systems require around 4 ha (10 acres) per MW. Additional impacts of solar power generation include access roads and transmission lines. Solar power potential is concentrated in southern States of the sagebrush biome (Colorado, Nevada, Utah, and southwest Wyoming; see Manier and others, 2013, fig. 20). Further information about potential impacts, as well as a detailed discussion of the technology required for generation of solar-based electricity, can be found in the "Draft Programmatic Environmental Impact Statement (PEIS) for Solar Energy Development in Six Southwestern States" (https://solareis.anl.gov).

Information on the impacts of solar energy developments on sagebrush ecosystems and associated wildlife is limited. There are two parabolic facilities in the sagebrush ecosystem (Tonopah and Fallon, Nevada). Parabolic solar energy systems have been implicated in the direct mortality of bats and birds when they fly through the high-temperature solar beams. Insects are also attracted to the mirrors used in the parabolic systems, increasing the attraction for many birds and bats. Larger birds, including raptors, may be singed or burned by the solar beams. The impact to these species is not well-documented as surveys for wildlife are not required outside the solar farm boundaries (https://www.bv.com/perspectives/impact-solar-energy-wildlife-emerging-environmental-issue). Population impacts to affected species are not known, but mortality rates for smaller species can be significant (Walston and others, 2016). Large solar fields appear as lakes to migrating waterbirds, resulting in collisions causing death or injury and in stranding those birds that cannot take off from nonwater surfaces. There are several solar projects already operating or under development within the sagebrush ecosystem. A comprehensive database and mapper are available at https://www.seia.org/research-resources/major-solar-projects-list.

Reclamation Requirements for Mining and Energy Development Activities

Reclamation is required for most mining and energy development activities, via State or Federal regulations (Boyce, 2002). Restoring lands disturbed by mining and energy development to preexisting habitats can be challenging depending on seed availability and sourcing, precipitation patterns, and the resulting change in topography from resource removal and overburden contouring that may affect microclimate habitats (Bonta and others, 1997; Boyce, 2002). A return to predisturbance habitat conditions following reclamation in arid sagebrush environments may require several decades (Braun, 1998) or may never occur without substantial funding and intensive restoration techniques. Habitat for sagebrush-dependent or -associated species may be lost or made unavailable during development, during reclamation activities, and for the time period until recovery. If habitat is removed for extended periods, movement patterns may be disrupted, and there may be complete extirpation of local populations of sagebrush-associated species (Hayden-Wing Associates, 1983).

For mining, the degree of reclamation varies by mine type, the amount of disturbance associated with mining activities, and postmining land uses. Reclamation requirements for supporting infrastructure (roads, transmission lines, and rail lines) are typically addressed separately (chap. P, this volume). Initial reclamation activities focus on stabilizing soils to prevent soil loss and frequently use exotic grasses and legumes as they readily establish on bare soils and are less expensive to acquire (Burger, 2011). Use of exotic plant species for reclamation may not re-create plant composition and structure needed for wildlife habitat and may result in the spread of these species into adjoining intact habitats, thereby reducing their suitability for sagebrush wildlife species (Boyce, 2002). Reclamation activities for other energy and nonfuel mineral development face the same challenges as mining reclamation.

Reclamation of lands mined for coal occurs according to an approved postmine land use (for example, livestock grazing or wildlife habitat) and is required to be completed contemporaneously with mining activity. Successful reclamation of lands mined for coal often creates a cool-season, grass-dominated system where natural rugged topography is smoothed and recontoured to meet Federal or State regulations, which may directly impact vegetation diversity and surface water movement. In Wyoming (which produces the most coal in the sagebrush ecosystem; table O1), coal mine operators are required to re-establish sagebrush (or the dominant premine shrub[s]) at a minimum aerial extent and density of one shrub per square meter over 20 percent of the affected mine lands. Other western coal-producing States have shrub density requirements for reclamation on lands being returned to wildlife habitat, and a few States (Colorado, Montana, New Mexico, and Wyoming) also have shrub density requirements for reclamation on native rangelands. The re-establishment of sagebrush on reclaimed surface coal mines is costly, and it may take decades to return these areas to a functional sagebrush ecosystem.

Wildlife response to reclamation activities is poorly understood, with most studies focused on birds (Buehler and Percy, 2012). However, these studies rarely examine demographic data (but see Boisvert, 2002). Postmining restoration to grasslands may contribute to grassland bird conservation (Galligan and others, 2006), but such reclamation has demonstrated limited benefits for sagebrush wildlife species. The proximity of source populations after reclamation will influence small mammal and amphibian species that will recolonize a mine site (Buehler and Percy, 2012). Retention, and not reclamation, of portals for subsurface mining may provide habitat for bats (Buehler and Percy, 2012). Reconstruction of mesic areas or retention of created wetlands because of mining activity may provide wetland habitat for amphibians that was not previously available if water quality is sufficient to support these species (Buehler and Percy, 2012). Some species benefit from reclamation despite initial displacement from energy development activities. Columbian sharp-tailed grouse (*Tympanuchus phasianellus columbianus*) experienced higher lek densities, larger leks, increased clutch size, increased nesting success, and increased chick survival on reclaimed mine lands within mountain shrub communities in Colorado (Hoffman, 2001; Boisvert, 2002; Collins, 2004).

Current Federal and State Regulatory and Mitigation Approaches

State and Federal agencies, and some counties, have developed a variety of voluntary and regulatory approaches to protect public health and the environment when permitting land use activities. Within sagebrush habitats, emphasis has been placed on reducing disturbance to sage-grouse or their habitats. These voluntary and regulatory approaches, and the efficacy of sage-grouse as a conservation umbrella, are reviewed in chap. Q (this volume).

Several efforts have assessed conservation actions adopted in the 2015 BLM and Forest Service land use plan amendments and State-level protections to estimate how effective the conservation actions may be at limiting effects of mining to sage-grouse. Current conservation actions in management zones (MZs; derived by Stiver and others, 2006) I and II reduced the future exposure of greater sage-grouse to oil and gas development by about one-third and two-thirds, respectively (Juliusson and Doherty, 2017), but the conservation measures are not expected to reverse the declines where active oil and gas operations are present (see Green and others, 2017). The probability of lek collapse in Wyoming core areas was positively associated with development density outside the core area, and the risk of lek collapse decreased as distance from the edge increased up to a distance of 4.83 km (3 mi) from the core area boundary (Spence and others, 2017). The rate of decline was minimized for leks that were more than 4.83 km (3 mi) inside a core area, and oil and gas well densities inside core areas were unrelated to the probability of lek collapse.

Winter habitats outside of core areas support core area sage-grouse populations, and sage-grouse may use those habitats for a longer period than is identified by current regulatory restrictions in Wyoming. These results, and those presented by Gamo and Beck (2017), support the conclusion that overall, the Wyoming Governor's Executive Order for Greater Sage-Grouse Core Area Protection is helping safeguard essential sage-grouse habitats at the statewide scale, though populations are still likely to decline (see below, and Heinrichs and others, 2019). Spatially explicit simulation models in Wyoming tracking sage-grouse resource conditions indicate that, with current energy development projections through 2050 and the core areas strategy in place, sage-grouse are projected to decline 40 percent owing to energy development alone (Heinrichs and others, 2019).

Voluntary Conservation Actions

Several coal operators in northeast Wyoming are members of the Thunder Basin Grasslands Prairie Ecosystem Association (TBGPEA) and have enrolled their property in TBGPEA's Candidate Conservation Agreement with Assurances (CCAA; approved by the FWS in 2017 and in effect for 30 years) for eight species in northeast Wyoming, four of which are sagebrush-associated: greater sage-grouse, sagebrush sparrow, Brewer's sparrow, and sage thrasher (Thunder Basin Grassland Prairie Ecosystem Association and U.S. Fish and Wildlife Service, 2017). The CCAA includes 110 conservation measures for sagebrush-obligate species and covers Wyoming's five northeastern counties (approximately 5.3 million ha [13.2 million acres]) and two peripheral locations in Montana (approximately 97,000 ha [239,700 acres]). If implemented, the conservation measures would reduce sagebrush habitat fragmentation and conversion, reduce disease transmission and predation, and address the inadequacy of existing regulatory mechanisms across a landscape that is primarily private land and split estate. The measures would further address other factors that impact sagebrush-associated species, such as drought. Partnerships between coal operators in northeast Wyoming and TBGPEA have resulted in extensive sagebrush mapping, vegetation monitoring, and sage-grouse data collection with the objective of developing a sage-grouse resource selection function model for northeast Wyoming.

In Nevada, Barrick Gold of North America, Inc., established a bank enabling agreement (BEA) with the DOI, working through the FWS and the BLM. The agreement acts as a mechanism for the establishment, use, operation and maintenance of the "bank" to compensate for impacts to greater sage-grouse and sagebrush ecosystems resulting from Barrick's proposed mining activities. Project plans have been developed that describe the conservation actions, methods, monitoring, financial assurances, and other requirements as specified in the BEA. The project area includes both private and public lands in northeastern Nevada that encompass approximately 96,300 ha (238,000 acres).

Best Management Practices to Avoid, Minimize, or Mitigate Impacts of Mining and Energy

At local scales, recent research is consistent with past findings that the implementation of certain mitigation techniques or design features for oil and gas operations may be beneficial in reducing, but not eliminating, adverse effects to sage-grouse (Fedy and others, 2015; Holloran and others, 2015; Kirol and others, 2015b; Garman, 2017). Increased sagebrush land cover and minimization of disturbance are consistent characteristics of high-quality sage-grouse habitat. The strength and consistency of these relationships, from local studies to across the species range, provides managers with information and a set of tools for understanding the potential effect of management actions on habitat conditions for sage-grouse. For example, analysis of mitigation measures for a single energy development field (Kirol and others, 2015b) and development of source-sink maps that inform the prioritization of areas for conservation (Kirol and others, 2015a; Heinrichs and others, 2017; Kane and others, 2017) identify opportunities to maintain local sage-grouse populations and provide opportunities for colonization of reclaimed sites after energy extraction. These models have an additional advantage of being tuned to smaller extents that generally allow better spatial predictions.

New habitat-mapping tools developed at local and landscape scales can often provide important insights that can be expanded (for example, Homer and others, 2015), and potentially help with restoration of habitats after disturbance owing to energy development (Monroe and others, 2020). Areas where there is an alignment of habitat selection, nest-site selection, and reproductive success can help identify important locations across the landscape and may help with the targeting of conservation actions and mapping critical habitats (Aldridge and Boyce, 2007; Gibson and others, 2016).

Mining and Energy—Key Gaps

The global demand for energy resources indicates that sagebrush ecosystems will continue to be affected by energy development. A better understanding of how energy development ultimately affects the long-term functioning of sagebrush ecosystems and the persistence of associated wildlife species is needed. Efforts should be made to minimize the associated disturbances and to develop technologies to reduce impacts. Restoration of sagebrush communities affected by mining and energy development is critical. Current practices still facilitate the presence of nonnative grasses and forbs (and species that are not typically part of the understories of undisturbed sagebrush), which can alter habitat quality for sagebrush-associated wildlife. While in some industries (for example, coal) an extensive amount of wildlife and vegetation data is collected over many contiguous years, it is often not used to extrapolate long-term trends or impacts.

Cumulative effects of site-level disturbances characteristic of mining and energy development (for example, individual well pads, coal mines, gravel pits, and more) and sagebrush habitat functionality at larger spatial scales are critical gaps in the information necessary for effective management of the ecosystem. Habitat protection measures tend to be implemented at site levels (for example, infrastructure density and surface disturbance levels are minimized within a relatively short distance of the analyzed impact) not taking into account the potential larger scale ramifications of the impact either individually (for example, impact to a limiting seasonal habitat; impact to an important travel/migration corridor) or in combination with other disturbances on the landscape. The disconnect in spatial scales between approaches taken to assessing cumulative impacts in NEPA and in the development actions actually pursued as a result of NEPA currently ensures cumulative effects are not addressed in conservation and management of the sagebrush biome.

Chapter P. Land Use and Development

By Adrian P. Monroe,[1] Victoria Drietz,[2] Heather H. McPherron,[3] and Catherine S. Wightman[4]

Executive Summary

With the population of the United States over 300 million people and growing, the demands on our lands are increasing. Western landscapes are targeted, in part, to help meet increasing needs for energy, agricultural products, new housing, and recreational opportunities. However, changes to the landscape resulting from human land use and development can alter wildlife, invertebrate, and plant communities. For example, conversion of native sagebrush (*Artemisia* spp.) to cropland agriculture can result in direct habitat loss for sagebrush-obligate wildlife species, such as greater sage-grouse (*Centrocercus urophasianus*) and Gunnison sage-grouse (*C. minimus*), hereafter called sage-grouse, and Brewer's sparrow (*Spizella breweri*). However, alfalfa (*Medicago sativa*) and grain crops can provide foraging opportunities for mule deer (*Odocoileus hemionus*). Contemporary, well-managed grazing can foster productive rangeland for cattle and wildlife; however, poorly managed grazing leads to a reduction in grass cover and soil erosion and compaction. Tall structures, roads, and other infrastructure can fragment habitat, leading to avoidance by some species, such as ground-nesting birds, but can also provide additional perching habitat for sensitive species, such as golden eagles (*Aquila chrysaetos*). Residential development and recreational activities can introduce invasive plant species and domestic pets but provide housing for our growing human population.

Regulatory and voluntary mechanisms are being implemented across the sagebrush biome to help reduce negative impacts from human land use. Federal land-management agencies (U.S. Department of the Interior, Bureau of Land Management; U.S. Department of Agriculture, Forest Service) have established range condition targets to support sustainable grazing practices on public lands. State, Federal, and not-for-profit partners are providing voluntary protection mechanisms (for example, conservation easements) and cost-share opportunities (for example, the U.S. Department of Agriculture, Natural Resources Conservation Service, Environmental Quality Incentive Program) to help private landowners conserve and maintain resilient rangelands. Federal, State, and county entities are closing roads, constructing wildlife road crossing structures, and managing recreational activities to minimize human and wildlife conflicts. Mitigation programs are also active in many States to help offset adverse impacts from ongoing land use development, such as new pipelines, transmission lines, and cell towers.

Future decisions about land use in the sagebrush biome will need to continue to consider options to balance human needs and wildlife habitat impacts. Human development in rural areas of the sagebrush biome—such as additional cell towers for improved cell reception, electricity via transmission lines and poles, and fiber optic cables for internet communications—will impact sagebrush-associated species. However, strict avoidance of these activities is often not a viable solution for rural communities. The following summary of the impacts of land use change agents in the sagebrush biome can be used to help understand ecological tradeoffs in land use decisions, acknowledging that social implications will also need to be considered.

Introduction

Humans have used natural resources and affected landscapes throughout their evolutionary history. Land and natural resources shape economies, communities, and quality of life. This chapter discusses contemporary land use and development activities influencing the conservation and management of the sagebrush (*Artemisia* spp.) biome. The chapter is separated by discrete topics of cropland conversion, grazing management, infrastructure, subdivision development, and recreation. However, there is considerable spatial overlap among these change agents on the landscape. To be successful, conservation efforts will need to consider these impacts cumulatively, along with other stressors such as energy development and invasive species.

Conversion of Sagebrush to Croplands

Successive homestead acts (Homestead Act of 1862, Dominion Lands Act of 1872 [Canada] and Desert Land Act of 1877), beginning in 1862, encouraged settlement and development of the Midwest and West (including sagebrush-steppe) for agriculture (Knick and others, 2011). Approximately 10 percent of sagebrush-steppe was privatized and cultivated (West, 1996), typically at low elevations with deep, fertile soils (fig P1; Vander Haegen and others, 2000; Knick and Connelly, 2011a). Commodities markets and recent technological advances may lead to further conversion of sagebrush that is less productive for agriculture. For example,

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between 2008 and 2012, over 202,000 hectares (ha; 500,000 acres) of shrubland in the United States were converted to cropland such as alfalfa (*Medicago sativa*) and wheat (*Triticum* spp.) crops (Lark and others, 2015). Conversion of sagebrush ecosystems to agriculture is particularly consequential because once broken out, these lands usually remain under cultivation. Cultivated lands that are no longer in production may take more than 90 years to recover (Morris and others, 2011) or may be prevented from recovering through the use of herbicides.

The rate of ongoing conversion varies across the sagebrush biome, with the slightly wetter and more productive soils of eastern Washington and eastern Montana and Wyoming experiencing the most conversion (summarized in Chambers and others, 2017a). Mapping of conversion potential (based on climate, soils, and topography) by the U.S. Department of Agriculture, Natural Resources Conservation Service's (NRCS) Sage Grouse Initiative (SGI) for the eastern part of the biome has identified the prairies of Montana and the Dakotas and the San Luis Valley in south-central Colorado as the most vulnerable areas (Smith, J.T., and others, 2016; Natural Resources Conservation Service, 2019b). Roughly 91 percent of all rangelands in the Great Plains and 56 percent of rangelands in the western United States are in private ownership, and private rangelands are twice as productive, on average, as rangelands in the public domain (Robinson and others, 2019). Conservation efforts that help landowners maintain economic stability without converting native landscapes to croplands can have a significant impact on limiting additional conversion.

Primary impacts from the conversion of sagebrush to croplands stem from loss of sagebrush habitat, landscape fragmentation, and reductions in sagebrush patch size. Cropland conversion can influence wildlife even if much of the landscape

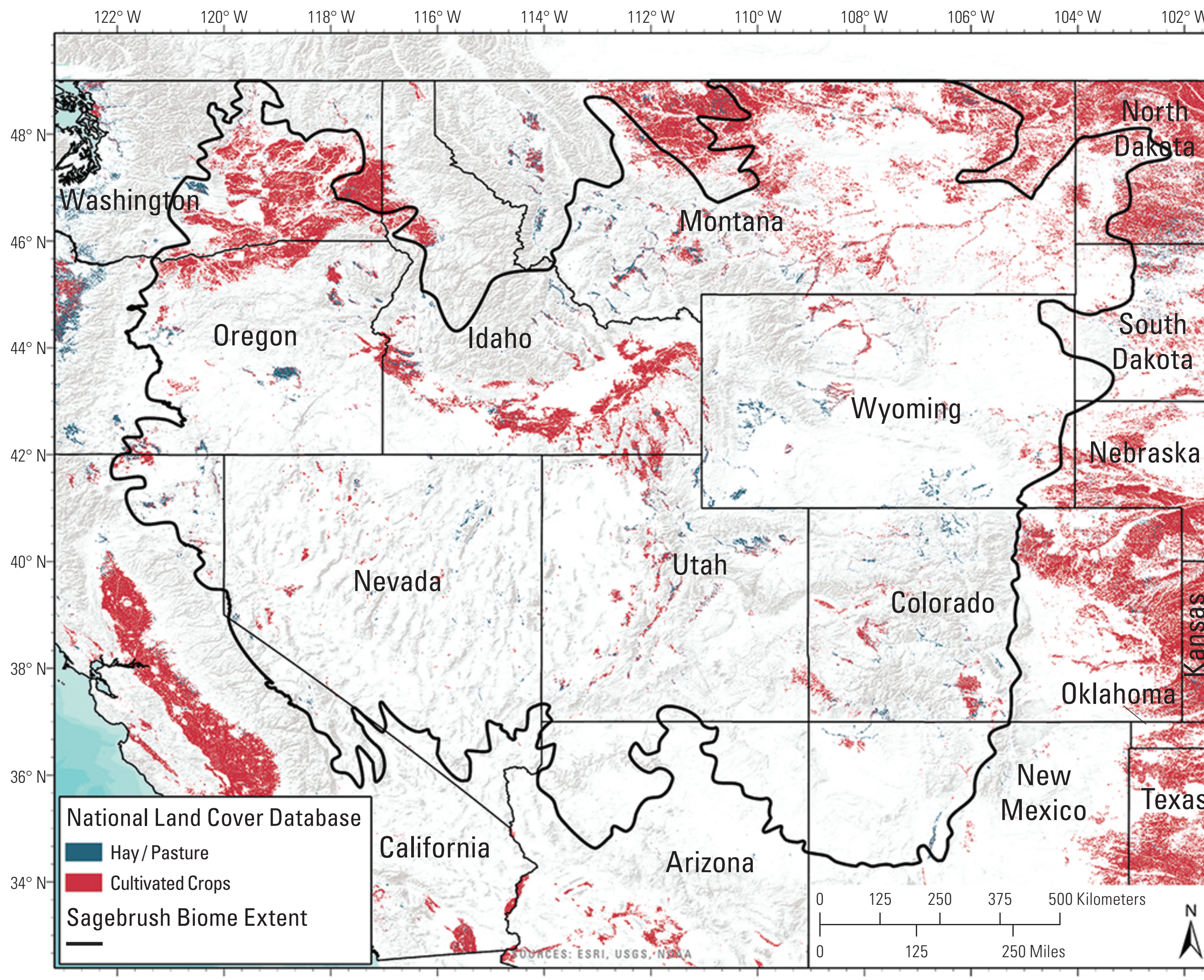

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Figure P1. Current distribution of cropland and pastureland (U.S. Geological Survey, 2019b) within the sagebrush (*Artemisia* spp.) biome extent.

remains untilled. For example, tillage rates of 21–25 percent of the landscape can lead to abandonment of display grounds by greater sage-grouse (*Centrocercus urophasianus*; (Tack, 2009; Knick and others, 2013). For songbirds, conversion of sagebrush to agriculture can lead to a community shift from obligate sagebrush species to generalist species along edge habitats (Vander Haegen and others, 2000; Knight and others, 2016). This pattern and distribution may result from increased nest predation and lower reproductive success because of corvids and rodents (Vander Haegen and others, 2002; Vander Haegen, 2007; Knight and others, 2014) or changes in vegetation composition (Knight and others, 2016). Increased predation rates are attributed to fragmentation of native habitat, although predator response varies by taxa and landscape context (Chalfoun and others, 2002).

Conversion of native habitats to cropland has been associated with greater sage-grouse population declines (Swenson and others, 1987; Leonard and others, 2000, reviewed in Knick and others, 2011; Smith, J.T., and others, 2016), but the relative contributions and interactions among direct habitat loss and indirect mechanisms, such as disturbance or changes in predator abundance, are unclear. Beyond the direct loss of sagebrush habitats, cropland and associated developments can subsidize sage-grouse predators, such as common ravens (*Corvus corax*; Engel and Young, 1989, 1992). Direct and indirect impacts of cropland conversion may affect as much as 61–99 percent of the sagebrush area within Western Association of Fish and Wildlife Agencies (WAFWA) delineated sage-grouse management zones (MZ; Stiver and others, 2006) when the effect size is considered around cropland (Knick and others, 2011).

The amount of tilled cropland was ranked as a minor factor in predicting greater sage-grouse breeding habitat (Doherty and others, 2016). However, the analysis by Doherty and others (2016) underestimated the historical impact of cropland conversion on sage-grouse distribution, as many productive lands with deeper soils that once supported greater sage-grouse were among the first lands converted to cropland (Vander Haegen and others, 2000) and are no longer considered in analyses of greater sage-grouse habitat within their current range. Several studies indicate that greater sage-grouse populations cannot persist in areas with less than (<) 25 percent landscape cover of sagebrush (Aldridge and others, 2008; Wisdom and others, 2011; Knick and others, 2013), and sage-grouse extirpations have occurred in areas where cultivated crops exceeded 25 percent landscape cover (Aldridge and others, 2008). Recent studies show that 96 percent of active leks are surrounded by <15 percent cropland in MZ I (Sage Grouse Initiative, 2015a; Smith, J.T., and others, 2016).

Sage-grouse are known to use agricultural fields periodically as strutting grounds and brood-rearing habitat. For example, alfalfa fields have been used by brood-rearing sage-grouse (Wallestad, 1971). Although the presence of sage-grouse is not necessarily indicative of habitat quality and enhanced fitness (Van Horne, 1983), alfalfa fields and irrigated croplands can be one of the few areas available to sage-grouse that produce invertebrate food resources in late summer, especially in drier years. However, mortalities have been attributed to pesticide applications associated with agricultural fields (Blus and Henny, 1997). The amount and configuration of sagebrush habitat in the surrounding landscape also influences habitat use (Schroeder and Vander Haegen, 2011).

Pygmy rabbits (*Brachylagus idahoensis*) can be directly impacted by sagebrush conversion to croplands through mortality at the time of tilling and clearing and indirectly through reductions in forage and cover, home-range abandonment, increased habitat fragmentation, increased predation, and population declines. Future loss of sagebrush habitat to cropland conversion is thought to be limited within pygmy rabbit range, as most of the tillable areas have already been converted (U.S. Department of the Interior, 2010b). However, the disproportionate loss of deep-soil sagebrush communities contributed to the decline of the Columbia Basin pygmy rabbit, and the last known wild subpopulation was extirpated in 2004 (Hayes, 2018). Recent winter surveys (2017) evaluating transplanted pygmy rabbits indicate that the majority of active burrows are located on croplands enrolled in the Conservation Reserve Program (CRP) in the Columbia Basin. These CRP lands are primarily deep soil areas that have been replanted with native vegetation and have a sagebrush component (Hayes, 2018).

Responses to cropland conversion for some species and taxonomic groups can be positive if conversion results in increased food resources. For example, mule deer (*Odocoileus hemionus*) use agricultural lands such as alfalfa and grain crops (Austin and Urness, 1993; Selting and Irby, 1997; Stewart and others, 2010; Anderson and others, 2012). Croplands may similarly attract corvids (O'Neil and others, 2018). Jackrabbits (*Lepus* spp.) responded positively to the conversion of sagebrush to cropland (Fagerstone and others, 1980; Simes and others, 2015), and can contribute to crop damage. Increasing jackrabbit populations also may attract shared predators of pygmy rabbits such as coyotes (*Canis latrans*; Lawes and others, 2012). Although golden eagles generally avoid agriculture (Marzluff and others, 1997; Domenech and others, 2015), at least one study noted that eagles may select cropland for foraging when compensating for loss of shrub habitat following fire (Kochert and others, 1999).

A relatively understudied aspect of cropland conversion is the response of arthropod species and communities. Sagebrush ecosystems adjacent to cropland can serve as sources of beneficial arthropods for agriculture (Miliczky and Horton, 2005; James and others, 2018), which may encourage retention of sagebrush plants in these edge habitats. However, reduced populations of arthropod species and communities were observed in such areas compared to undisturbed shrub-steppe (Quinn, 2004). The degree to which different types of croplands affect neighboring ecosystems and communities is also understudied. For example, arthropod populations were lower in small sagebrush fragments than large fragments and also lower adjacent to annual than perennial crops (Quinn, 2004).

Federal and State Agricultural Programs

Federal farm policy through the Agricultural Improvement Act of 2018 (commonly known as the Farm Bill; Public Law 115–334, 132 Stat. 4493) has tied commodity support programs to environmental programs such as CRP, where marginal cropland is taken out of production and converted to herbaceous cover (Burger, 2006). Fields converted to CRP may be used by species such as mule deer (Selting and Irby, 1997) and by greater sage-grouse depending on vegetation composition, availability of native habitat in the surrounding landscape, and management practices (Shirk and others, 2017). Seed mixes and plantings by the CRP often do not include a sagebrush component, although this can vary regionally. The lack of sagebrush in plantings means CRP can help to keep plant cover on the landscape and may help to minimize indirect impacts associated with cropland agriculture but may not provide direct habitat for sagebrush wildlife species. However, additional study is needed that links demographic responses to the distribution and population trends of species, and how these may vary with the age, structure, and composition of CRP fields (Shirk and others, 2017). The CRP program has limited capability to provide long-term protection for sagebrush ecosystems as CRP contracts are 10–15 years in length, and aggregate enrollment varies with changes in commodity prices (Rashford and others, 2011). Therefore, long-term persistence of CRP in sagebrush landscapes is uncertain.

Other provisions in the Farm Bill, such as crop insurance, disaster assistance, and other agricultural subsidies, are working at cross-purposes to preventing conversion and, by providing a safety-net against risk, resulting in conversion being more attractive to landowners (Claassen and others, 2011). The socioeconomic demand for biofuels and rising commodity prices also encourages conversion of native habitats to croplands (Lark and others, 2015).

Existing and emerging Federal, State, and foundation programs are available to help provide long-term protection for sagebrush ecosystems. Perpetual and long-term conservation easements are voluntary yet legally binding agreements by private landowners that typically limit or prohibit the removal of sagebrush and other native vegetation. The U.S. Fish and Wildlife Service (FWS) and some State fish and wildlife agencies hold perpetual grassland easements that preclude conversion of grass and shrub-steppe systems to cropland (fig. P2). For example, Habitat Montana, a license revenue program established by the State legislature in 1987, generates roughly $5–6 million per year to conserve important wildlife habitats. Montana Fish, Wildlife and Parks has leveraged Habitat Montana and other funding sources to enroll over 80,000 ha (200,000 acres) of sagebrush habitat in perpetual conservation easements that prevent conversion to cropland. Interest in conservation easements has increased dramatically in recent years, and States and land trusts are currently working with private landowners to enroll more acres in conservation easements across the sagebrush ecosystem, especially leveraging new cost-share opportunities under the Farm Bill. States are also implementing term protection mechanisms to support landowners who agree not to convert native grasslands and sagebrush in the near term. For example, Montana Fish, Wildlife and Parks holds over 80,000 ha (200,000 acres) of privately-owned sagebrush habitat in 30-year term leases that prevent conversion and currently has applications from private landowners for an additional 40,000 ha (100,000 acres). Other States, conservation organizations, and land trusts also hold conservation easements with varying degrees of protection from conversion to cropland agriculture (fig. P2). Many of these entities are also taking advantage of recent cost-share opportunities under the Farm Bill and State mitigation programs (for example, Wyoming and Montana).

Cost-shared financial and technical assistance to producers maintaining their sagebrush lands as grazed pasture or wildlife habitat can also help limit loss to cropland agriculture. Programs like the NRCS Environmental Quality Incentive Program (EQIP), FWS Partners for Fish and Wildlife, State-based private lands programs, and local partnerships supported by grant opportunities (for example, Ranchers Stewardship Alliance in northeastern Montana) help landowners sustain ranching operations or create opportunities for hunting that reduce the need for producers to convert native vegetation to croplands to meet the financial needs of their agricultural operations. Communities are also working to sustain ranching livelihoods. For instance, the Ranchers Stewardship Alliance has established a “grasslink” designed to connect producers who have expired CRP lands or other idle pasture with young and beginning ranchers in need of rental pasture. The grasslink helps to keep expired CRP lands from being planted back to crops by providing the landowner with revenue from leasing versus converting the lands.

Livestock Grazing

Sagebrush communities encompass almost 65 million ha (160 million acres) of land in the Intermountain West (chap. A, this volume; fig. A1), and most of those lands are grazed by domestic cattle, sheep, or horses. Understanding the differences between poorly managed and contemporary, well-managed grazing is key to understanding impacts, since different management approaches result in different outcomes. Vegetation response to livestock grazing will also vary across sagebrush ecoregions depending on local precipitation, soils, phenology of plants and timing of grazing, and selective grazing for specific plants within the community (Chambers and others, 2017a).

Numerous studies have investigated the impacts of livestock grazing on vegetation structure and composition in sagebrush ecosystems, especially as they relate to sage-grouse conservation (Hobbs and others, 1996; Beck and Mitchell, 2000; Davies and others, 2010, 2014c, 2016b). Sage-grouse also depend on sagebrush for food, protection, and nesting (Beck and Mitchell, 2000; Smith and others, 2018a; chap. D, this volume). The greatest potential for livestock grazing to affect greater sage-grouse habitat is by changing the composition, structure, and productivity of herbaceous plants used for nesting and early brood-rearing (Beck and Mitchell, 2000; Hockett, 2002; Boyd and others, 2014b). Improper livestock management such as heavy, repeated grazing may degrade rangelands through reductions in perennial herbaceous cover (Davies and others, 2011). In turn, reduced grass cover from poorly managed grazing also allows for unchecked growth of sagebrush (Daddy and others, 1988; Davies and others, 2010). Small sagebrush plants may be destroyed by cattle (Beck and Mitchell, 2000), but this is likely not enough to offset growth of sagebrush following release from competition with herbaceous cover (Daddy and others, 1988). Few studies directly link the effects of specific grazing systems to sage-grouse (for example, Smith and others, 2018b). Sage-grouse response likely depends on the long-term effects of specific grazing systems on plant-community attributes, especially the relative abundance of perennial grasses and forbs versus sagebrush (Dahlgren and others, 2015). At broad temporal and spatial scales, trends in sage-grouse populations may correspond with the timing and level of grazing on public lands (Monroe and others, 2017).

Cattle and native ungulates compete for the same forage, so loss of grassy forage is exacerbated in areas where cattle and native ungulates coexist (Hobbs and others, 1996; Veblen and others, 2015). Loss of grass cover in sagebrush ecosystems reduces the amount of available forage for native ungulates (Hobbs and others, 1996) and reduces habitat

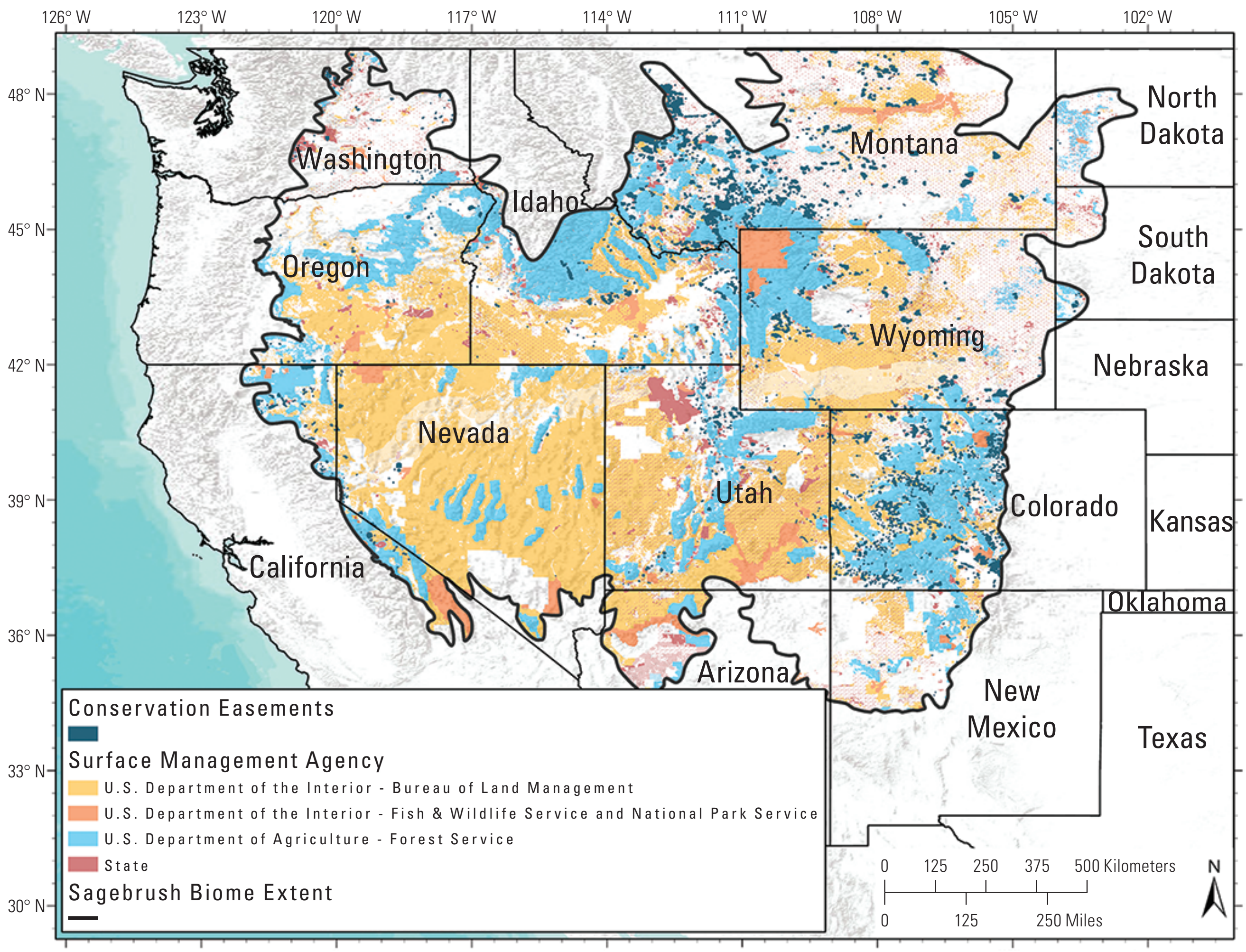

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Figure P2. Lands within the sagebrush (*Artemisia* spp.) biome extent that are generally protected from cropland conversion and residential development. Conservation easement data were obtained from the National Conservation Easement Database (U.S. Endowment for Forestry and Communities, 2018) and the Protected Areas Database (U.S. Geological Survey, 2018b). Surface management agency data was obtained from U.S. Department of the Interior, Bureau of Land Management, surface management layer (Bureau of Land Management, 2019c).

for facultative grassland or grassland-obligate bird species (Golding and Dreitz, 2017; Berkeley and Szczypinski, 2018). Grasses also provide hiding and nesting cover for sage-grouse (Miller and Eddleman, 2001; Hagen and others, 2007; but see Smith and others, 2018c) and other small animals. Conversely, sagebrush songbirds such as sagebrush sparrow (*Artemisiospiza nevadensis*), Brewer's sparrow (*Spizella breweri*), and sage thrasher (*Oreoscoptes montanus*) are positively associated with larger sagebrush plants and may have no response or a negative response to grass cover (Miller, R.A., and others, 2017).

General degradation of soil quality and water content are also seen in grazed sagebrush ecosystems. Desertification of heavily grazed areas because of greater surface evaporation, lower infiltration of water into the soil, and higher transpiration by growing sagebrush has been reported (Daddy and others, 1988). The shallow roots and taproots in sagebrush also cause them to be highly competitive for water, which prevents perennial and herbaceous plants from taking root (Daddy and others, 1988). Watering herbaceous plants did little to increase herbaceous cover; however, systems where sagebrush was removed and herbaceous plants were watered had significantly greater forb cover (Berlow and others, 2003). Rest for up to 20 years from grazing does not significantly alter soil crust cover, quality, or water content (Daddy and others, 1988; Davies and others, 2016b). This combination of factors indicates that communities dominated by big sagebrush (*A. tridentata*) may take a long time to recover because of the persistence of sagebrush and unfavorable soil quality for herbaceous plants (Davies and others, 2014c, 2016b).

Livestock grazing has been shown to affect fire regimes and invasive plant species in sagebrush ecosystems. Ungrazed sagebrush ecosystems have higher fuel accumulation and continuity than grazed systems (Davies and others, 2010). In systems where fire suppression is desired, grazing helps by decreasing fuel loads and reducing fire severity (Davies and others, 2010, 2014c) and by reducing the mortality of perennial bunchgrasses, likely from the reduction of highly flammable litter (Davies and others, 2009, 2010, 2016b). Perennial grasses play an important role in sagebrush ecosystems by preventing establishment of invasive annual grasses (Booth and others, 2003; Chambers and others, 2007; James and others, 2008; Condon and others, 2011); thus, moderate grazing may be a useful tool to mitigate effects of fire and cheatgrass (*Bromus tectorum*) invasion on sagebrush plant communities (Davies and others, 2011). However, fire suppression at higher elevations also may allow conifer encroachment, whereas repeated, heavy grazing at lower elevations may contribute to proliferation of exotic annual grasses (Davies and others, 2011). Invasive grass encroachment reduces foraging efficiency and success of small mammals, leading to their declines in sagebrush ecosystems (Bachen and others, 2018). These declines further lead to reduced populations of mesopredators such as badgers (*Taxidea taxus*), snakes, and raptors (Ceradini and Chalfoun, 2017). Deer mice (*Peromyscus* spp.) favor shrubs as hiding places, and they will choose native grasses for hiding cover over invasive grasses (Ceradini and Chalfoun, 2017).

Livestock grazing is often accompanied by changes in local infrastructure that can affect communities. Native ungulates, and especially pronghorn (*Antilocapra americana*), are often impeded, injured, or killed by fences (Harrington and Conover, 2006; Gates and others, 2012; Jones and others, 2019). Sage-grouse and other grouse species also collide with fences which causes injury or death (Van Lanen and others, 2017). Common ravens—a pervasive sage-grouse nest predator—respond strongly and positively to human use of sagebrush ecosystems (Coates and others, 2014b; Howe and others, 2014). Increased predation by common ravens may be partially owing to the presence of features such as stock ponds and troughs, associated perching structures (for example, windmills, tanks, and fences), and carcass dumps (Coates and others, 2016). The odds of common raven occurrence increased by approximately 46 percent in areas where livestock were present (Coates and others, 2016). This increase is potentially problematic for sagebrush-associated wildlife; as generalist predators, common ravens eat anything from carrion to fawns and calves and are strong drivers of prey populations (Coates and others, 2014b; Howe and others, 2014; Coates and others, 2016).

There are numerous financial- and technical-assistance programs available to support producers with sustainable grazing. Staff with the NRCS can help producers develop ranch-management plans and cost-share on grazing management practices, from water pipeline installation and fencing to reseeding native vegetation and other activities. The FWS Partners for Fish and Wildlife Program, State private lands programs, and local community partnerships also offer cost-share assistance for activities designed to support sustainable grazing on private lands to try to meet producer and conservation needs. Despite considerable Federal and other funding authorized for these activities, there is currently more interest in rangeland cost-share opportunities than there are funds available.

Some conservation easements include grazing- or land-management plans as an attachment to the easement, which helps to ensure long-term sustainable grazing practices. For example, conservation easements funded in part by the Agricultural Conservation Easement Program—Agricultural Land Easements (ALE) under the 2014 Farm Bill (Public Law 113–79, 128 Stat. 649) required an ALE management plan. Conservation easements held by some State fish and wildlife agencies require management plans or adherence to grazing standards; for example, Montana Fish, Wildlife and Parks attaches a management plan and grazing management standards to conservation easements. In addition, conservation easements held by some not-for-profit organizations, like The Nature Conservancy, are also crafted to support sustainable, long-term range management. The requirement to include an ALE management plan was removed in the 2018 Farm Bill (Public Law 115–334, 132 Stat. 4490), however, allowing range management to remain at landowner discretion in ALE-funded conservation easements.

The U.S. Department of the Interior, Bureau of Land Management (BLM) and U.S. Department of Agriculture, Forest Service have incorporated grazing management standards specifically to support sage-grouse habitat in recent land use plan amendments and revisions. The BLM and Forest Service have developed a habitat assessment framework designed to measure if pastures are meeting established standards. This will help to ensure that public rangelands are and will continue to be managed sustainably. Additional innovative solutions to support sustainable range management continue to emerge. For example, The Nature Conservancy has established a grass bank in northeastern Montana. In exchange for reduced grazing rates on the grass bank, enrolled landowners agree to implement certain conservation practices on their fee title lands. This model helps to keep sustainable grazing profitable for the producers, on their home pastures and on the grass bank.

Infrastructure

Roads

The effects of roads in the sagebrush biome are a function of (1) the degree that these linear features remove sagebrush and fragment the landscape and (2) the resulting change in plant and animal distribution that may impact sagebrush wildlife species. The presence of roads and vehicles also directly impacts mortality rates of many sagebrush-associated wildlife species through collisions, facilitates disturbance by providing people access into sagebrush areas, and increases vehicle noise levels that alter animal behavior and habitat use. Distribution, survival, and reproductive success of greater sage-grouse are negatively affected by roads (Lyon and Anderson, 2003; Holloran, 2005; Webb and others, 2012; Hovick and others, 2014; Smith and others, 2018a). Although interstate and major paved highways cover an estimated 0.1 percent of the land cover in sage-grouse MZs, they influence 38 percent of the sagebrush land cover when their effect size is considered (see Knick and others, 2011, table 12.3; effect area of 7 kilometers [km]; 4.3 mile [mi]). Secondary roads, railroads, and especially transmission lines may have additional fragmentation effects, with the greatest overall influence on sagebrush area in the Columbia Basin, Wyoming Basins, and Colorado Plateau (fig. P3; see also Knick and others, 2011, table 12.3; Manier and others, 2014).

Depending on road width and traffic volume, roads may represent barriers to the movement of large ungulates such as pronghorn (Gates and others, 2012; Christie and others, 2017) and mule deer (Sawyer and others, 2009b; Sawyer and others, 2017). Roads can indirectly affect wildlife populations if they attract predators such as rodents (Mahoney and Chalfoun, 2016), coyotes (Lawes and others, 2012), felids (Lendrum and others, 2018), corvids (O'Neil and others, 2018), and raptors (Benítez-López and others, 2010). Roads may contribute to direct mortality because of collisions with vehicles (Summers and others, 2011; Nenninger and Koper, 2018), and roadkill in turn could attract predators (O'Neil and others, 2018; Benítez-López and others, 2010). Noise from traffic has been posited as a potential mechanism for affecting wildlife (Smith and Dwyer, 2016). Road noise may reduce lek attendance for greater sage-grouse (Blickley and others, 2012a), but evidence that traffic noise impacts passerines is lacking (Summers and others, 2011; Nenninger and Koper, 2018).

However, the level of traffic and associated noise from roads may negatively affect other species. Mule deer tended to occur further from roads with higher traffic levels (Sawyer and others, 2009a; Anderson and others, 2012), and predators were more likely to hunt near infrastructure without human activity (Lendrum and others, 2018). Greater sage-grouse also respond to the density of highly trafficked roads; 93 percent of active leks were located in landscapes with <0.01 kilometers/square kilometer (km/km^2; 0.02 miles/square mile [mi/mi^2]) interstate highway densities (Knick and others, 2013). Higher densities of secondary roads, up to <0.10 km/km^2 (0.16 mi/mi^2) were associated with higher sage-grouse habitat suitability (Knick and others, 2013). This suggests that an interaction between the density of roads and traffic level may influence wildlife species' responses to roads.

Species may respond differently depending on how vegetation is removed or altered along roads. For example, the abundance of sagebrush-obligate passerines may decline near roads, whereas short structure specialists such as horned larks (*Eremophila alpestris*) and chestnut-collared longspurs (*Calcarius ornatus*) are more abundant (Ingelfinger and Anderson, 2004; Nenninger and Koper, 2018). Similarly, roadsides with reduced sagebrush cover were avoided by pygmy rabbits but attractive to other leporids, which could lead to competition with pygmy rabbits and potentially increased pygmy rabbit mortality rates if shared predators are drawn by greater leporid abundance (Pierce and others, 2011). Additionally, roads may alter sagebrush communities by serving as conduits for exotic plant invasion and establishment (Gelbard and Belnap, 2003; Manier and others, 2011; Barlow and others, 2017).

The impacts of roads on wildlife can be mitigated through seasonal closures and over/under passes at strategic locations. For instance, in Gunnison County, Colorado, the BLM and the Forest Service temporarily close a number of roads to all motor vehicles annually from March 15 to May 15 or mid-June (Forest Service) to protect Gunnison sage-grouse (*C. minimus*) during their mating season. The Colorado Department of Transportation, in cooperation with Colorado Parks and Wildlife and many other partners, recently implemented Colorado's first-of-its-kind wildlife overpass and underpass system on State Highway 9, a busy highway that bisected mule deer and elk movement corridors and winter range. This project consisted of 2 wildlife overpasses, 5 wildlife underpasses, 9 pedestrian walk-throughs, 62 wildlife escape ramps, and 29 wildlife guards along a 17.7-km (11-mile) section of highway, all of which are connected by an 8-foot high wildlife fence. This project resulted in a 90-percent reduction in wildlife-vehicle collisions in the first year, and from 82 to 98 percent of mule deer crossing attempts were successful at these structures. In Montana, the Department of Transportation, in cooperation with the State fish and wildlife agency, has developed a new project assessment process that specifically includes potential fish and wildlife connectivity and crossing issues and requires consultation with relevant agencies.

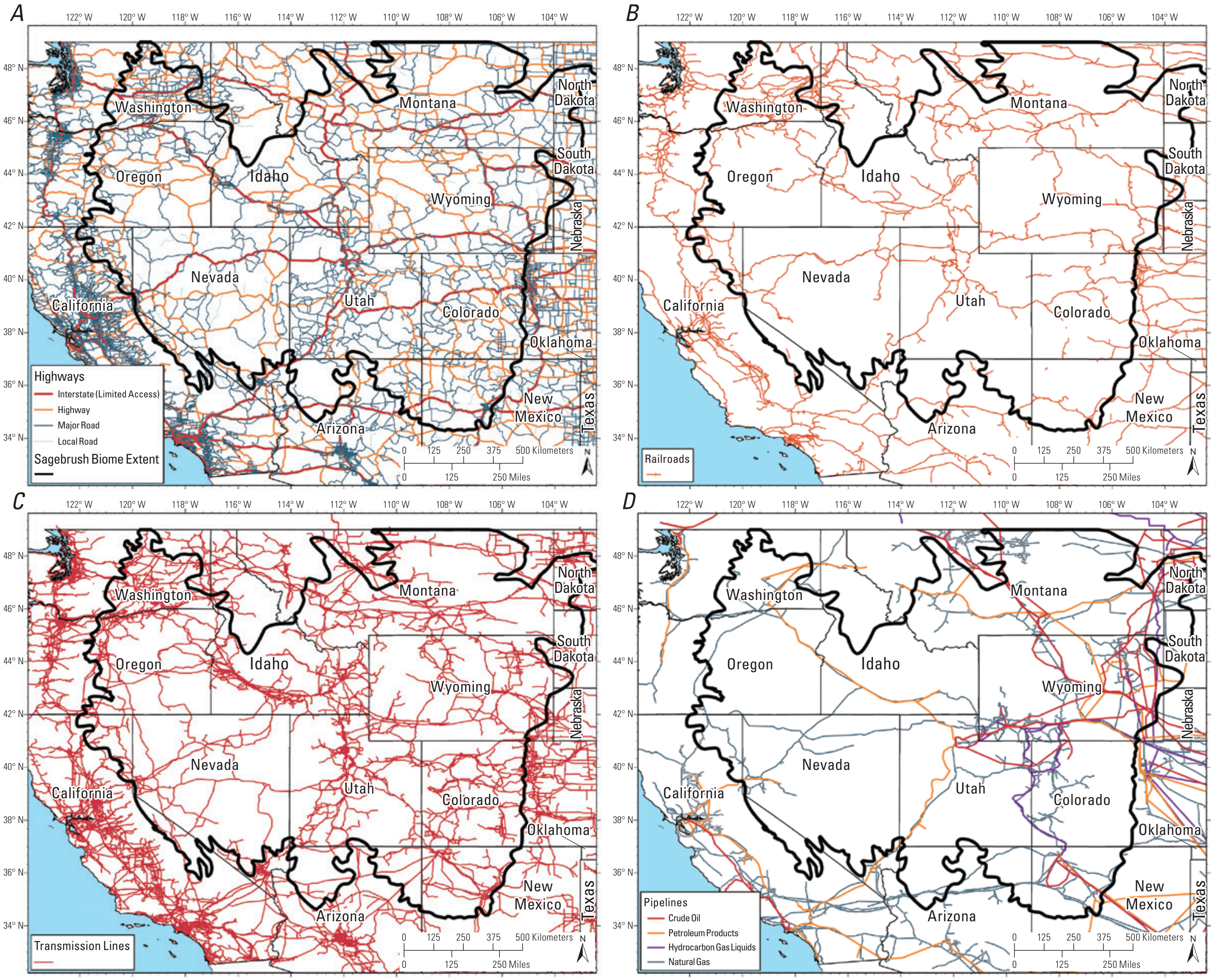

Figure P3. Distribution of infrastructure across the sagebrush (*Artemisia* spp.) biome including *A*, roads (Interstates, highways, major roads, and local roads), *B*, railroad tracks, *C*, transmission lines, and *D*, pipelines. Data were obtained for panel *A* and *B* from the Esri Street Map purchased July 29, 2014, for panel *C* from the Esri Living Atlas Electric Power Transmission Lines Feature Service (Homeland Security Infrastructure Program Team, 2019), and for panel *D* from U.S. Energy Information Administration (U.S. Energy Information Administration 2019a, b, c, d).

Railroads

Effects of railroads on sagebrush ecosystems are relatively understudied and are estimated to impact a smaller proportion of the landscape than other types of infrastructure. However, railroads contribute to the spread of exotic grasses and sparks from trains serve as ignition sources for wildfire (Knick and others, 2011).

Pipelines

Similar to roads, buried pipelines may fragment and alter landscapes; however, right-of-ways (ROWs) are typically revegetated instead of paved or left barren. Depending on revegetation practices, the impact is likely greatest in the initial years following construction but may dissipate over time (Gasch and others, 2016; Pierre and others, 2017). Other effects associated with pipelines include spills, off-road transportation, and the introduction of invasive plant species during construction and maintenance (Ramirez and Mosley, 2015). However, invasive plants can be controlled during installation of pipelines when paired with herbicide treatments (Johnston, 2015). Reported negative effects include reductions in the abundance and distribution of pygmy rabbits (Germaine and others, 2017). Mule deer predation risk was lower than expected along pipelines and higher near roads, perhaps owing to greater visibility of potential predators (Lendrum and others, 2018). The density of pipelines may also be an important influence on wildlife habitat suitability. Sage-grouse leks were absent from the landscape when pipeline densities exceeded 0.47 km/km^2 (0.76 mi/mi^2), and areas of highest sage-grouse habitat suitability had densities of 0.01 km/km^2 (0.02 mi/mi^2) of pipeline (Knick and others, 2013) or less, although this may be due more to the infrastructure that pipelines were serving than the presence of pipelines themselves. State and Federal regulatory and mitigation programs help to minimize impacts from new pipeline construction.

Transmission Lines

Like other infrastructure, sagebrush is removed along the ROW during installation of transmission lines, which results in direct habitat loss and fragmentation. Transmission towers or poles create vertical structures that remain on the landscape for many decades, and often have supporting guy wires and conductor wires that present a collision and electrocution risk for many bird species (Avian Power Line Interaction Committee, 2006, 2012). The indirect influences that transmission lines can have on vegetation community dynamics and species occurrence have been documented to extend beyond their physical footprint (Knick and others, 2011; Dinkins and others, 2014a; Gibson and others, 2018). In a comparative study between extirpated and extant sage-grouse populations, distance to transmission and power lines was one of several explanatory variables inferring extirpation (Wisdom and others, 2011).

Transmission-line structures provide hunting perches and nesting substrate for raptors and corvids, often in habitats that are typically devoid of trees or other natural tall structures (Steenhof and others, 1993; Vander Haegen and others, 2002; Manville, 2005; Howe and others, 2014; O'Neil and others, 2018). Raptors and common ravens have been shown to use transmission lines during the breeding season postconstruction (Steenhof and others, 1993), and increases in their abundance or hunting activity near transmission lines has been described (Steenhof and others, 1993; Coates and others, 2014b, c; Gibson and others, 2018), which is negatively associated with sage-grouse demographic vital rates (Dinkins and others, 2014a; Gibson and others, 2018). Presumably, based on the presence of transmission lines and the associated increased presence of predators, sage-grouse have been observed to shift habitat use away from these areas (Braun, 1998; Hanser and others, 2011a; Gillan and others, 2013; Dinkins and others, 2014b).

Synthesis of connectivity work in Washington suggests that transmission lines increased resistance to sage-grouse movement, gene flow, and lek activity (Shirk and others, 2015). Wildlife may also avoid the electromagnetic fields produced by transmission lines (Wisdom and others, 2011). Electromagnetic fields can alter behavior, physiology, endocrine systems, and immune function in birds, with negative consequences on reproduction and development (Fernie and Reynolds, 2005). Ground-nesting birds may be sensitive to ultraviolet light (not visible to the human eye) emitted by transmission lines as standing coronas and irregular flashes on insulators (Tyler and others, 2014). The construction and maintenance of transmission lines can also facilitate the spread of nonnative invasive plant species (such as cheatgrass) as equipment travels off-road and in habitats that would not normally be traveled through (Gelbard and Belnap, 2003; Knick and others, 2003; Connelly and others, 2004).

In an effort to reduce the impact of transmission lines on wildlife, the Avian Powerline Interaction Committee (APLIC) developed a set of best management practices that the industry could employ to reduce wildlife impacts (Avian Power Line Interaction Committee, 2012). In addition to complying with regulatory requirements under the National Environmental Policy Act of 1969 (42 U.S.C. 4321 et seq.)—or in some cases the Endangered Species Act of 1973 (16 U.S.C. 1531 et seq.)—the power industry has implemented voluntary measures at their own costs to reduce impacts. For instance, in Colorado, Tri-State Electric, in order to reduce impacts on Gunnison sage-grouse when replacing an existing transmission line, voluntarily changed their standard wood H-frame design to a single, self-supporting steel structure; eliminated guy wires on turning structures to make the poles self-supporting; and installed perch discouragers on horizontal surfaces and the top of each structure at a cost of over $2 million. In Montana, NorVal Electric Cooperative, Inc. voluntarily agreed to bury existing overhead transmission lines in a sage-grouse core area as mitigation for new overhead, non-nest facilitating transmission lines and buried distribution lines elsewhere.

Communication Towers

Communication towers emit high levels of electromagnetic radiation, which has been linked to decreased populations and reproductive performance of some bird and amphibian species (Balmori, 2005; Balmori and Hallberg, 2007; Everaert and Bauwens, 2007). Millions of birds, primarily passerines, are killed annually in the United States through collisions with communication towers (including cellular towers) and their associated structures (for example, guy wires, lights; Manville, 2005; Longcore and others, 2012). Proximity to communication towers was a strong indicator of sage-grouse extirpation (Wisdom and others, 2011); however, there is no mechanistic explanation for a direct relationship. Distance to communication towers is also indicative of the most intensive human developments, concentrated along major highways, and within and near larger urban areas, which could confound the effects of communication towers on sage-grouse populations. Similar to other types of infrastructure, additional effects associated with communication towers include spreading exotic plant species and increased predation risk by providing perches for corvids and raptors. Existing State executive orders and Federal land-management plans encourage proponents of new communication towers to place towers in locations that will have minimal impact on sage-grouse.

Future Research Needs

Our understanding of the influence that infrastructure has on wildlife occurrence and vital rates is not complete. Additional study is needed linking animal distribution to road proximity and density; examining mechanisms such as collisions, cover, food resources, and predators; and how these affect survival, reproduction, and population trends across sagebrush landscapes (Sawyer and others, 2006; Gates and others, 2012; Mutter and others, 2015; Smith and Dwyer, 2016; Nenninger and Koper, 2018). For transmission lines and communication towers, several studies have reported conflicting results, documenting no observed response, or reported an inability to isolate the effect of tall structures (Johnson and others, 2011; Messmer and others, 2013; Walters and others, 2014; Smith and Dwyer, 2016) from roads or other disturbances. Distinguishing between numerical and functional responses by predators is also needed (Chalfoun and others, 2002). Additionally, studies to isolate effects of infrastructure from other anthropogenic structures when they co-occur, such as with energy development, are needed (Smith and Dwyer, 2016).

A lack of predisturbance data and proper controls is also a common limitation of infrastructure studies (Sawyer and others, 2006; Benítez-López and others, 2010; Messmer and others, 2013; Smith and Dwyer, 2016; Christie and others, 2017; Sawyer and others, 2017). Additionally, many responses to roads may depend on a study's spatial and temporal context. For example, the effects of roads may vary with predator abundance (Mahoney and Chalfoun, 2016), the density of roads, and other anthropogenic features in the landscape (Frair and others, 2008; Mutter and others, 2015). Thus, greater replication of studies over space and time may clarify species' relationships with infrastructure and may increase the inferential base to make predictions across landscapes. Manipulative studies also would be useful for clarifying mechanisms behind species and community responses, but this may not be feasible at broader scales (Sawyer and others, 2009a). Instead, studying existing treatments and adaptive management approaches may be suitable for inferences.

Residential Development

Approximately 6,068 square kilometers (km^2; 2,343 square miles [mi^2]) of undisturbed lands were lost to residential development in the western United States between 2000 and 2011, making urban sprawl the leading cause of recent open-space loss in the West (fig. P4; Theobald and others, 2016). The estimated area lost to urban development within sage-grouse MZs ranges from 0.2 (Great Plains) to 1.1 percent (Columbia Basin), with indirect effects ranging from 16.4 (Great Plains) to 48.5 (Columbia Basin) percent (see Knick and others, 2011). The greatest residential development within the sagebrush biome is centered around larger cities, such as Salt Lake City, Utah; Spokane, Washington; Boise, Idaho; and Reno, Nevada (Theobald and others, 2016). However, residential development in rangelands outside of cities and towns produces a footprint that is 5 to 10 times larger than an urban footprint, which can be problematic for maintaining adequate patch size and connectivity for wildlife (Reeves and others, 2018b). For example, in Montana, nearly 50 percent of new homes built between 1970 and 2016 were constructed on lots averaging 4.05 ha (10 acres; Gude, 2018).

Residential development tends to shift plant communities toward nonnative plant species (Maestas and others, 2003), shift bird communities toward nonnative species (Marzluff, 2001), and support lower species richness of insects (including bees) and plants (Hansen and others, 2005). At least 19 bird species associated with grassland or shrub habitats are sensitive to patch size or fragmentation, including sage thrasher and Brewer's sparrow (Freemark and others, 1995). Human disturbance can also directly influence wildlife occurrence and abundance. In Colorado, grassland bird abundance decreased significantly when human activity influenced 5 percent or more of every 40 ha (100 acres; Haire and others, 2000). Greater sage-grouse occurrence was influenced by human disturbance in the western part of their range (Knick and others, 2013). Lower densities of ground- and shrub-nesting birds and higher densities of domestic mammalian predators (that is, domestic cats [*Felis catus*] and dogs [*Canis lupus familiaris*]) have been reported in exurban development areas in comparison with ranches and wildlife reserves (Maestas and others, 2003). The presence of domestic pets can negatively impact nesting birds, leading to lower reproductive success along with lower abundance (Faaborg and others, 1995; Loss and Marra, 2017). There is little empirical data on the impact of residential development specifically on sagebrush ecosystem process and species, but available research suggests the implications of continued development could be significant. Hansen and others (2005) state "* * * exurban development is a pervasive and fast-growing form of land use that is substantially understudied by ecologists and has large potential to alter biodiversity."

Efforts to mitigate residential development in sagebrush landscapes are primarily voluntary. Roughly 62 percent of all rangelands in the United States are in private ownership, and private landowners typically retain development rights, meaning that landowner decisions can have a significant impact on sagebrush conservation (Reeves and others, 2018b). Local governments set the policy and regulatory framework for residential development, which includes comprehensive plans and development regulations. However, long-range planning and resource values at landscape scales (for example, wildlife migration corridors) are often not considered in these regulations. State fish and wildlife agencies have the opportunity to provide input on local government policies and development projects, but it is challenging to develop useful information for decision makers that will be effective in reducing impacts to wildlife. There are some examples of States working with county governments to protect important wildlife habitats, such as Montana's Model Subdivision Recommendations for Wildlife.

Conservation easements and long-term leases are voluntary on the part of the landowner but can provide legal protection from development. The terms of some of these protection mechanisms prohibit subdivision development, whereas others allow for development if some minimum acres of open space are maintained. Some larger communities in the western United States have approved open space bonds that provide funding

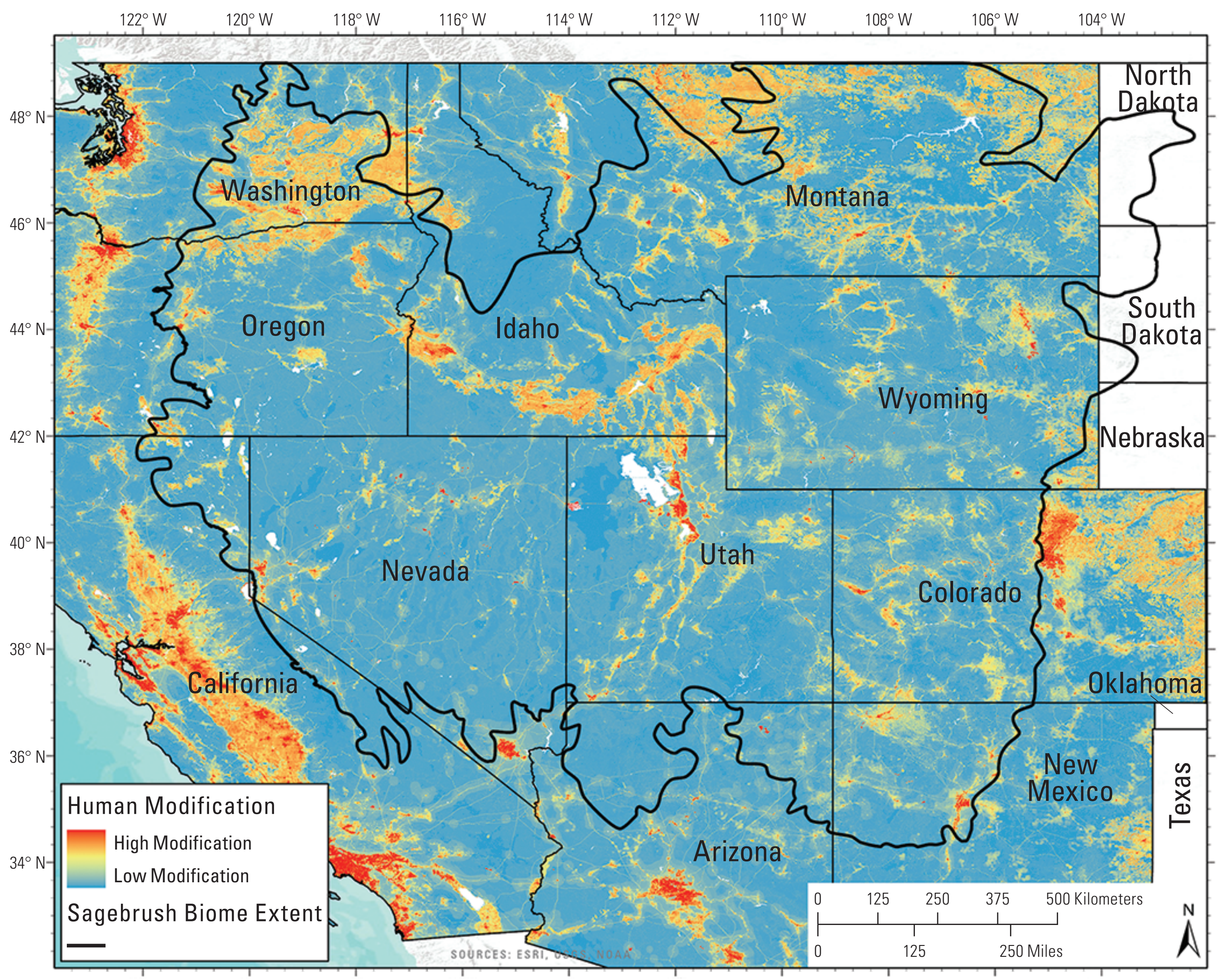

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Figure P4. Human modification (Theobald, 2013; Theobald and others, 2016) within the sagebrush (*Artemisia* spp.) biome by 2011.

that can be leveraged by land trusts to help protect important areas associated with those communities. However, many of the smaller communities in the sagebrush biome do not have open space bonds and rely on State and national level land trusts and agencies to purchase conservation easements and leases, when there are willing landowners.

There are examples where State-led initiatives are conserving sagebrush landscapes from subdivision development through the use of conservation easements (see fig. P2). In Colorado, sportsmen and sportswomen, concerned about loss of wildlife habitat to exurban development and loss of hunting access, pushed for legislation in 2006 that created the Colorado Wildlife Habitat Program, funded by a requirement for hunters and anglers to purchase a now $10 habitat stamp prior to applying for, or purchasing a hunting or fishing license. These habitat stamp funds typically accrue $6–7 million annually. They are matched with Great Outdoors Colorado lottery, and sometimes Federal and other funds and are used to purchase both permanent conservation easements and perpetual and term-limited public access easements. Since its inception, the Colorado Wildlife Habitat Program has invested over $164 million to protect 104,000 ha (257,000 acres), many of which are in sagebrush landscapes to benefit Gunnison or greater sage-grouse, mule deer, and other sagebrush-associated species.

Research from Wyoming suggests that $250 million of targeted conservation easements—along with their State core area regulatory protections—is the most effective conservation strategy for sage-grouse (Copeland and others, 2013). The modeled addition of targeted conservation easements that limited development could avert an additional 9–11 percent of declines in sage-grouse populations over what was expected with core area regulatory protections alone. Random acquisitions of conservation easements within the sagebrush ecosystem, however, had limited effects on sage-grouse populations. Targeting priority areas can be useful to land managers making decisions on how to allocate limited financial resources, but the opportunity to acquire conservation easements in priority areas remains dependent on landowner choice.

Recreation

Recreational activities (hunting, hiking, camping, off-highway vehicle [OHV] use, snowmobiling, mountain biking, fishing) are common in sagebrush ecosystems and have both direct and indirect impacts on sagebrush communities. Hundreds of thousands of hunters hunt for mule deer, elk, and pronghorn in sagebrush and adjacent habitats. Lead ammunition used in hunting can have detrimental health impacts to nontarget wildlife, especially for scavenging species such as coyotes, common ravens, and golden eagles (Hunt, 2012; Legagneux and others, 2014; Katzner and others, 2018). Antler shed hunting in late winter may inadvertently stress wintering mule deer, elk, and potentially sage-grouse, while shed hunting seasons designed to mitigate winter impacts may impact nesting sage-grouse.

Use of OHVs in sagebrush habitats has been shown to have negative effects on songbird breeding success by increasing the likelihood of nest abandonment near roads (Barton and Holmes, 2007). Abundance declined in a few songbird species with OHV use, but this relationship is unclear (Barton and Holmes, 2007). Golden eagle populations were also projected to decline with an increase in OHV and hiking recreation (Pauli and others, 2017). Native ungulates flushed in response to both hiking and mountain biking activity at anywhere from 50 to 200 meters (m; 164 to 656 feet [ft]) from trails (Taylor and Knight, 2003). Bald eagles (*Haliaeetus leucocephalus*) in remote, forested habitats flushed from nests in response to human activity at distances over 300 m and did not habituate to human activity (Fraser and others, 1985). Frequent movement as a result of human activity is energetically costly to animals and may decrease fitness or cause animals to avoid areas that they would otherwise occupy (Taylor and Knight, 2003). The direct impacts of drones, hiking, biking, and dispersed camping are unclear, but any associated disturbance to wildlife is energetically costly and may decrease fitness or cause animals to avoid areas that they would otherwise occupy (Taylor and Knight, 2003). It is also unclear if OHV or other forms of motorized recreation increase the risk of fire in vulnerable sagebrush ecosystems.

Increased efforts by BLM and Forest Service to manage recreational use on public lands through road closures and established campsites are helping to reduce impacts to sagebrush landscapes from recreational use. States routinely implement seasonal road closures on Wildlife Management Areas to protect important winter sagebrush habitat for mule deer, elk, and other wildlife species.

Cumulative Impacts and Conclusions

Land use and development can be a challenge or an opportunity within the sagebrush biome. At times, the same action will have negative implications for some aspects of the system and positive implications for others. Research is needed to better understand the role of infrastructure, grazing management, residential development, and recreation on the ecology of the sagebrush biome, especially research that seeks to understand the implications of multiple uses. For example, the cumulative human footprint influenced sage-grouse lek attendance trends across the range of sage-grouse, regardless of the type of anthropogenic stressor (Johnson and others, 2011). Voluntary strategies for sagebrush conservation will need to consider the tradeoffs related to each change agent and focus on the most impactful agents and corresponding conservation actions.

Part III. Current Conservation Paradigm and Other Conservation Needs for Sagebrush

Chapter Q. Sage-Grouse Management as an Umbrella for Conservation of Sagebrush

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Executive Summary

Management emphasis for sagebrush (*Artemisia* spp.) has largely shifted to conservation of sage-grouse (*Centrocercus* spp.) as State and Federal agencies, nongovernmental organizations, and landowners have joined in formal and informal partnerships to keep greater sage-grouse (*C. urophasianus*) from being listed under the Endangered Species Act of 1973 (16 U.S.C. 1531 et seq.). There is no single coordinated effort or plan to affect greater sage-grouse conservation; rather there are 11 different State plans with mitigation constructs and 98 Federal land use plans. These plans are implemented by Federal and State land and wildlife management agencies across the biome and supplemented by conservation practices implemented through State programs; the U.S. Department of Agriculture, Natural Resources Conservation Service Sage Grouse Initiative; and efforts by numerous nongovernmental organizations, working groups, and individual landowners. Land use restrictions and conservation efforts to protect sage-grouse are focused on lands identified as priority habitat management areas or core areas, a subset of all habitats used by sage-grouse. Disturbance caps, development density restrictions, lek-based buffers, and conservation and mitigation programs put in place for sage-grouse may also help conserve other sagebrush-associated species of concern depending on species and niche overlap. The degree to which other sagebrush-dependent and -associated species' use of ranges and habitats overlaps those of sage-grouse varies widely, and coverage of the sage-grouse umbrella shrinks when assessed at smaller spatial and temporal scales. The effectiveness of this conservation "umbrella" is likely to be limited for most sagebrush-dependent species because (1) key threats such as invasive plants, fire, and free-roaming equids are not effectively addressed by the umbrella; (2) important habitats for these species and sage-grouse are scale dependent and do not necessarily overlap at relevant spatial and temporal scales; and (3) responses to disturbance or habitat treatments differ. To ensure the conservation of sagebrush-dependent species, key habitats should be identified, and benefits or impacts to these species should be considered during project-level planning. Future management direction for the sagebrush biome may be more effective with a move towards maintenance of ecosystem resilience and resistance and conservation of the entire suite of sagebrush-dependent and -associated species.

Introduction

There are greater than 735 species of plants, vertebrates, or invertebrates inhabiting the sagebrush biome in the Great Basin alone (Wisdom and others, 2005). Approaches to management and conservation of sagebrush (*Artemisia* spp.) are as diverse and varied as the land ownership of sagebrush landscapes, but recent management emphasis has typically focused on conservation of habitats for sage-grouse (*Centrocercus* spp.). Conservation actions for this widely distributed species are frequently considered as an "umbrella"—benefitting other sagebrush species that often lack data or resources for development of individual conservation strategies (Rich and Altman, 2001; Rowland and others, 2006; Hanser and Knick, 2011; Carlisle and others, 2018a; Runge and others, 2019; Pilliod and others, 2020a). However, the degree to which conservation for sage-grouse conserves other species depends on species distribution overlap and efficacy of conservation actions to protect potentially dissimilar habitat requirements. A broad-scale analysis of management approaches, land uses, and conservation actions in sagebrush, and their effect on sagebrush-associated species, can help identify any conservation gaps between sage-grouse and other sagebrush species and inform efforts to prescribe actions to reduce them.

Management of Sagebrush

Landscapes that support sagebrush plant and animal communities are working landscapes that host a variety of land uses (for example, grazing and recreation, transmission corridors, mining, or oil and gas development). The current ownership patterns across the sagebrush biome and the amount, extent, and condition of sagebrush habitats are artifacts of past policy and practices (Knick and Rotenberry, 2000; Morris and others, 2011). Thus, to be effective, contemporary management and conservation strategies for

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sagebrush ecosystems will need to consider this legacy of land use, ownership, and past management practices both locally and across landscapes or regions of interest.

Concern over declining populations and a potential listing of sage-grouse (primarily greater sage-grouse [*Centrocercus urophasianus*] but also Gunnison sage-grouse [*C. minimus*]) under the Endangered Species Act of 1973 (ESA; 16 U.S.C. 1531 et seq.) began to influence management and conservation of sagebrush on public and private lands in the early 2000s. This management emphasis followed nine petitions to list various populations or presumed subspecies of greater sage-grouse and Gunnison sage-grouse under the ESA between 1999 and 2005. The 2000s generally saw management improvements for sage-grouse by State and Federal agencies. State wildlife agencies formed local working groups that then developed local conservation plans. Funding and support for mechanical or chemical sagebrush elimination treatments disappeared. Long-standing seasonal timing and other stipulations designed to protect sage-grouse breeding activity from development or other disturbances were updated and, less commonly, waived. Restoration efforts on burned or otherwise degraded habitats increased (Pilliod and others, 2017b). Research efforts on sage-grouse and attempts to better understand and map seasonal habitats were accelerated. The Western Association of Fish and Wildlife Agencies (WAFWA), with U.S. Fish and Wildlife Service (FWS) funding, produced "The Conservation Assessment of Greater Sage-grouse and Sagebrush Habitats" in 2004 (Connelly and others, 2004), followed by the "Greater Sage-grouse Comprehensive Conservation Strategy" in 2006 (Stiver and others, 2006). The U.S. Department of Agriculture (USDA), Natural Resource Conservation Service (NRCS) began the Sage Grouse Initiative (SGI) in 2010, a cost-share program to incentivize landowners and public land management agencies to adopt positive conservation measures for sage-grouse.

In March of 2010, the FWS found that greater sage-grouse were warranted for listing as threatened under the ESA but that further action on the listing was precluded by higher listing priorities (U.S. Department of the Interior, 2010a). This finding also established the Bi-State population, along the California-Nevada border, as a distinct population segment (DPS) of greater sage-grouse and put both the Bi-State population and greater sage-grouse rangewide on their candidate species list. The reasons for the finding included increased threats to habitat, habitat loss, a greater understanding of the impact of those threats to sage-grouse, and the inadequacy of regulatory mechanisms on Federal lands to alleviate these threats (U.S. Department of the Interior, 2010a). Concern about declining sage-grouse populations and the potential economic impact of an ESA listing accelerated activity by State and Federal agencies and communities potentially affected by such a listing. Litigation against the FWS for failure to make expeditious progress on resolving candidate species resulted in an agreement to complete a status review of greater sage-grouse by September of 2015.

The U.S. Department of the Interior (DOI), Bureau of Land Management (BLM) chartered a National Technical Team (NTT) in August of 2011 to review scientific information and advise the agency on developing new or revised regulatory mechanisms through resource management plans (RMPs) to conserve and restore greater sage-grouse and its habitat on BLM-administered lands. The NTT released a report in December of 2011 (Sage-grouse National Technical Team, 2011), which described a variety of conservation measures that the BLM could implement through land use plan amendments to protect priority sage-grouse habitats from anthropogenic disturbances. Priority sage-grouse habitats were defined, in coordination with the States, as areas that have the highest conservation value to sage-grouse populations, and include seasonal habitats and, where known, migration or connectivity corridors.

States and Federal agencies increased monitoring (lek counts); mapping of important seasonal habitats, research, and translocation efforts; and, if not already in place, established protocols for restricting or eliminating hunting in small or declining populations. Most Governors of States within the range of sage-grouse convened working groups to develop State plans for conservation and mitigation. The Western Governors' Association Sage-Grouse Task Force was formed shortly after a December 2011 meeting cohosted by then Wyoming Governor Mead and then Secretary of the Interior Ken Salazar to promote rangewide sage-grouse conservation efforts among the 11 sage-grouse States and BLM, FWS, NRCS, and USDA Forest Service. In response to a request from western Governors for FWS to provide further clarity over what conservation actions were needed to deter listing, then FWS Director Dan Ashe convened a Conservation Objectives Team (COT) in the spring of 2012. This team of State and Federal agency representatives with sage-grouse expertise or responsibility developed rangewide conservation objectives for greater sage-grouse. These conservation objectives defined the degree to which threats needed to be reduced or ameliorated to conserve sage-grouse so that the species was no longer in danger of extinction or likely to become in danger of extinction in the foreseeable future. The COT report (U.S. Fish and Wildlife Service, 2013) was released in February 2013. It identified priority areas for conservation (PACs) and key habitats where conservation efforts should be emphasized for 39 sage-grouse populations across their range. Because both PACs and BLM priority habitat management areas (PHMAs) used State-provided key habitat maps as baselines, there was considerable overlap between the two.

A key listing factor in the 2010 warranted-but-precluded finding for greater sage-grouse was inadequate regulatory authority to address threats on public lands. Consequently, both BLM and the Forest Service amended land use plans across the range of sage-grouse. The NTT report (Sage-grouse National Technical Team, 2011) and the COT report (U.S. Fish and Wildlife Service, 2013) served as reference documents for this effort. These reports in turn built upon other reference documents including the WAFWA sage-grouse conservation assessment and strategy (Connelly and others, 2004; Stiver and others, 2006), the greater sage-grouse "Studies in Avian Biology" volume (Knick and Connelly, 2011b), and the U.S. Geological Survey's (USGS) "Conservation Buffer Distance Estimates for Greater Sage-Grouse—A Review" (Buffer Report; Manier and others, 2013).

Ninety-eight BLM and Forest Service land use plans covering 10 of the 11 western States containing greater sage-grouse (excluding Washington) were amended through 15 different environmental impact statements (EISs; records of decision can be accessed at https://www.blm.gov/programs/fish-and-wildlife/sagegrouse/blm-sagegrouse-plans, https://www.fs.usda.gov/sites/default/files/sage-grouse-great-basin-rod.pdf, and https://www.fs.usda.gov/sites/default/files/sage-grouse-rocky-mountain-rod.pdf. Although these amended plans differed to some degree, plans generally

- Identified PHMAs and general habitat management areas (GHMAs) for sage-grouse within each State where land use restrictions would be focused;
- Created sagebrush focal areas (SFAs) totaling approximately 4 million hectares (ha; 10 million acres) of particular importance to sage-grouse where no development or locatable mineral extraction would be permitted;
- Established disturbance thresholds within PHMAs (3 percent, except 5 percent in Wyoming);
- Set a density disturbance cap of one energy or mining facility per 259 ha (640 acres);
- Prohibited surface disturbance within 5 kilometers (km; 3.1 miles [mi]) of active leks, except in Wyoming, where surface disturbance was prohibited within 1 km (0.6 mi) in PHMA and 0.4 km (0.25 mi) in GHMA, and in South Dakota, where the lek buffer in GHMA was 1 km (0.6 mi);
- Stated that oil and gas leasing and development would be prioritized outside sage-grouse habitat;
- Designated that PHMA and GHMA were open to fluid mineral leasing but subject to no surface occupancy (NSO) stipulations with some State-specific exceptions. For example, in Wyoming, NSO applied within 1 km (0.6 mi) of a lek in PHMA and within 0.4 km (0.25 mi) of a lek in GHMA. In Colorado, areas within 1.6 km (1 mi) of a lek within both PHMA and GHMA were closed to leasing;
- Designated PHMA as exclusion areas and GHMA as avoidance areas for wind-energy development, except in Wyoming, where PHMA was designated as avoidance, and no restrictions were placed on GHMA;
- Designated both PHMA and GHMA as exclusion areas for solar energy development, with minor exceptions in GHMA in some States such as Oregon, North Dakota, and parts of Montana;
- Established hard and soft adaptive management triggers based on changes in habitat or sage-grouse populations, which, if tripped, would result in a causal factor analysis and potentially alter management prescriptions; and
- Required compensatory mitigation for projects that degraded habitat in PHMAs, and this mitigation included a standard for net conservation gain.

Several DOI Secretarial Orders (SO) and resultant strategies complemented the land use plan amendments and their efforts to conserve greater sage-grouse habitat. These included SO 3330 on "Improving Mitigation Policies and Practices of the Department of the Interior," SO 3336 on "Rangeland Fire Prevention, Management, and Restoration," the report to the Secretary "An Integrated Rangeland Fire Management Strategy" (IRFMS; U.S. Department of the Interior, 2015a), the "National Seed Strategy for Rehabilitation and Restoration" (Plant Conservation Alliance, 2015), "The IRFMS Actionable Science Plan" (Integrated Rangeland Fire Management Strategy Actionable Science Plan Team, 2016), and "Advancing Science in the BLM: An Implementation Strategy" (Kitchell and others, 2015). The net effect of the Federal actions summarized above, as well as State conservation and planning efforts summarized below, was a not warranted listing determination in October 2015 (U.S. Department of the Interior, 2015c).

The land use plan amendments were challenged immediately when the records of decision were published. The Governors of Utah and Wyoming protested the plans, while the Governors of Idaho, Nevada, North Dakota, South Dakota, and Utah requested changes through the consistency review process. Multiple lawsuits challenging the amendments were filed, including one by the State of Idaho and another joined by nine Nevada counties and the State of Nevada. In March of 2017, the U.S. District Court for the District of Nevada ruled that the BLM violated National Environmental Policy Act (NEPA; 42 U.S.C. 4321 et seq.) by failing to issue a supplemental EIS for the designation of SFAs and ordered them to prepare a supplemental EIS for SFAs in Nevada. To comply with that order, as well as Executive Order 13868 (Promoting Energy Independence and Economic Growth), SO 3349 (America's Energy Independence), SO 3353 (Greater Sage-grouse Conservation and Cooperation with Western States), and better align sage-grouse land use plan amendments with State sage-grouse management plans, the BLM (Bureau of Land Management, 2019e) and the Forest Service began the process to amend the 2015 land use plans for States that had objections with the original amendments. This process culminated in the issuance in March of 2019 of six records of decision adopting final resource management plan amendments to BLM's 2015 Sage-grouse Plans in California, Colorado, Idaho, Nevada/northeastern Oregon, Utah, and Wyoming (available at https://www.blm.gov/programs/fish-and-wildlife/sagegrouse/blm-sagegrouse-plans). Revisions to previous plans were not identical, but some or all of the amendments did the following:

- Eliminated SFAs from all revised plans, and these areas reverted to PHMAs;
- Removed compensatory mitigation requirement unless required by State policies or law and the related net conservation gain standard;
- Required design features to minimize impacts of projects in greater sage-grouse habitat were removed from GHMA in several States and changed from a requirement to something that can be applied as necessary and when appropriate in Wyoming, changed to voluntary best management practices in GHMA in Idaho, and dropped along with the GHMA designation in Utah;
- Removed the requirement to prioritize oil and gas leasing and development outside of GHMA in Wyoming and Idaho and reduced lek buffers within GHMA in Idaho from 5 km to 1 km (3.1 mi to 0.6 mi);
- Removed the requirement that exceptions, modifications, and waivers of NSO stipulations within PHMA required consent of FWS, in Colorado counties could determine whether these apply; and
- Reduced the size of lek buffers relating to surface disturbance in Colorado, Idaho, and Utah, and also in Colorado, lek buffers will be evaluated as opposed to will be applied.

The Forest Service also revised amendments to their 2015 land management plans, with the final EIS and draft records of decision published in August 2019 (available at https://www.fs.usda.gov/detail/r4/home/?cid=stelprd3843381, accessed August 9, 2020). The plans proposed by the Forest Service are complementary, but not identical, to the plans published by BLM. For instance, Forest Service proposed plans retain a compensatory mitigation requirement.

State-Level Restrictions on Land Use and Conservation Efforts in Sagebrush

The relative importance of State, versus Federal, protections for sage-grouse vary by State and across the distribution of sage-grouse. The Federal government owns 57 percent of the sagebrush biome, but the percentage of Federal ownership varies widely across States (table Q1). For example, Nevada's sagebrush acreage is 87 percent federally owned versus only 25 percent in Montana (table Q1). New Mexico and Arizona have about 3 percent of the sagebrush biome each (table Q1) but have no known sage-grouse populations and no sage-grouse specific restrictions on land use. States have regulatory authority over sage-grouse habitat on State-owned lands, which ranges from near 0 to almost 16 percent of sagebrush within States (table Q1). States also have regulatory authority on private lands but only for activities requiring State permits, such as oil and gas development and mining. Thus, States mostly affect sagebrush habitats primarily through nonregulatory, voluntary means. Most States within sage-grouse range have developed greater sage-grouse plans, which are generally backed by Executive Order, legislation, or both. Approaches to sage-grouse conservation vary but generally include some degree of land use restrictions in core habitats and a strong emphasis on habitat protection and restoration. Activities of private landowners on private land are generally unregulated. State-permitted activities such as oil and gas development and mining in sage-grouse habitats undergo, at a minimum, discussions with State wildlife agencies to avoid impacts to sage-grouse and provide compensation for impacts when they are unavoidable. Wyoming, for instance, has a core-area strategy that has been largely integrated into Federal land use plan amendments for sage-grouse but also applies to State land and State-permitted activities on private land. This strategy limits disturbance to 5 percent of the land area in core areas, and caps development to an average of 1 well pad per 259 ha (640 acres).

Most States within the range of sage-grouse are also developing or have developed mitigation programs to compensate for impacts of development on sage-grouse habitat that have not been avoided. These mitigation programs all include approaches to quantifying habitat conditions as a means of measuring debits (impacts) and credits (offsets). Currently, credits are usually provided through conservation banks that generally sell perpetual credits (for example, the Sweetwater River Conservancy; https://www.pathfinderranches.com/), or through habitat exchange programs that generally sell term-limited credits that temporally align with the debits being offset (for example, Montana's conservation program). These programs are mandatory for activities requiring a State permit in Montana, Nevada, Oregon, and Wyoming and are voluntary in Colorado, Idaho, Utah, and Wyoming. In States with voluntary mitigation constructs, compensatory mitigation is currently not required by the Federal Government on Federal lands. To date, voluntary mitigation programs in Nevada and Colorado have garnered little support from industry. Then Governor Sandoval of Nevada issued an Executive Order (2018-32) in December 2018, which ordered the Nevada Sagebrush Ecosystem Council to adopt regulations requiring compliance with the Nevada sage-grouse conservation plan and Nevada conservation credit system for anthropogenic disturbances on Federal or State land not otherwise avoided or minimized. On April 29, 2019, the Sagebrush Ecosystem Council adopted temporary rules to that effect (available at https://www.leg.state.nv.us/Register/2018TempRegister/T006-18A.pdf). Rulemaking by the Colorado Oil and Gas Conservation Commission pursuant to Senate Bill 19-181 (https://cogcc.state.co.us/documents/sb19181/Overview/SB_19_181_Final.pdf) may strengthen protections and mitigation efforts for sage-grouse in Colorado related to oil and gas development.

Table Q1. Hectares of sagebrush (*Artemisia* spp.), percent of biome-wide sagebrush, and land ownership of sagebrush by State. Calculated from the sagebrush distribution (Jeffries and others, 2019) and the U.S. Department of the Interior, Bureau of Land Management surface management layer (Bureau of Land Management, 2019c).

[The sum of Federal, State, Tribal, and private ownership does not always equal 100 percent for every State because of geographic uncertainties and mixed ownership. %, percent]

State	Total hectares	State total (%)	Federal (%)	State (%)	Tribal (%)	Private (%)
Arizona	1,968,736	3.0	39.9	6.9	42.6	10.6
California	2,104,771	3.2	97.7	1.4	0.2	0.0
Colorado	4,452,109	6.8	53.3	2.9	3.3	39.3
Idaho	6,888,974	10.6	67.5	6.6	2.6	23.2
Montana	6,823,370	10.5	25.3	7.9	4.9	61.8
Nevada	12,026,702	18.5	86.6	0.0	1.2	12.2
New Mexico	2,090,140	3.2	30.8	6.4	38.9	23.9
North Dakota	498,616	0.8	34.8	6.4	5.6	53.2
Oregon	7,066,806	10.9	64.5	3.0	0.6	31.9
South Dakota	352,063	0.5	8.9	15.5	0.0	75.5
Utah	6,601,505	10.1	61.2	9.0	4.4	25.4
Washington	1,683,469	2.6	17.9	13.2	11.9	57.0
Wyoming	12,574,383	19.3	45.0	7.6	3.2	44.0
Total	**65,131,642**	**100.0**	**57.4**	**5.4**	**5.3**	**31.8**

The quantification approach used by States with credit/debit mitigation constructs imparts value for habitat preservation, with some mitigation programs incentivizing (but not requiring) habitat restoration and enhancement by increasing the value of credits that result from these activities. Preservation credits inherently result in the loss of sagebrush habitats for all dependent and associated species through time because impacts can occur in one area while future impacts are prevented in another. Habitat preservation in and of itself does not offset an impact but can be used as a tool for ensuring durability of compensatory mitigation projects and has other value when the resources being preserved contribute significantly to ecological sustainability.

State habitat protection or restoration efforts include but are not limited to fee-title acquisitions of critical habitat, term or perpetual conservation easements, support for local working groups, and habitat improvement and restoration efforts, such as attempting to control invasive annual grasses and encroaching conifers, preventing or quickly extinguishing wildfires, postfire rehabilitation, reseeding, grazing management plans, and other efforts. In many States, including Colorado, Idaho, Montana, Nevada, Oregon, Utah, and Wyoming, habitat protection and restoration programs and efforts for or that can benefit sage-grouse are supported by millions of dollars appropriated by the legislature or through lottery funding. FWS Candidate Conservation Agreements with Assurances with landowners for greater and Gunnison sage-grouse are in place in Colorado, Montana, Nevada, Oregon, Utah, and Wyoming.

Sage-Grouse Conservation as an Umbrella

Land use restrictions and mitigation or conservation programs, such as those described above, when implemented for widely distributed species such as sage-grouse may serve as an “umbrella” and simultaneously benefit other species of conservation concern (Rich and Altman, 2001; Rowland and others, 2006; Hanser and Knick, 2011; Carlisle and others, 2018a; Runge and others, 2019; Pilliod and others, 2020a). Such conservation by proxy is appealing given the limited resources available for the management of all sagebrush-associated species of concern. The degree to which sage-grouse may serve as an umbrella species for sagebrush-dependent or -associated species depends on how large the sage-grouse umbrella is and how porous the umbrella is in protecting or conserving other species (Carlisle and others, 2018a). In other words, to what extent do land use restrictions and conservation practices established for sage-grouse overlap with core habitats and ranges of other sagebrush species and to what extent are land use restrictions or conservation practices implemented for sage-grouse effective at protecting or restoring habitat for other sagebrush species.

Size of the Sage-Grouse Umbrella

At broad spatial scales (regional and landscape), efforts protecting sagebrush ecosystems from further conversion, fragmentation, and development (for example, disturbance caps) indirectly benefit other sagebrush-associated species such as passerines (Donnelly and others, 2017) and mule deer (*Odocoileus hemionus*; Copeland and others, 2014). The extent of those benefits, however, will depend upon the amount of spatial overlap between sage-grouse and other species, which can vary greatly (for example, see Rowland and others, 2006, for the Great Basin). More important than overall spatial overlap is the degree to which areas prioritized for sage-grouse conservation overlap with core areas or important habitats for sagebrush-dependent species (for example, see Carlisle and others, 2018a, for Wyoming). Sage-grouse, for instance, do not occur in Arizona or New Mexico (fig. Q1). Land use restrictions to benefit sage-grouse generally only apply in PHMAs on Federal land or to their State equivalent, such as core areas in Wyoming. These areas are highly important to sage-grouse, but they constitute only 33 percent of occupied greater sage-grouse range (fig. Q1) and only 26 percent of areas classified as having 5 percent sagebrush cover or more (fig. Q2). Thus, if these restrictions are protective of other sagebrush wildlife species, the extent will be limited to PHMAs on Federal land or State equivalents.

Protections for sage-grouse through land use restrictions on non-Federal land vary widely, and where they occur, they are generally restricted to permitted activities on private land and to State lands. Regulatory approaches or land use restrictions are not effective tools at stopping the spread of invasive plant species such as cheatgrass (*Bromus tectorum*),

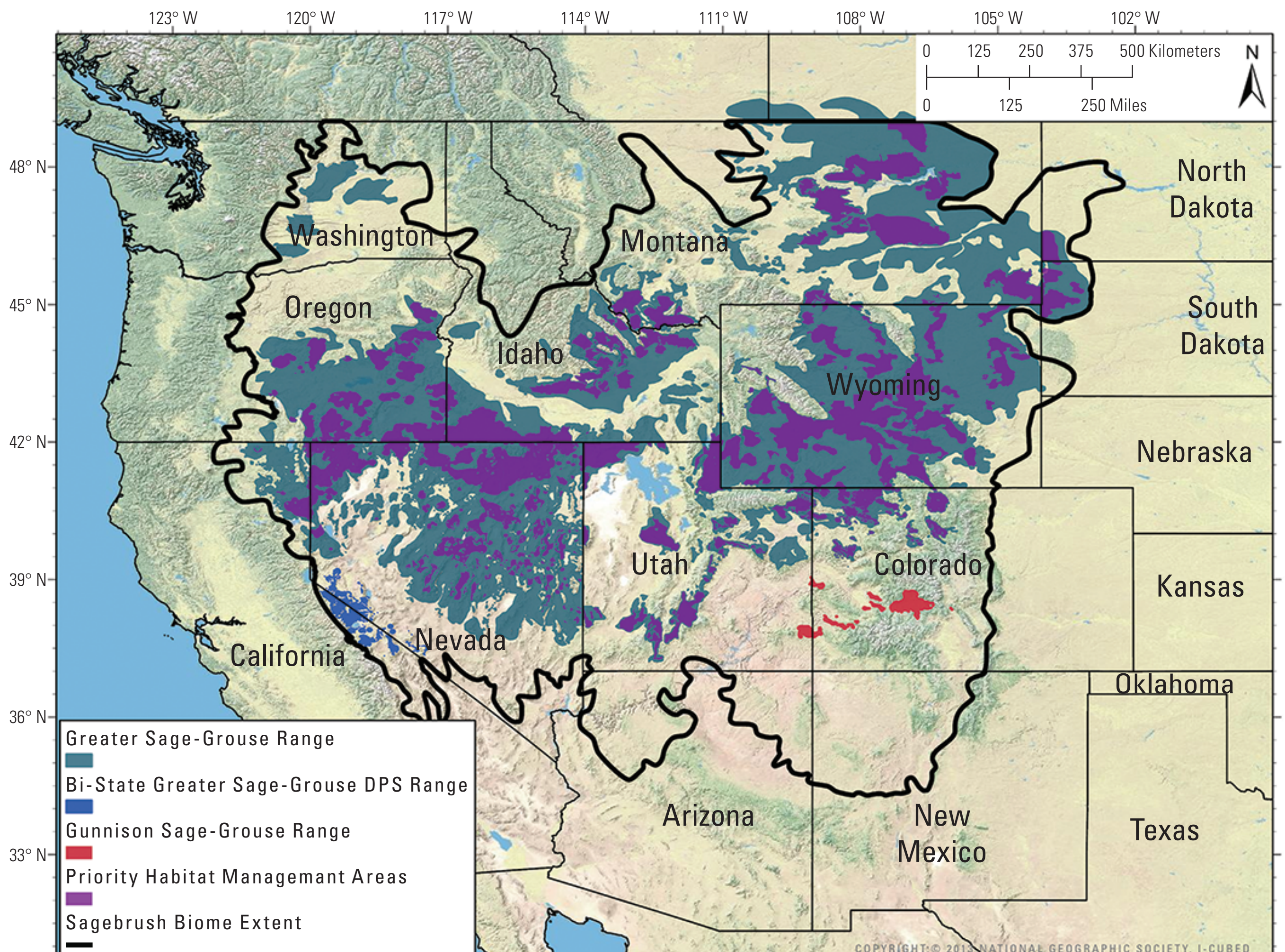

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Figure Q1. Greater sage-grouse (*Centrocercus urophasianus*), including the Bi-State Distinct Population Segment (DPS; U.S. Fish and Wildlife Service, 2014), and Gunnison sage-grouse (*C. minumus*; Braun and others, 2014) ranges within the sagebrush (*Artemisia* spp.) biome (Jeffries and Finn, 2019) and priority habitat management areas designated for greater sage-grouse (Bureau of Land Management, 2019b).

preventing wildfires, or reducing habitat degradation by free-roaming equids, the three dominant threats to sagebrush in the western portion of the biome (chap. J, this volume; chap. K, this volume; chap. N, this volume). The limited scope and scale of efforts to address invasive plant species and wildfire for sage-grouse or for the sagebrush ecosystem relative to the need is perhaps the greatest limitation on the utility of the sage-grouse umbrella.

In addition, the efficacy of sage-grouse as a conservation umbrella will vary by species depending on the degree of habitat similarity within the overlapping ranges (Rowland and others, 2006; Hanser and Knick, 2011; Sage Grouse Initiative, 2015b; Pilliod and others, 2020a). Comparisons of overlap in land cover associations and spatial overlap between sage-grouse and target vertebrate species showed sage-grouse could serve as an effective umbrella species for sagebrush obligates but far less so for other species associated with sagebrush (Rowland and others, 2006).

Among birds, Brewer's sparrow (*Spizella breweri*), sagebrush sparrow (*Amphispiza belli*), and sage thrasher (*Oreoscoptes montanus*) were most abundant where sage-grouse were most abundant (as measured by lek density and size), indicating that conservation practices for sage-grouse in these locations would serve as an umbrella for these species (Donnelly and others, 2017). Ten of 13 passerine species had at least moderate overlap along four environmental gradients with sage-grouse, indicating that the broad diversity of sagebrush habitats used by sage-grouse, if conserved, may serve as an effective umbrella for them (Hanser and Knick, 2011). However, this was not true for Savannah (*Passerculus sandwichensis*) and grasshopper sparrows (*Ammodramus savannarum*), two primarily grassland species, or the

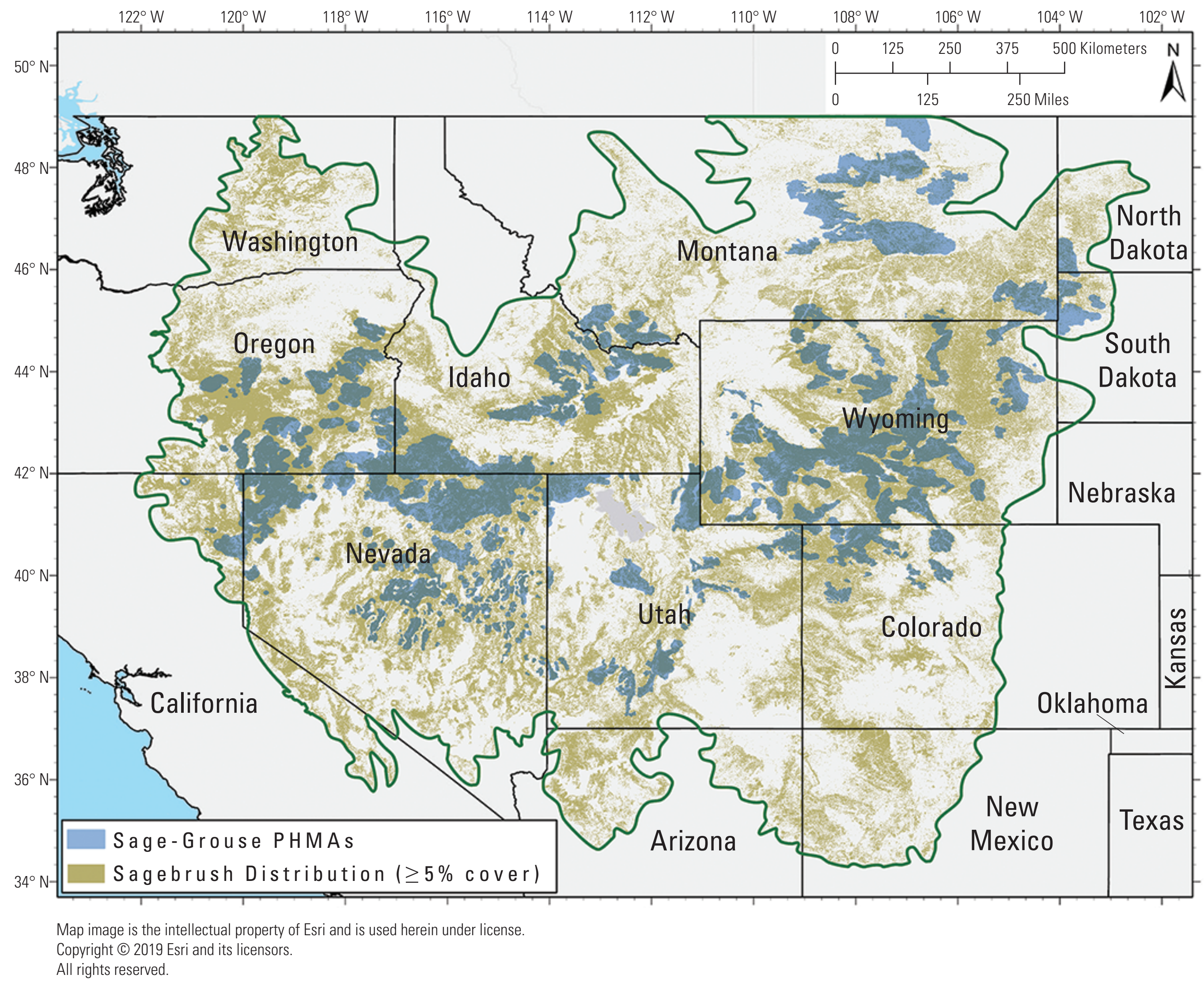

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Figure Q2. The overlap of priority habitat management areas (PHMAs) designated for greater sage-grouse (*Centrocercus urophasianus*; Bureau of Land Management, 2019b) and the distribution of sagebrush (*Artemisia* spp.) vegetation cover greater than or equal to (≥) 5 percent (Jeffries and others, 2019). %, percent.

green-tailed towhee (*Pipilo chlorurus*), a woodland/shrubland ecotone species (Hanser and Knick, 2011). Important year-round habitats used by sage-grouse within the greater Wyoming Basins (Hanser and others, 2011a) effectively capture habitats predicted to support breeding sage-thrashers with moderate capture of sagebrush sparrow and Brewer's sparrow breeding habitat (Aldridge and others, 2011). Among mammals, both minimum occupied area (MOA; 93 percent) and primary habitat (91 percent) of pygmy rabbits (*Brachylagus idahoensis*) overlapped extensively with sage-grouse range (Smith and others, 2019), even though pygmy rabbits were documented in only approximately 4 percent of the sage-grouse distribution (Smith and others, 2019). Areas with some level of land use restrictions on Federal land (that is, PHMAs) for sage-grouse make up a much smaller percentage of pygmy rabbit MOA (61 percent) and primary habitat (53 percent; fig. Q3; analysis methods in Smith and others, 2019; updated to the 2019 PHMA layer [Bureau of Land Management, 2019b]).

Ungulates that rely on sagebrush landscapes for migration and seasonal habitats also overlap with sage-grouse. Conservation measures in the Upper Green River area of Wyoming overlapped with 66–70 percent of mule deer migration corridors, 74–75 percent of stopover habitat, and 52–91 percent of wintering areas, with about half of the benefit attributed to the core area policy and private conservation easements directed toward sage-grouse (Copeland and others, 2014). Similarly, migration pathways for sage-grouse and pronghorn (*Antilocapra americana*) overlapped substantially in the Northern Great Plains, and over 50 percent of pronghorn corridors were within State and federally designated sage-grouse core areas (Tack and others, 2019).

Among reptiles, 22 of 70 species (31 percent) have greater than 10 percent of their distribution area within the sage-grouse range, and 14 of these (8 snake and 6 lizard species) have relatively similar habitat associations to those of sage-grouse (Pilliod and others, 2020a). The pygmy short-horned lizard (*Phrynosoma douglasii*) had the highest percent of its distribution within the current range of the greater sage-grouse at 63 percent and the highest similarity of habitats to sage-grouse. Eight other reptile species had relatively small ranges which overlapped from 25 to 40 percent with greater sage-grouse distribution, indicating sage-grouse conservation efforts could affect their habitat conditions. Many reptile species had large ranges which included much of the greater sage-grouse range and thus would be influenced by management actions directed at sage-grouse in areas of overlap. For example, the common sagebrush lizard (*Sceloporus graciosus*) was predicted to occur within 75 percent of the range of sage-grouse, but 62 percent of the common sagebrush lizard distribution lies outside of the sage-grouse range (Pilliod and others, 2020a). Important year-round habitats used by sage-grouse within the greater Wyoming Basins (Hanser and others, 2011a) effectively capture habitats predicted to support greater short-horned lizards (*Phrynosoma hernandesi*; Hanser and others, 2011b; Carlisle and others, 2018a).

Among amphibians, 27 amphibian species overlap with the range of the sage-grouse by more than 10 percent, but only the Great Basin spadefoot (*Spea intermontana*) occurs predominantly in PHMAs and lives in sagebrush habitats for most of its life cycle (chap. I, this volume; Rowland and others, 2006). In Wyoming, the Great Basin spadefoot is expected to benefit from core areas managed for sage-grouse, whereas other species, such as the plains spadefoot (*Spea bombifrons*), northern leopard frog (*Lithobates pipiens*), and Columbia spotted frog (*Rana luteiventris*), might be harmed if development is shifted to areas outside of the sage-grouse reserve system (Carlisle and others, 2018a). The range of the Columbia spotted frog overlaps with that of sage-grouse substantially (about 75 percent of the frog's range), but this frog is closely associated with surface water and conservation measures to protect or restore habitat are unlikely to affect this species (chap. I, this volume). Riparian or wet meadow habitat management for brood-rearing sage-grouse may positively affect amphibians.

Porosity of the Sage-Grouse Umbrella

At finer spatial scales, the sage-grouse may not be a suitable umbrella species for other sagebrush-dependent species. The preferred nesting habitats of sage-grouse, for example, do not coincide with those of two of the three songbirds that are sagebrush obligates, Brewer's sparrow and sage thrasher, that prefer areas with taller shrubs and higher shrub cover (Chalfoun and Martin, 2007; Carlisle, 2017). The maintenance of structural heterogeneity of sagebrush habitats within landscapes is therefore critical for sage-grouse management to successfully benefit other sagebrush species (Hanser and Knick, 2011). Animals as diverse as pygmy rabbits and Great Basin spadefoot have additional habitat requirements relative to sage-grouse, including soils that are suitable for burrow construction, and may use different microhabitats even when their general habitats overlap those of sage-grouse. Given this and a patchy distribution throughout their range, targeted management decisions at regional scales will likely be necessary to ensure that sage-grouse-focused habitat conservation or restoration has the best chance of also enhancing habitat for pygmy rabbits (Smith, 2019) and other sagebrush wildlife species.

Some protections are likely to be more applicable to some sagebrush wildlife species than others. For instance, NSO stipulations or surface disturbance caps that prevent or reduce disturbance in large blocks of sagebrush within PHMA will likely be more protective to other species than narrow 1-km (0.6-mi), lek-based buffers. Other species may be more or less tolerant of noise, traffic, and other disturbances associated with developments, so 3 or 5 percent disturbance caps or a threshold of 1 well pad per 259 ha (640 acres) chosen for sage-grouse may or may not result in continued use of those areas or maintain demographic processes by other species. For example, provisions of surface disturbance allowed by the

Wyoming core area policy do not fully prohibit disturbance levels that can impact mule deer migration (Copeland and others, 2014).

Conservation practices that protect or restore functional sagebrush plant communities, such as conservation easements, invasive plant control, fire prevention, and restoration performed to benefit sage-grouse may have greater utility for other sagebrush-dependent and -associated species than land use restrictions. These practices will likely benefit species that depend on intact and well-connected sagebrush plant communities if they are of sufficient extent. Sage-grouse management has also historically included active habitat management and manipulation, including burning, mowing, and herbicidal treatments to reduce sagebrush cover and encourage the release of the herbaceous understory to benefit sage-grouse during brood-rearing. State-sponsored compensatory mitigation programs attempt to replace sage-grouse habitat loss from development by habitat improvements elsewhere. Practices designed to satisfy sage-grouse seasonal habitat needs are unlikely to benefit, and may in fact negatively impact, species with dissimilar habitat needs. For example, reductions in sagebrush cover tend to decrease the local abundance of sagebrush-obligate songbirds and lead to the complete loss of nesting habitat for shrub nesters within the affected area (Lukacs and others, 2015; Carlisle and others, 2018b). Other types of habitat treatments, however, such as conifer reduction, may benefit several species of sagebrush birds and other wildlife (Knick and others, 2014a; Holmes and others, 2017; but see Bombaci and Pejchar, 2016). Mule deer have complex

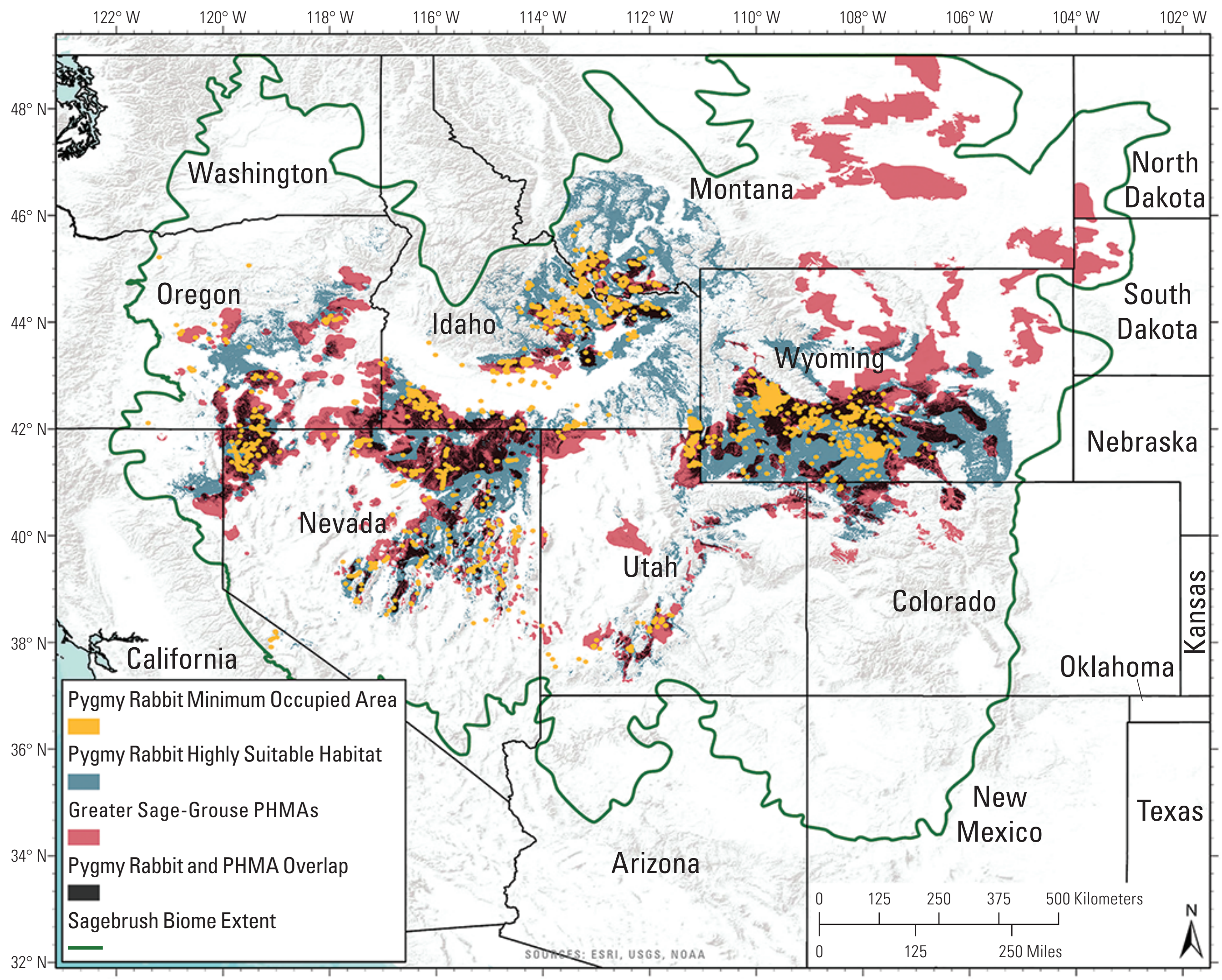

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Figure Q3. Pygmy rabbit (*Brachylagus sylvilagus*) minimum occupied area and primary habitat (Smith and others, 2019), and priority habitat management areas (PHMAs) established for greater sage-grouse (*Centrocercus urophasianus*; Bureau of Land Management, 2019b).

relationships with juniper (*Juniperus* spp.) woodlands, selecting areas with higher canopy cover of western juniper (*J. occidentalis*) in Oregon (Coe and others, 2018), yet experiencing enhanced body condition (Bender and others, 2013) leading to higher fawn survival (Bergman and others, 2014a) when junipers were removed. A literature review of impacts of reducing pinyon (*Pinus* spp.)-juniper woodlands on sagebrush obligates and sagebrush-associated species found that an equal proportion of studies reported either negative or nonsignificant results on native ungulate populations (Bombaci and Pejchar, 2016). However, these treatments may negatively affect populations of pinyon-associated species, such as pinyon jays (*Gymnorhinus cyanocephalus*; chap. M, this volume).

In addition to treatments negatively impacting habitat for other species, critical habitats for other sagebrush-associated species that fall outside sage-grouse core areas or PHMAs (such as migratory corridors or winter ranges for mule deer; Copeland and others, 2014) may be subject to increased development displaced by restrictions within sage-grouse core areas or PHMAs. In addition, the singular focus on sage-grouse management by land and wildlife management agencies may withdraw financial and other resources that could be devoted to conservation efforts for other species.

The sagebrush biome has had a long and varied management history. From past attempts to remove sagebrush to current attempts to protect and restore it from a myriad of threats, management of the sagebrush biome has been undergoing near constant change since European settlement in the early 1800s. Current attempts to manage the remaining sagebrush ecosystems for wildlife by focusing on priority or important habitat for sage-grouse have the potential to benefit other species in the sagebrush biome as well. However, species whose ranges are outside of the sage-grouse PHMAs or whose microhabitat requirements may be different from those of sage-grouse can be negatively impacted by sage-grouse focused management. There are greater than 735 species of plants, vertebrates, or invertebrates inhabiting the sagebrush biome in the Great Basin alone (Wisdom and others, 2005), and for many vertebrates and a few invertebrates, there is some degree of conservation concern (ESA listed and species of greatest conservation need). Conservation of this broad assemblage will require identifying and prioritizing sagebrush landscapes based on ecological principles such as resistance and resilience, degree of intactness and connectivity, and patch size, among others, and then layering on species-specific requirements not otherwise met such as sage-grouse and pygmy rabbit core areas and mule deer migration corridors.

Chapter R. Restoration

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Executive Summary

Vast expanses of the sagebrush (*Artemisia* spp.) ecosystem have been degraded by disturbances, including plant invasions, wildfire, and improper grazing, necessitating restoration efforts to maintain wildlife habitats, reduce future wildfire risks, and recover ecosystem services. Restoration treatments, such as conifer removal, seeding, and herbicide applications, have been extensively applied. However, treatment success has been mixed, and many other acres are degraded or are at risk but have not been treated. A primary objective of restoration in sagebrush communities is to maintain or increase desirable perennials, such as sagebrush and forbs, that are key to wildlife, along with perennial grasses that provide resistance to invasion and resilience to future disturbance. This objective is challenging because of variable environmental conditions, including frequent drought, exotic plant invasions, recurrent wildfire, and inadequate postfire grazing management. Additional challenges include the large extent of areas that need treatments, lack of basic site information, and logistical challenges to treatment application. Moreover, restoration efforts have typically been short-term, single applications. Restoration planning now emphasizes prioritizing areas that need intervention and are likely to have a positive response. Treatment success is likely to improve in the future given prioritization of sites, adaptive management approaches that incorporate learning, and the involvement of multiple stakeholders that allows for repeated interventions over longer time periods. While current research is improving the understanding of factors affecting restoration success and restoration techniques, there are clear opportunities to better incorporate current knowledge into restoration practice.

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Introduction

Restoration of habitats that have been altered by anthropogenic and natural disturbance and the introduction of invasive plant species is a significant concern in sagebrush (*Artemisia* spp.) ecosystems. Restoration, broadly defined here to include rehabilitation and reclamation, is most feasible when objectives are clear, residual ecosystem components are present, and environmental conditions are favorable. However, severe loss of biotic diversity and ecosystem functioning creates substantial obstacles for recovery to original or even desirable alternative, stable vegetation states. This is the case for sagebrush habitats, especially those in the warmest and driest regions of the biome. Nearly half of the estimated 651,316 square kilometers (km^2; 251,473 square miles [mi^2]) of sagebrush habitats in the western United States are now in a disturbed and degraded condition that is undesirable for wildlife, livestock, and other land uses (Miller and others, 2011). Impacts from surface disturbing activities, such as energy development and mining, are often compounded by wildfire, nonnative plant invasions—particularly annual grasses that fuel increased wildfire, and improper grazing in these habitats.

Following European settlement, historical land treatments in the sagebrush biome often focused on shifting vegetation communities from shrub to grass dominance to increase forage production for domestic livestock (Knick, 2011). Prescribed fire, mechanical, and herbicide treatments were used to remove sagebrush and favor seeded bunchgrasses. These bunchgrasses were typically introduced Eurasian species, such as crested wheatgrass (*Agropyron cristatum*) and Siberian wheatgrass (*A. fragile*). Sagebrush removal continued into the late-1970s at low elevations and into the mid-2000s at high elevations until wildfire and wildlife (for example, greater sage-grouse [*Centrocercus urophasianus*]) concerns increased. Since then, managers and researchers have been developing new treatments and seed sources to improve the ecological condition of sagebrush communities with an emphasis on the conservation of native wildlife species. However, documenting the types and locations of treatments and their outcomes has only begun to be standardized and tractable (Pilliod and others, 2017b).

The complex suite of threats to sagebrush communities has led to an increase in the size, number, and type of restoration treatments used by land management agencies, as well as an increased use of sagebrush and other native seed in those treatments (fig. R1; Pilliod and others, 2017b; Copeland and others, 2018). To meet restoration challenges, the management and science communities have developed tools for prioritizing resources across sagebrush ecosystems and strived to increase the efficiency and effectiveness of treatments.

Increasing restoration success requires identifying the necessary knowledge, strategies, and tools for implementation. While seemingly straightforward, diverse management objectives, scale issues, and a range of conditions and habitats increase the challenges associated with developing successful strategies for sagebrush habitat restoration. This chapter addresses restoration planning at several spatial scales and provides an overview of approaches and tools for increasing the likelihood of success. Appendix R1 provides additional sources of information on restoration in general and details specific to the sagebrush biome.

While the term “restoration” is often used broadly in sagebrush and other ecosystems, some agencies and practitioners recognize specific meanings for rehabilitation or reclamation. Restoration means bringing an ecosystem back to an original state of structure (for example, native species) and function (for example, nutrient cycling, erosion prevention, and primary production; Bradshaw, 2002). In contrast, rehabilitation aims to reinstate part of the original structure with a focus on recovering ecosystem functions, and reclamation focuses on restoring ecosystem function often with little regard to structure.

Postfire rehabilitation in sagebrush ecosystems has been guided and implemented by the U.S. Department of the Interior, Bureau of Land Management’s (BLM) Emergency Stabilization and Rehabilitation (ESR; (Bureau of Land Management, 2007); see “BLM Emergency Stabilization and Rehabilitation (ESR) Program” sidebar) and the U.S. Department of Agriculture (USDA) Forest Service’s Burned Area Emergency Response (BAER; National Interagency Fire Center, 2020) programs on public lands. The planning and evaluation of BLM ESR and Forest Service BAER treatments typically involve interagency and stakeholder partners. Rehabilitation and restoration actions are also planned and implemented by private landholders, municipalities, counties, State-level agencies (for example, Utah’s Watershed Restoration Initiative [WRI; https://wri.utah.gov/wri/]) and nongovernmental organizations (NGOs). In addition, the USDA Natural Resources Conservation Service (NRCS) frequently provides consultation and support to these groups as well as cost-share funds for affected private landowners. In contrast, reclamation is often regulated by governmental agencies but implemented by single private entities or their contractors.

In general, restoration will be effective, efficient, and engaging if (1) efforts are referenced to native ecosystems and consider their response to broader environmental changes (chap. L, this volume); (2) key ecosystem attributes are identified prior to developing goals and objectives; (3) natural recovery is augmented with assisted recovery when ecosystems are impaired; (4) actions aim towards full recovery; (5) practices are based on all pertinent knowledge, which can be enhanced with science and technology transfer and outreach to restorationists; and (6) efforts genuinely engage stakeholders early and actively in the restoration process (McDonald and others, 2016).

Restoration for Wildlife Conservation

The goal for many restoration projects is improvement of wildlife habitats. Projects aimed at restoring wildlife habitats should include (1) information on species distributions and population abundances; (2) information about baseline habitat conditions; (3) an understanding of the most important habitat features required for species colonization or persistence; (4) specific objectives for focal species and benchmarks that define restoration success; (5) a monitoring plan; and (6) comparisons with nearby unrestored sites, either representing baseline or target conditions (Borgmann and Conway, 2015). Additionally, monitoring wildlife response is crucial. The suitability of restored habitats for wildlife can be influenced by conditions at multiple spatial and temporal scales, as is addressed in more detail in the “Sagebrush Restoration” section of this chapter. Consideration of landscape context, size of restored areas, and time since restoration can assist in

BLM Emergency Stabilization and Rehabilitation (ESR) Program

The ESR program’s objectives are to (1) stabilize fire-damaged sites in order to protect life and property and prevent further degradation of burned areas, and (2) rehabilitate lands that have a low probability of recovering on their own (Bureau of Land Management, 2007). Emergency stabilization may be achieved by repairing structures crucial to public health and safety, minimizing erosion, applying treatments to critical habitat for species of concern, protecting cultural resources, and mitigating invasive plants. Burned area rehabilitation focuses on longer term treatments, such as noxious weed removal, ecosystem recovery, tree planting, and repairing damage to less critical facilities. Proposals must be submitted to the ESR program within 21 days after fire containment and address a broad range of topics including treatment details and cost estimates. In years when many fires occur, budgets may limit the number of treatments that can be implemented in their entirety.

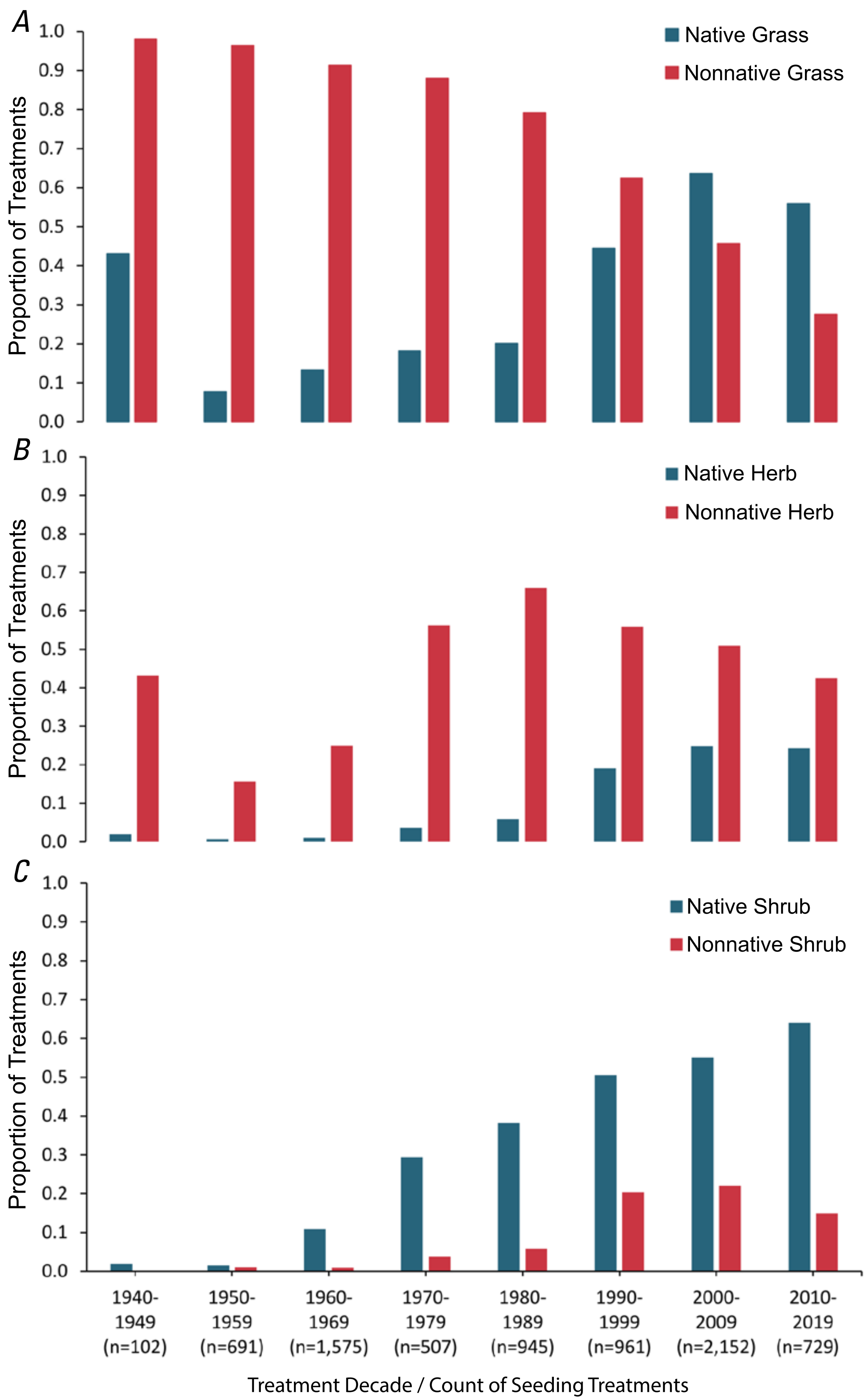

Figure R1. Proportion of seeding treatments on U.S. Department of the Interior, Bureau of Land Management lands with seed mixes that *A*, have at least one native or nonnative grass, *B*, native or nonnative forb, and *C*, native or nonnative shrub (Pilliod and Welty, 2013). This is not a comprehensive sample of all treatments in the area for the entire time period, as not all seeding treatments contain explicit seed lists, and data entry is not yet complete for seedings from 2015 to 2019. n, number.

understanding and predicting wildlife responses to restoration (Ortega-Álvarez and Lindig-Cisneros, 2012). A prioritization process can guide site selection that will result in the biggest gains for focal species (see "Landscape-Level Characterization and Prioritization and Project-Level Prioritization and Planning" sections of this chapter; Pyke, 2011; Reinhardt and others, 2017; Ricca and others, 2018).

A central consideration regarding the value of restoration for wildlife is how best to make habitats functional for a suite of animal species. The umbrella species concept has been invoked for multispecies management, particularly when implementing treatments focused on greater sage-grouse (Rowland and others, 2006; Hanser and Knick, 2011; Carlisle and others, 2018b; chap. Q, this volume). There are species that will likely benefit from this approach solely because some of their needs will be met indirectly by protecting and restoring large areas and providing resources for umbrella species. However, because habitats are inherently species-specific, restoration aimed at providing habitat for one species may not necessarily result in suitable habitat for another species. For example, conifer removal can positively affect sagebrush-dependent wildlife species (see review of studies by Bombaci and Pejchar, 2016; Holmes and others, 2017; Knick and others, 2017; Peterson and others, 2017; chap. L, this volume) but negatively affect pinyon jays (*Gymnorhinus cyanocephalus*; Johnson and others, 2018). Restoration implemented under the umbrella species concept is mostly applicable at broad scales (Carlisle and others, 2018b), but, at local scales, restoration efforts may need to target particular species or guilds, and species-specific habitat requirements should be considered.

Criteria for successful restoration of wildlife habitats, including measurable changes in habitat structure, proximity of suitable habitat to treatment area, availability of colonists, provisioning of resources for focal species, and the importance of increasing animal productivity and avoiding population sinks or ecological traps, are outlined in Hale and Swearer (2017; fig. R2). Information on wildlife responses to sagebrush restoration are also detailed in Dahlgren and others (2006), Johnson and Chalfoun (2013), Petersen and others (2016), Severson and others (2017a, b), and Smith and Beck (2018).

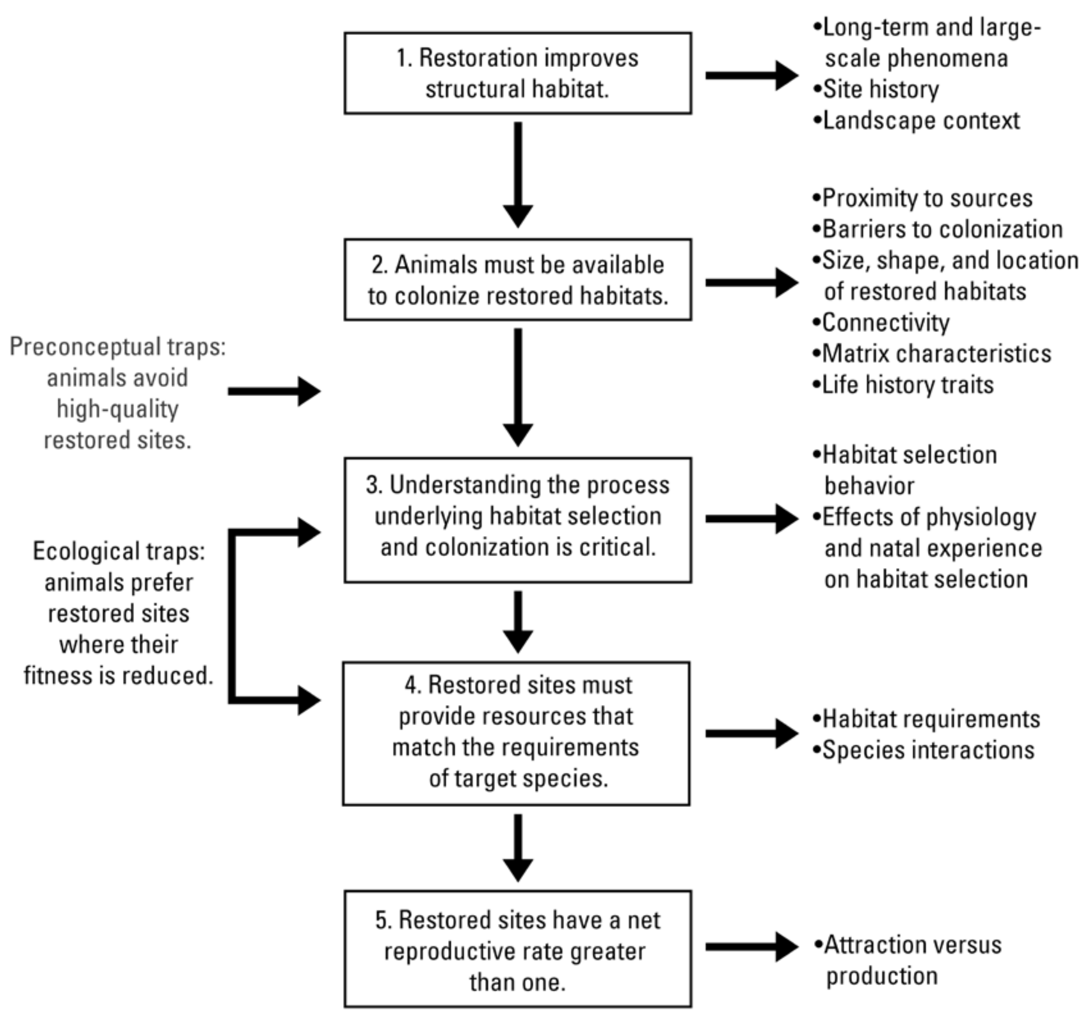

Figure R2. Five critical criteria for ensuring habitat restoration is successful (modified from Hale and Swearer, 2017).

Sagebrush Restoration

The vast extent of sagebrush communities and the limited resources available for restoring degraded sagebrush habitats have led to development of processes and information to help evaluate and prioritize areas for restoration (fig. R3). Prioritizing restoration activities across large spatial extents helps ensure that treatments are placed within the context of the surrounding landscape and maximize their effectiveness. After the landscape is prioritized, individual projects and treatments are prioritized to achieve project objectives and overall landscape objectives. Evaluation of outcomes and implementation of adaptive management throughout this process helps improve the efficiency and effectiveness of restoration actions. The following sections discuss each step of this process, give examples of ongoing activities, and provide resources to help with implementation.

Landscape-Level Characterization and Prioritization

At the landscape scale, fragmentation of large, continuous habitat patches into smaller, discrete units is a significant problem, and restoration investments are critical to reconnect habitat patches or maintain continuous landscapes. Landscape units often differ in merit for investment relative to their wildlife values, need for restoration actions to advance recovery, and recovery potential. Moreover, a complexity of multiple stressors, jurisdictions, land uses, stakeholders, and economic development investments usually exists. Thus, restoration at the landscape scale (ecoregion to planning unit) requires collaboration with partners across jurisdictional boundaries and addresses: (1) extent and connectivity of sagebrush patches or spatial resilience, (2) resource values such as habitat for wildlife or threatened and endangered species, (3) relative resilience to disturbance and resistance to invasive plants, and (4) disturbances or land uses that may impact restoration outcomes.

Tools and knowledge are increasingly available to help link landscape characteristics, such as vulnerability to invasive plants and resilience to disturbance, to both landscape recovery potential and importance for wildlife populations (Knick and others, 2013; Chambers and others, 2014a, 2017a, b; Doherty and others, 2016; Ricca and others, 2018; Crist and others, 2019). Habitat fragmentation impacts the function of the sagebrush and other ecosystems and ability to recover following disturbance (Holl and Aide, 2011; Knick and others, 2013). Functional connectivity is the ability of a landscape to support movement (that is, dispersal and migration) and is necessary to support local populations (Knick and Hanser, 2011; Crist and others, 2017). Examples include the need for a landscape to contain and support movement of sage-grouse within and between seasonal habitats and the ability of mule deer (*Odocoileus hemionus*) to migrate from summer to winter range. Maps are particularly useful to show the spatial variability in (1) the locations and connectivity of high-value resources and habitats and (2) the resilience and resistance for

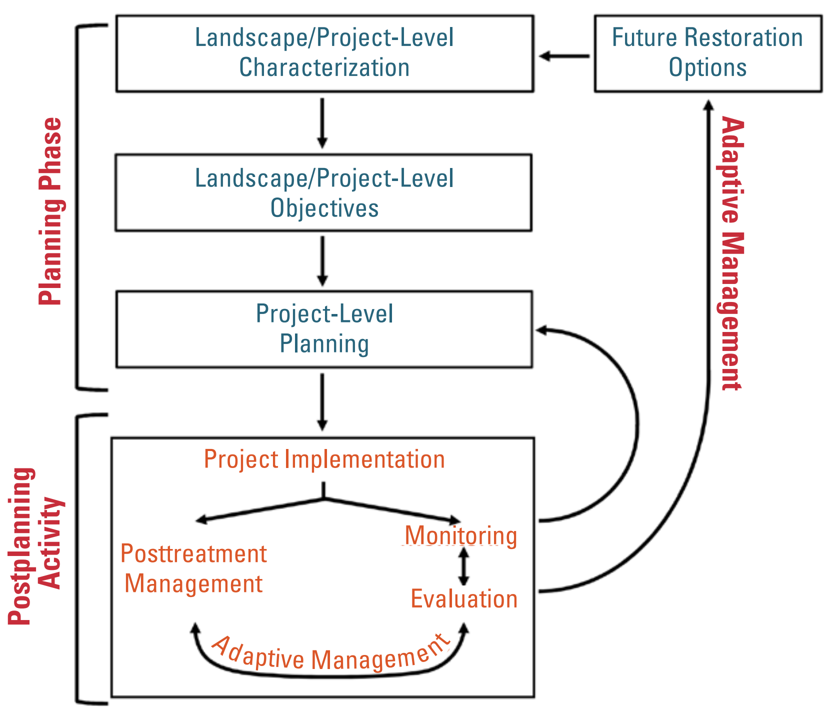

Figure R3. Workflow of an idealized restoration process that includes setting landscape and project objectives, determining monitoring protocols and design, selecting project-level treatments, and then ensuring regulations are met.

prioritization of locations to improve habitat patches, reduce threats to habitat patches, and reduce fragmentation (for example, Chambers and others, 2017a, b; Ricca and others, 2018). Assessing likelihood of successful habitat improvement and prioritization of the investments may lead to increased connectivity and overall functioning of the landscape.

Landscape-scale restoration implementation is often a collaborative or multiproject effort. Only managers of large land units at the watershed or larger scale, such as the BLM or Forest Service, are able to develop large projects solely on lands under their jurisdiction, and even then it may take multiple years to complete provided that agency resources are available. When land ownerships vary, timing and coordination requirements can be difficult to achieve.

Project-Level Prioritization and Planning

At local scales, wildlife habitat suitability is affected by both structural and floristic characteristics of vegetation. Thus, a focus on restoring appropriate vegetation structure and species composition is critical for creating functional wildlife habitats. Structural characteristics of vegetation are important in providing thermal and escape cover (McAdoo and others, 2004; Coates and others, 2017b). Key structural attributes in sagebrush include cover and height of trees, shrubs, forbs, and grasses; plant density; bare ground; and litter depth. Floristic characteristics include species composition, richness,

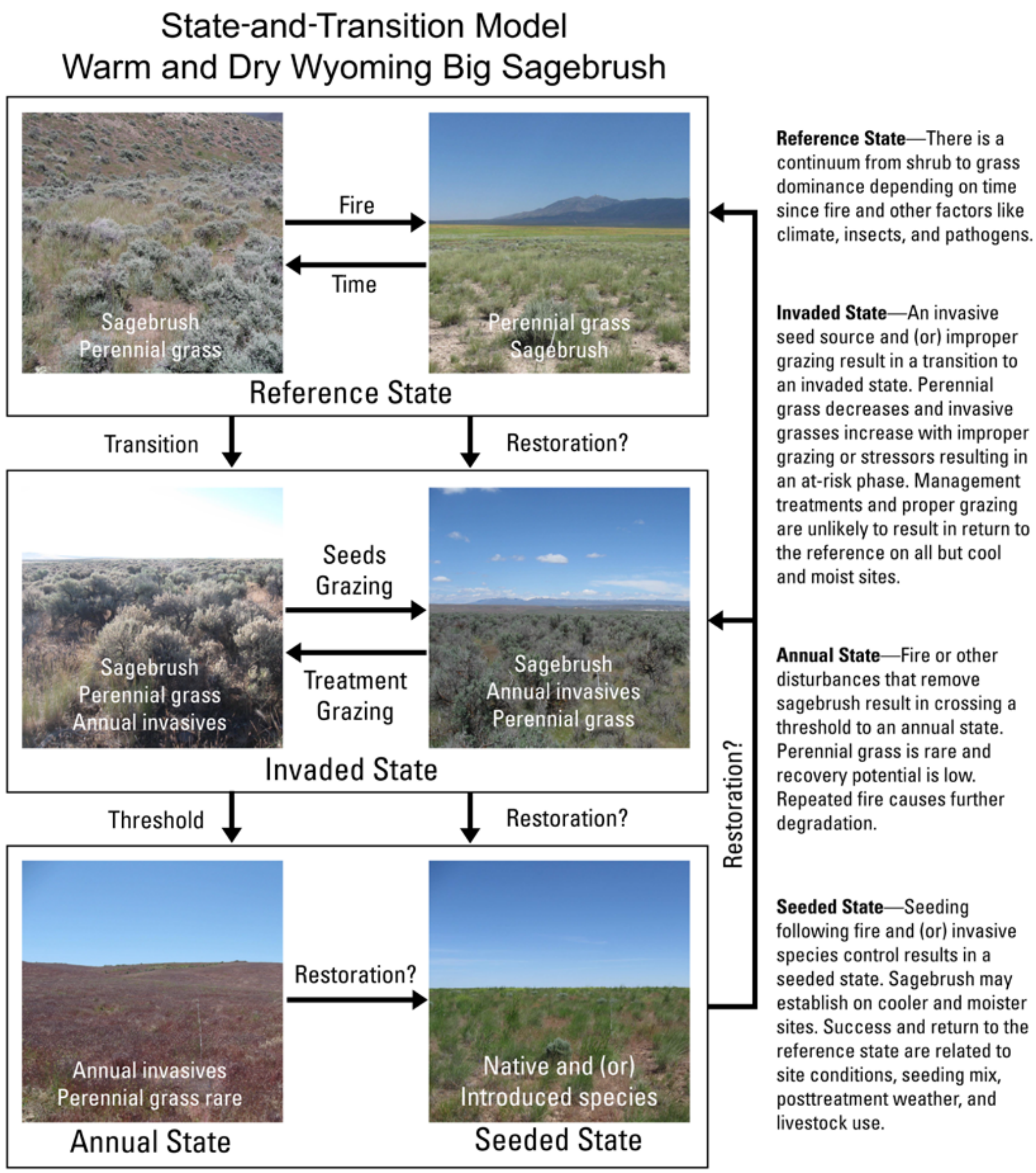

Figure R4. Generalized state-and-transition model for a Wyoming big sagebrush (*Artemisia tridentata wyomingensis*) community. Note that the reference condition or state is dynamic and depends on many temporal factors (modified from Pyke and others, 2015b).

Resilience and Resistance

Sagebrush landscapes differ significantly in terms of their relative resilience to disturbance and resistance to invasive annual grasses (Chambers and others, 2014a, c, 2019b, c). Information on how resilience and resistance differ across landscapes can assist managers in evaluating the recovery potential of an area and thus the magnitude of the investment needed for successful restoration (Chambers and others, 2017a). Areas with high resistance to invasive annual grasses and native plant persistence often have the potential to recover without intervention following fire or other disturbance. Seeding or transplanting of native species following surface disturbance usually results in successful establishment, especially at cooler, moister sites at higher elevations. Areas with moderate resilience and resistance also typically recover unassisted following wildfire, especially if characterized by cooler and moister conditions. Seeding or transplanting success often depends on environmental conditions, and more than one intervention may be required for restoration success in warmer and drier areas. Areas with low resilience and resistance to invasive annual grasses often have low potential for recovery after fire or other disturbance without seeding, especially if the perennial grasses and forbs have been depleted. Seeding or transplanting success depends on site characteristics, the relative abundance of invasive plants, and posttreatment quantity and timing of precipitation. Thus, more than one intervention may be required in these areas.

Soil temperature and moisture strongly influence plant species composition and abundance on a site and are closely related to sagebrush ecosystem resilience to disturbance and resistance to invasive annual grasses (Chambers and others, 2007, 2014a, c). Consequently, soil temperature and moisture regimes that are characterized in soil taxonomic designations (that is, mapping unit) can be used as indications of resilience and resistance at landscape scales in sagebrush ecosystems. Areas with cool to cold and wet to moist soil conditions are typically characterized by high resilience and resistance (see Chambers and others, 2014a; fig. R5). They tend to exhibit less change, recover more rapidly, and are less susceptible to invasion by nonnative invasive annual grasses after stressors and disturbances. In contrast, areas with warm and moist to dry conditions are typically characterized by low resilience and resistance. They tend to exhibit slower ecosystem recovery after stressors and disturbances and may be at greater risk of conversion to alternative states (for example, conversion of sagebrush-perennial grass systems to invasive annual grass systems). In conjunction with information on other landscape threats, such as wildfire risk, this information can be used to help prioritize management actions.

The relative resilience and resistance of a site are closely related to sagebrush ecological types and soil temperature and moisture regimes. Soil moisture availability and plant productivity increase over elevation gradients resulting in greater recovery potential and more competition with cheatgrass. Disturbances that increase soil water and nutrients and reduce competition can decrease both resilience and resistance. Understanding these relationships and mapping them across the landscape (fig. R5) is useful for prioritizing areas for restoration and determining the most effective management strategies (Chambers and others, 2017a).

or diversity within the plant community. Restoring species composition will influence trophic and pollinator relationships to help create functional wildlife habitats and likely result in cascading responses throughout the sagebrush community (Dumroese and others, 2016). Specific sagebrush communities provide food sources for herbivores, especially during fall and winter when herbaceous vegetation is dormant (Campos and others, 2011; Beck and others, 2012; Frye and others, 2013; see chap. D, this volume; chap. E, this volume). Sagebrush species and subspecies of big sagebrush (*A. tridentata*) differ considerably in their height and palatability to sage-grouse, pygmy rabbits (*Brachylagus idahoensis*), and mule deer. Recognizing these differences in habitats is important for successful restoration. Additionally, during the growing season, grasses become an important source of forage for native ungulates, and forbs are a critical feature of diets for ungulates (Scotter, 1980; Mule Deer Working Group, 2009) and greater sage-grouse.

Existing conditions are key factors for consideration when prioritizing at the site scale. Site conditions can be characterized by physical factors such as climate and microclimate, slope, aspect, soil depth, stoniness, restrictive layers, and vegetation factors including the presence of invasive plant species. All of these have the potential to affect treatment success (for example, Germino and others, 2018; Davidson and others, 2019). The potential resilience to disturbance and resistance to invasion of a site (see "Resilience and Resistance" sidebar) are influenced by factors including soil characteristics, elevation and climate, vegetation composition, and disturbance history. Ecological site descriptions (ESDs) and their associated state-and-transition models (STMs; fig. R4) use these factors to provide site-specific information to help determine potentially effective restoration treatments (see "Ecological Site Descriptions, State-and-Transition Models, Species Distribution Models, and other Geospatial Tools" sidebar). Species distribution models (SDMs) and other geospatial tools are also important data sources.

Project-Level Prioritization

Selection of locations to treat within areas prioritized for restoration is influenced by (1) the contribution of an individual site to landscape-scale goals; (2) site conditions, including the site's influence on the landscape and expected responsiveness of the site to restoration intervention; (3) key resources present at the site that impact planning, such as cultural or wildlife concerns or buildings or other infrastructure; and (4) logistics of treating individual sites. If rare or endangered species are present, a protocol of surveillance/detection followed by protection, including avoidance of collateral impacts from restoration treatments such as herbicides and drill seeding, may need to be included.

Designing project-level restoration treatments to meet landscape-level objectives is critical for success and may require considerable analysis and planning. Sites that increase the overall function of the landscape, such as increasing the size of habitat patches or improving connectivity among existing patches, should be a high priority for treatment. For example, site suitability is improved for greater sage-grouse when large, contiguous, or more connected sagebrush patches occur on the landscape (Stiver and others, 2015). Project-scale treatments can also improve landscape resistance to nonnative species by removing invasive plant species at the edges of otherwise intact and noninvaded landscapes.

Local resources can also constrain the tools available for implementation, particularly when restoring one part of a site requires the temporary disturbance or removal of another part required for a sensitive wildlife species. For example, treatments to increase forb or deep-rooted perennial grass cover may reduce annual grasses but still compete with sagebrush recovery and subsequently influence site value to greater sage-grouse (Davies and others, 2011; Germino and others, 2018).

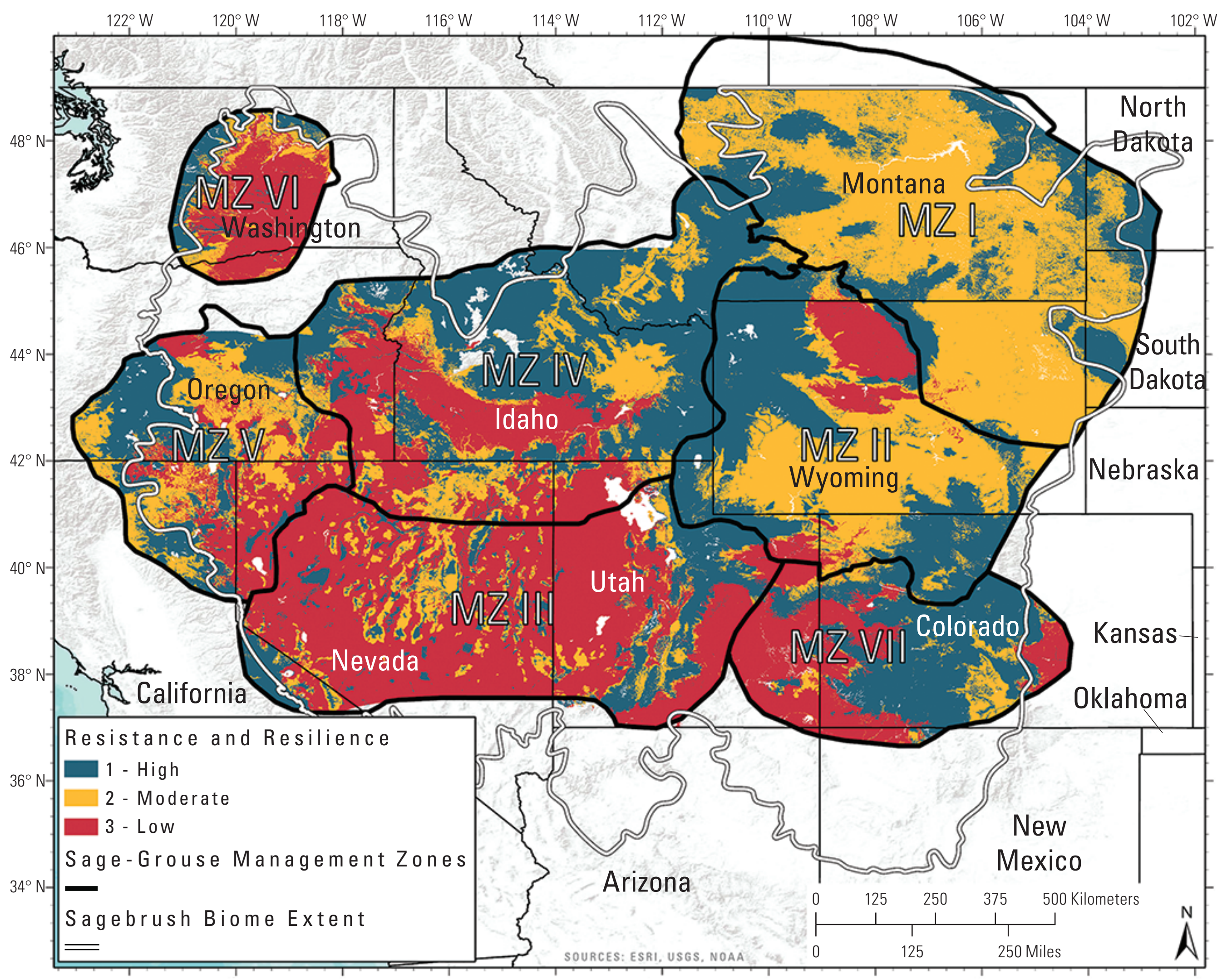

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Figure R5. Resistance and resilience classifications in management zones (MZ) for greater sage-grouse (*Centrocercus urophasianus*; Campbell and others, 2016).

Table R1. Resources to help select and prioritize treatments at the project scale.

Topic	Resource
Site-scale planning	U.S. Geological Survey Circular 1426 (Pyke and others, 2017; https://pubs.er.usgs.gov/publication/cir1426) Benson and others, 2011; (https://wdfw.wa.gov/publications/01330) Bureau of Land Management Technical Note 443 (Dunwiddie and Camp, 2013); https://www.blm.gov/documents/national-office/blm-library/technical-note/enhancement-degraded-shrub-steppe-habitats)
Field guide and score sheets to assist selection of mechanical and prescribed fire treatments and postfire treatments	A Field Guide for Selecting the Most Appropriate Treatment in Sagebrush and Piñon-Juniper Ecosystems in the Great Basin (Miller and others, 2014a; https://www.fs.fed.us/rm/pubs/rmrs_gtr322.pdf) A Field Guide for Rapid Assessment of Post-Wildfire Recovery Potential in Sagebrush and Piñon-Juniper Ecosystems in the Great Basin (Miller and others, 2015; https://www.fs.fed.us/rm/pubs/rmrs_gtr338.pdf)
Importance of various structural and floristic components of habitat for sagebrush-dependent species	Birds in a Sagebrush Sea (Paige and Ritter, 1999; https://www.sagegrouseinitiative.com/wp-content/uploads/2013/09/Birdsinasagebrushsea.pdf) Restoration Handbook for Sagebrush Steppe Ecosystems with Emphasis on Greater Sage-Grouse Habitat (Pyke and others, 2015b; https://pubs.er.usgs.gov/publication/cir1416) Bird Habitat Guide: Sagebrush Communities in the Intermountain West (http://abcbirds.org/wp-content/uploads/2015/05/SagebrushGuide.pdf) Guidelines to Manage Sage-grouse Populations and their Habitats (Connelly and others, 2000a; https://www.fws.gov/nevada/nv_species/documents/sage_grouse/connellyetal2000.pdf) Pocket Guide to Sagebrush Birds (Shultz, 2012; https://digitalcommons.usu.edu/sagestep_reports/20/), and McAdoo and others (2004) and Shaw and others (2005)

Logistical constraints also affect project-level prioritization. Ability to complete all regulatory documentation, access constraints, availability of planting materials, or current partnerships may all influence prioritization and the feasibility of project implementation. Spatial factors such as whether restoration equipment can access and operate given the topography of sites or temporal factors such as suitable weather windows relative to plant community recovery are critically important logistical issues. Incorporating the full set of available information into the prioritization process may improve overall project success (table R1).

Project-Scale Restoration Objectives

For each project, it is essential to establish site-scale management and monitoring objectives (Pyke and others, 2017). Management objectives set targets for attaining an ecological condition while monitoring objectives measure the extant ecological condition. Objectives can be improved by ensuring they are "SMART"—Specific, measurable, achievable/accountable, realistic, and time-bound (Pyke and others, 2015a). Specificity refers to the target species or desired ecological conditions, geographical area, attributes or indicators measured, action or directionality of change in the attribute or indicator, level or values of the indicator that will specify success, and timeframe over which the outcome is expected. Additional detail is provided in the Adaptive Management and Monitoring section of Science framework for conservation and restoration of the sagebrush biome, "Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to Long-Term Strategic Conservation Actions—Part 2—Management Applications" (Wiechman and others, 2019).

Objectives should have some degree of precision to the expected response and be measurable using quantitative metrics. Although it may be easy to identify variables to measure (for example, perennial grass cover in treatment and control sites), it is often harder to identify threshold values in the objectives. For perennial grasses, only a few studies provide guidance on what level of perennial grasses will provide resistance and resilience, and the guidance is generalized and approximate (for example, Chambers and others, 2014c). Furthermore, the objectives must be achievable and reasonable based on available capacity. The time needed to meet the objective must be stated and realistic (for example, 20-percent perennial grass cover in 5 years).

Implementation Requirements

Most restoration implementation will require some degree of regulatory review and conformance. Specifics of regulatory conformance depend on the actions taken, the agency or group performing restoration, and resources present in and adjacent to the treatment area (table R2). Early coordination with interested parties (for example, Tribes, agencies, adjacent landowners, and other stakeholders) is critical given that timelines, documentation requirements, and other needs vary by regulation, regulatory agency, and project complexity.

Weather and Grazing—Two Factors that May Affect Project Implementation and Outcomes

Weather

High spatial and temporal variability in soil temperature and moisture make restoration implementation and success in sagebrush and other semiarid ecosystems very difficult. Transitions from undesirable to desirable plant communities require successful establishment of the perennials being restored, along with effective treatments to reduce exotic annuals. Seed germination, plant establishment, and effectiveness of herbicides are all highly sensitive to weather events (Westoby and others, 1989; Call and Roundy, 1991; Hardegree and others, 2011; James and others, 2011, 2013; Svejcar and others, 2014; Brabec and others, 2017). Long-term patterns of soil microclimate vary with soil type and topography and determine underlying ecological resilience and resistance to annual weed dominance (Knutson and others, 2014). Landscape gradients of resilience and resistance are also correlated with soil microclimate factors that affect seedling establishment in any given year (Hardegree and others, 2013).

Ecological Site Descriptions, State-and-Transition Models, Species Distribution Models, and Other Geospatial Tools

Ecological Site Descriptions (ESDs) are part of a land-classification system that describes the potential of a set of climates, topographic, and soil characteristics and natural disturbances to support a dynamic set of plant communities. ESDs are widely available (for example, https://www.nrcs.usda.gov/wps/portal/nrcs/main/national/technical/ecoscience/desc/.) but incomplete for some areas within the sagebrush ecosystem. State-and-Transition Models (STMs) describe the possible alternative states of sagebrush ecological sites (for example, reference, annual-grass dominated, or seeded perennial-grass dominated; fig. R4; Pyke and others, 2015b; Chambers and others, 2017a, app. 5). State-and-Transition Models (STMs) are a simplified way of characterizing dominant plant composition of sites and how different drivers (for example, fire, grazing, and restoration) may cause shifts in composition and function. They may be useful both at the site/project level and at the landscape-level where STM simulations help project how vegetation changes on an assemblage of sites may impact the broader landscape.

Evaluating existing site conditions relative to a set of reference conditions can help prioritize a list of potential treatments, although choosing reference sites is not trivial and requires many site-matching considerations (Herrick and others, 2019). The level of departure from reference or desired condition can be visualized using STMs, and site conditions can be classified into relevant states based on field-collected data and soil verification of the ecological site (Pellant and others, 2005), compared to rangeland health standards used by BLM or predicted natural conditions such as LANDFIRE biophysical setting models (U.S. Geological Survey, 2014a).

Geospatial and remotely sensed data are typically more available than field data for supporting restoration project planning and may provide information about potential resources at the project-level. Geospatial and remotely sensed data are best considered a supplement to other sources of information on landscape cover, such as field data and local knowledge of landowners or managers. Digital models of vegetation cover, such as the National Land Cover Database Shrubland Products (Xian and others, 2015) or conifer canopy cover mapping (Coates and others, 2017c; Falkowski and others, 2017) can provide estimates of relative cover and height of important vegetation characteristics useful at coarse scales.

Species distribution models provide information about the potential occurrence or abundance of wildlife and sensitive plant species. Species distribution models can help identify the potential occurrence of important wildlife species or sensitive plants that may trigger follow-on assessment and possible adjustments to the placement of restoration treatments. While general tree cover and water bodies are readily mapped with remote sensing (for example, Sankey and Germino, 2008), differentiating and resolving different plant types or species using geospatial data is relatively difficult to achieve in sagebrush ecosystems. The most important project-level consideration for remotely sensed products and derivative models is that the models have unspecified lower-limits of appropriate and reliable spatial application. Specifically, it is difficult or impossible to know if (or where or when) vegetation cover estimates derived from remotely sensed could provide acceptable accuracy and precision at the scale of a 4,047-hectares (10,000 acres) pasture, for example. At this time, accuracy assessments that use field data collected at the scale of the modeled data (that is, field data that have verified accuracy for entire pixels, over many pixels) are rare or nonexistent, and so the onus is currently on each restoration project to find ways of addressing the uncertainty and error in the model. This uncertainty increases the need for integrated monitoring and adaptive management approaches to assess, design, and implement restoration strategies and to monitor the results and adjust management strategies (chap. S, this volume).

Table R2. Example regulatory needs for different conditions or impacts from potential restoration actions.

[U.S.C., United States Code]

Project elements	Regulatory documentation	Regulation
In or near listed species	Biological assessment or State equivalent	Endangered Species Act of 1973 (ESA; 16 U.S.C. 1531 et seq.) or State equivalents
Applied fire	Burn plan, air quality conformance	Clean Air Act (42 U.S.C. 7401)
Effects to the human environment	Environmental assessment	National Environmental Protection Act (NEPA; 42 U.S.C. 4231 et seq.), State Environmental Protection Act (SEPA)
Herbicide use	Pesticide Use Plan (PUP) Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA)	State Department(s) of Agriculture
Cultural disturbance or proximity	Area of Potential Effects (APE), Section 106 report	Antiquities Act (16 U.S.C. 431 et seq.), others

The tools available for understanding vegetation change in sagebrush ecosystems incorporate the impacts of weather variability but provide little guidance for incorporation of weather prediction, variability, and modeling in restoration management planning (Hardegree and others, 2012a, b). These tools include STMs (Briske and others, 2003, 2005; Bestelmeyer and others, 2009), successional planning and management paradigms (Roundy, 2005; Krueger-Mangold and others, 2006; Sheley and others, 2006; James and others, 2010, Davies and others, 2011), and adaptive management strategies (Herrick and others, 2006; Reever-Morghan and others, 2006; Briske and others, 2008; Williams, 2011).

In addition to the soil moisture and temperature regime information described above, data on temporal variability in weather parameters are available from national and local meteorological datasets (table R3). Much of this weather data is limited to areas of high population density or are associated with airports and transportation corridors (Hardegree and others, 2012a). In the Great Basin, the BLM operates the remote automatic weather stations (RAWS) network of rural weather stations, although these sites do not produce data for the whole year and often have a limited period of record. Modeled/gridded weather datasets provide daily weather estimates on a 4 square-kilometer (km^2; 1.5 square-mile [mi^2]) grid across the sagebrush biome and are available from 1979 to present (Daly and others, 2008; Abatzoglou, 2013). In addition to these tabular and spatial datasets, meteorological data tools are being developed that aid in interpretation of weather conditions by characterizing historical site meteorological conditions and microclimatic constraints to seedling establishment (Moffet and others, 2019). These tools are especially useful when interpreting historical field-treatment data and developing long-term adaptive management scenarios that entail multiple treatments over years on challenging restoration sites.

Grazing

Herbivores can impact vegetation and soil conditions during and after restoration activities and may place ecological constraints on the ability to reestablish plants. Selective herbivory results in direct and indirect effects on plant community composition, vegetation structure, soil nutrients and cycling, and other ecosystem processes (Milchunas and Lauenroth, 1993; Frank, 1998; Jones, 2000; Manier and Hobbs, 2007). In sagebrush ecosystems, native plant diversity and landscape heterogeneity have been shown to increase with livestock exclusion (Anderson and Inouye, 2001).

Grazing deferment is perhaps the most broadly applied tool for restoration of sagebrush ecosystems. Permitted grazing is typically deferred for two growing seasons of rest after disturbance or seeding, although there is increasing recognition that greater flexibility in duration of rest is needed (Bureau of Land Management, 2007; Pyke and others, 2017). In general, grazing should not resume until perennial grasses can maintain productivity, recruit new individuals, and stabilize the site (Veblen and others, 2015). However, few datasets are available to test whether 2 years of rest is the appropriate time period. Further guidance on grazing deferment during and after restoration is found in Archer and Pyke (1991), Veblen and others (2015), Pyke and others (2017), and Wiechman and others (2019).

Management of grazing by free-roaming wild horses (*Equus caballus*) and burros (*E. asinus*; WHB) is often more challenging compared to domestic livestock, and effects of WHB should be considered within the context of co-occurring domestic livestock and large native ungulates (Pyke and others, 2017; Griffin and others, 2019). Effects of WHBs on sagebrush plant communities and associated wildlife are discussed in chapter N (this volume), Griffin and others (2019) and Crist and others (2019). In those areas where WHB occur, the ability to manage their populations to specified appropriate management levels (AML) is a primary consideration in deciding if restoration should be implemented (Griffin and others, 2019). Currently, the primary management tool is the gathering of WHB to reduce numbers to the high end of AML.

Foraging by native herbivores may result in fundamental differences in plant responses and ecosystem states compared to livestock and WHB grazing (Manier and Hobbs, 2007; Veblen and others, 2015). The influence of native ungulates should be considered when implementing restoration projects because excessive grazing by native ungulates may be as detrimental as excessive grazing by domestic livestock (Kay, 1995). Reducing numbers of native ungulates or associated grazing or browsing pressure in sensitive areas may be achieved through exclosures, reduction or relocation of supplemental feeding or irrigated forage, or increased harvest or hunting pressure. More information on the effects of native ungulate herbivores on shrubs, such as sagebrush, and their implications for restoration can be found in Kay (1995) and Wambolt and Sherwood (1999).

Tools for Implementation

Appropriate selection of tools and techniques to implement restoration may increase success and prevent unintended consequences. This section describes the primary tools used by managers and discusses the application of each tool and their potential limitations. Passive restoration using grazing deferment as a tool is addressed in the previous section and not repeated here.

Targeted Grazing

Targeted grazing is the application of a specific type of livestock at a determined season, duration, and intensity to accomplish vegetation or landscape goals (Launchbaugh and Walker, 2006). Targeted grazing differs from traditional livestock grazing given its focus on meeting specific vegetation goals instead of other objectives such as livestock production or watershed protection. Targeted grazing has typically been applied to control noxious weeds and, more recently, is being tested for efficacy in reducing invasive annual grass fuels and contributing to recovery of perennials (Launchbaugh and Walker, 2006; Frost and others, 2012; Freese and others, 2013; Schmelzer and others, 2014). Strand and others (2014) identified the four fuel characteristics (live/dead fuel mix, biomass composition, fuel amount, and continuity of fuels) that could be influenced by grazing and the factors that must be considered in modifying fire spread, severity, and intensity (fig. R6). Applying targeted grazing in the dormant season is expected to reduce livestock grazing impacts on perennial plants, reduce cheatgrass by removing litter that promotes germination, and partially remove residual fuels. Successful implementation of targeted-grazing programs aimed at reduction of fuels is challenging because of the need to influence invasive annuals over large and diverse landscapes across multiple years while also responding to variable precipitation and plant production.

In the context of restoration, areas where invasive annual grasses have become the dominant vegetation will only benefit from targeted grazing where enough perennial grasses and forbs exist to promote recovery and where grazing does not have negative impacts on the existing perennial grasses and forbs. The effects of this approach will be context specific, but insufficient research currently exists to predict longer term effects on either perennial native grasses or invasive annual grasses in different sagebrush types.

Targeted grazing has recently been used on fuel breaks. A fuel break is defined as a natural or humanmade change in fuel characteristics that affects fire behavior so that fires burning into them can be more readily controlled (Shinneman and others, 2019). The objective on fuel breaks requires spring livestock grazing to remove current year's growth (Diamond and others, 2009), but there are logistical and ecological challenges to meeting targeted grazing objectives (Shinneman and others, 2018). Demonstration areas, including a robust

Table R3. Sources of weather information.

[RAWS, Remote automatic weather stations; SNOTEL, snow telemetry, PRISM, Parameter-elevation regressions on independent slopes model; -, not applicable]

Data type	Data source	Link	Citation
National and local meteorological datasets	National Centers for Environmental Information	https://www.ncdc.noaa.gov/data-access/land-based-station-data/land-based-datasets	-
	SNOTEL	https://www.wcc.nrcs.usda.gov/snow/	-
	AgriMet	https://www.usbr.gov/pn/agrimet/	-
	AgWeatherNet	http://weather.wsu.edu/	-
	MesoWest	https://mesowest.utah.edu/cgi-bin/droman/whats_new.cgi	-
	RAWS	https://raws.dri.edu/	-
Modeled/gridded weather datasets	PRISM	https://www.prism.oregonstate.edu/	Daly and others (2008)
	GridMet	http://www.climatologylab.org/gridmet.html	Abatzoglou (2013)
Online tools	Great Basin Weather Applications	http://www.greatbasinweatherapplications.org	Moffet and others (2019)

USDA Agricultural Research Service monitoring program, have been established in southwest Idaho and north-central Nevada to attempt to more fully explore the efficacy of targeted grazing to reduce fine fuels on BLM lands (for details, see “Targeted Grazing” at https://www.greatbasinfirescience.org). A caveat is that most large fires in sagebrush ecosystems are burning under extreme fire weather conditions. During these conditions, the fuel type has less influence on fire behavior than during low to moderate fire weather conditions (Strand and others, 2014). For this reason, the focus of the BLM program is on strategically reducing fine fuels, primarily cheatgrass dominated areas, near sagebrush-dominated or previously burned and rehabilitated areas.

Mowing or Thinning Sagebrush Stands

Mowing shrubs and grasses with tractor-pulled mowers is common along roadways and other fuel breaks (Shinneman and others, 2019), and mowing of dense grass swards and increasing soil exposure is sometimes also used to prepare sites for herbicide or seeding applications (Brabec and others, 2015). Mowing and herbicides (especially tebuthiuron) are also used to thin sagebrush stands that are deemed too dense. High densities of sagebrush often result from an inadequate abundance of perennial grasses for postfire resistance and resilience and often do not meet wildlife values (McIver and others, 2014). The intent of thinning sagebrush stands is to promote growth of desirable understory herbaceous vegetation.

Herbicides to Control Invasive Annual Grasses

Precise, targeted spraying of postemergent herbicides or release of biocontrol agents are two tools used in the restoration of sagebrush ecosystems. However, these methods are generally not useful for broadly distributed invasive annual grasses. Instead, use of pre-emergent herbicides, such as imazapic, can reduce germination of annuals with minimal harm to perennials by spraying at the appropriate time or within a year or two after fire (Applestein and others, 2018a). The effects of herbicides typically only last for a few years, but the temporary suppression of annuals provides a window for remnant native bunchgrass populations to recover or establish after reseeding. However, perennial grass recovery often requires more than 2 years and necessitates longer acting herbicides, some of which are under development and not approved for use on lands grazed by livestock (for example, indaziflam [Esplanade]). Determining how to phase the application of both pre-emergent herbicide and seeding is an important area of research because there are currently few established guidelines on how to best apply herbicides (Applestein and others, 2018a). Bioherbicides such as the weed-suppressive soil bacteria *Pseudomonas fluoresens* are also in development, although experimental support for their effectiveness is mixed and often shows no effects (Germino and Lazarus, 2020; Lazarus and others, 2020; see also chap. K, this volume).

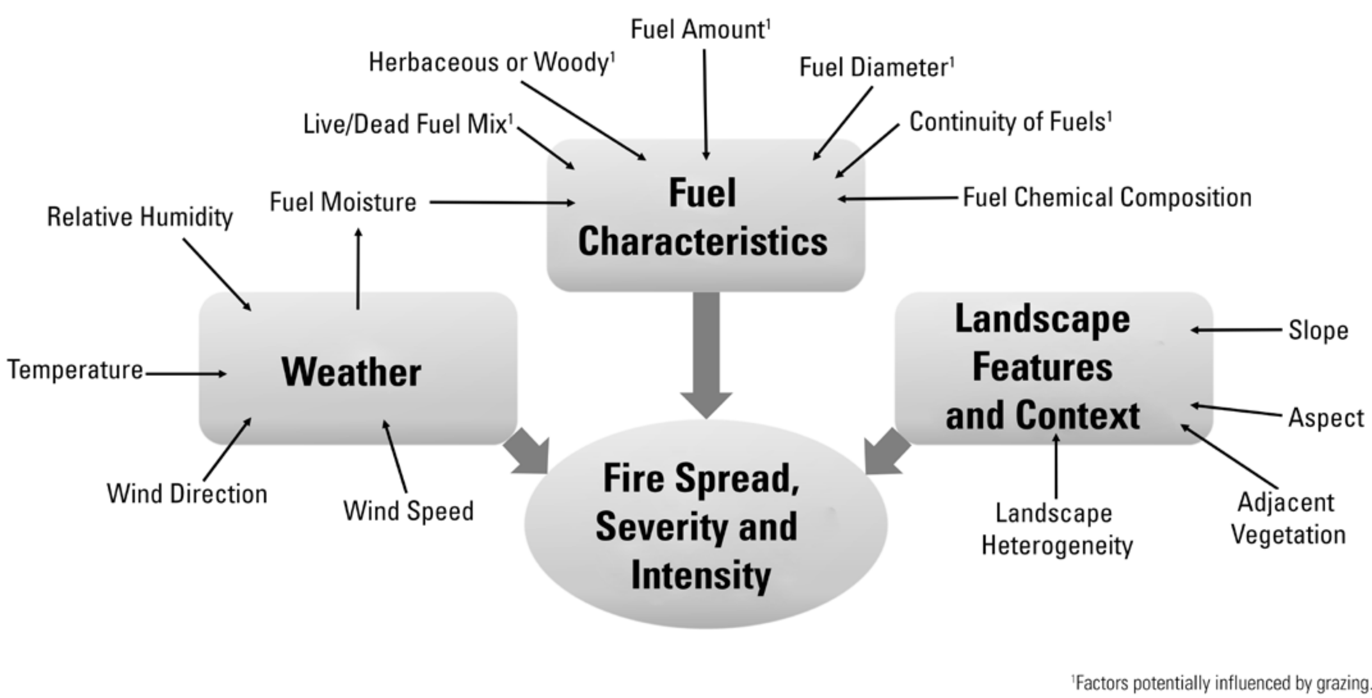

Figure R6. Factors that affect fire spread, severity, and intensity in the sagebrush (*Artemisia* spp.) ecosystem and potential opportunities for grazing to influence fuel characteristics (modified from Strand and others, 2014).

Seeding

Seeding, either using ground-based rangeland drills or aerial broadcast methods, is a common restoration treatment in sagebrush ecosystems (Pilliod and others, 2017b). Seeding decisions require site assessments to first determine if the site is capable of unassisted recovery (Miller and others, 2015). Many factors influence seeding success, including environmental and site conditions favorable for germination and establishment. First, restoration practitioners should ensure there is a proper seedbed with good seed-soil contact, adequate infiltration and nutrient cycling, minimal runoff, and minimal weed seedbanks. Seed beds may be treated mechanically or chemically before or after seeding, for example, using vegetation removal, herbicides, pulling chains or harrows over the ground to redistribute soil over seeds, and soil amendments (Shaw and others, 2005). Wildfire, particularly "clean" burns that leave bare exposed soil, confers benefits for seed germination and initial seedling survival, although with risks of erosion that can be exacerbated by soil disturbances associated with seed bed preparation (Miller and others, 2012; Germino, 2015). Granivores such as small mammals may impact existing seed banks or seeding efforts (Archer and Pyke, 1991), and their impacts to restored areas should be considered.

Selecting the appropriate species and seed source is important for improving success in seeding efforts (Bower and others, 2014), and a number of tools are available to assist with species and seed selection, including ESDs and the Seedlot Selection Tool (for example, https://seedlotselectiontool.org/sst; Doherty and others, 2017). However, in practice, compromises must be made because of seed-supply constraints, the narrow timeframes required to plan and acquire seed postfire, and budget limitations. While these points have not been evaluated in publications, the national ESR program typically receives many more proposals than it can fund, and sagebrush seed availability has not been adequate to meet the demand in some years. Weed contamination of seed mixes is a concern, although it is an even greater concern for straw or mulch applied to stabilize soils (Beyers, 2004).

Today, efforts are being made to seed diverse plant communities to establish species important to ecosystem structure, function, and wildlife. The National Seed Strategy (Plant Conservation Alliance, 2015) focuses on the importance of diverse seed mixes. For example, research is focusing on establishing native forbs in seed mixes that provide habitat for pollinators and forage for sage-grouse chicks. Only a small set of species are typically used for seeding because they are common and easily collected, thus readily available as a seed source (Shaw, 2004). The number of species is also limited from lack of knowledge about the species-specific requirements for germination and establishment, seeds of less dominant species are often scarce, and the low availability is compounded by high costs. In addition, seeding diverse mixes is difficult owing to species requirements in seeding depth, presence of seed appendages, and other factors that complicate application (Shaw, 2004).

Big sagebrush is the most commonly seeded shrub species, is most often seeded in winter, and must be collected just prior to aerial broadcast because of its short longevity in storage (reviewed in Meyer and Warren, 2015). Additionally, sagebrush exhibits high levels of local adaptation to climate conditions such as minimum temperatures (Brabec and others, 2017; Germino and others, 2019). Fortunately, seed-transfer guidelines (Chaney and others, 2017; Richardson and Chaney, 2018) provide a relatively new tool to help match the climate of origin to the seeding sites and may improve seeding success. Some of the population-level diversity of sagebrush is attributed to subspecies, and seeding the correct subspecies to an ecological site is important (Mahalovich and McArthur, 2004). Sagebrush seeding success is also influenced by snow-water abundance (Shriver and others, 2018) and by critical topographic, soil, and plant community properties that create patchiness, or unevenness, in recovery of competing vegetation that may assist in sagebrush establishment (Germino and others, 2018).

Native bunchgrass seed is often readily available and commonly planted in restoration projects in sagebrush ecosystems. Cultivars of dominant native species, such as bluebunch wheatgrass (*Pseudoroegneria spicata*, "Anatone") and Sandberg's bluegrass (*Poa secunda*, "Sherman") and nonnative bunchgrass species, including crested wheatgrass, Siberian wheatgrass, or Russian wildrye (*Psathyrostachys junceus*), are available (Asay and others, 2003). These cultivars, especially nonnatives, have relatively high establishment rates but may tend to dominate at the expense of plant species diversity once established (Fansler and Mangold, 2011). Cultivars may provide site resistance to invasion and resilience to disturbance; however, they may not always provide satisfactory palatability or habitat for wildlife (Ganskopp and others, 1997; Beck and Mitchell, 2000).

Owing to the many challenges involved with seeding efforts, seeding success has been mixed with many cases of failure, especially at lower elevations sites with low resistance and resilience (Knutson and others, 2014). However, new insights on the spatial and temporal factors affecting seeding success are improving our ability to plan successful treatments based on past treatment outcomes. Examples of factors that have been associated with positive treatment outcomes include the influence of soil-surface conditions known as "pedoderms," which include soil crusts, and the discovery that grasses do not invariably outcompete sagebrush (Germino and others, 2018). Moreover, new seeding technologies are currently being developed that may increase seeding success, including seed coatings that manipulate water availability and hormones that alter germination timing (Madsen and others, 2012, 2018).

Transplants

Transplants, or outplants of nursery seedlings, may be more effective than seedings in harsh environmental conditions that limit seed germination or seedling survival and subsequent seeding success (Knutson and others, 2014). Transplanting is considerably more expensive and requires greater time and effort than seeding, and thus is only applicable to smaller areas that will provide clear benefits relative to the effort and cost. Transplanting projects often entail a few thousand plants, although larger projects exist (for example, greater than 1 million sagebrush outplants on the 2015 Soda Wildfire). Some areas with high wildlife values receive repeated shrub and forb transplants nearly every year (for example, Birds of Prey National Conservation Area in Idaho).

While the science and practice of using transplants in sagebrush ecosystems is still in its infancy (McAdoo and others, 2013), transplants may be advantageous for species that are difficult to establish via seed in field conditions, attaining rapid soil stability, accelerating the development of wildlife habitats and forage production, and providing windbreaks (Shaw, 2004). Shrubs and forbs are most commonly and successfully transplanted, and while grasses also are successfully transplanted, they are more readily restored through seeding using a rangeland drill compared to forbs and shrubs.

Timing of planting can strongly affect transplant success. Some species and settings may be best planted in spring to avoid freezing-induced mortality. For sagebrush, outplanting in late fall, just prior to the onset of winter freezing and moisture accumulation, is often the most operationally feasible and ensures that seedlings can capitalize on spring moisture and warmth for root growth prior to seasonal drought (Stevens, 2004; Pyke and others, 2017). Modifications to increase soil moisture and nutrient availability, including hydrogels and woody material, may increase short-term survival (Minnick and Alward, 2012; Dettweiler-Robinson and others, 2013). Selection of the appropriate species and genotypes for the ecological site is also considered important (Edwards and others, 2019). However, in complex terrain, environmental factors such as slope, aspect, soil, and the abundance of annual or perennial grasses can be stronger predictors of transplant success than taxonomic/subspecies identity (Davidson and others, 2019).

Common transplant materials in uplands include bare-root or container stock reared in nurseries and plants collected from the field. Cuttings for vegetative propagation may be used with meadow or riparian plants, and some species (for example, willows [*Salix* spp.]) are easily field propagated. Container stock are sometimes more versatile because they have an existing growing medium, are less prone to drying out, and can be stored for longer before transplanting. However, bare-root stock has advantages including rapid establishment, the availability of older plants with stronger roots and shoots hardened to outdoor conditions, and lower cost compared to container stock (Stevens, 2004). If a local source is available, excavation and transfer of plants collected from wild plant populations to restoration sites has the advantage of already-established microbial communities (Pyke and others, 2017), although field studies have not found mycorrhizae presence to increase transplant survival (Minnick and Alward, 2012; Dettweiler-Robinson and others, 2013).

Transplants may be done by hand or with mechanical planters. Sites may need to be treated with herbicide prior to planting transplants to reduce competition from annual species (Van Epps and McKell, 1983; McAdoo and others, 2013). After transplanting, seedlings may need to be protected from herbivory, although the type of herbivore and timing of herbivory vary in their effects (Austin and others, 1994; McAdoo and others, 2013).

Conifer Removal to Reduce Tree Expansion

Expansion of juniper (*Juniperus* spp.) and pinyon pine (*Pinus* spp.), Douglas-fir (*Pseudotsuga menziesii*) and ponderosa pine (*P. ponderosa*) into sagebrush ecosystems may inhibit wildlife, particularly sage-grouse because trees provide perches for predators and displace sagebrush and associated understory species (Coates and others, 2017a; Miller, R.H., and others, 2017; Severson and others, 2017a). Moreover, the loss of perennial grasses with conifer expansion decreases resilience to fire and resistance to postfire invasion (Miller, R.H., and others, 2017). Thus, conifer removal is an extensive restoration activity in sagebrush ecosystems. The impacts of conifer expansion into sagebrush habitats on ecological processes and wildlife and the benefits of conifer removal are described in chapter M (this volume).

Conifer removal may be implemented by prescribed fire or mechanical methods (Miller and others, 2014a). Mechanical methods include removing entire trees using large equipment (for example, roller choppers or masticaters) or cutting individual trees with chainsaws. Cut or pulled trees can be left in place, piled and burned, removed from sites, or mulched; however, each method can have strong effects on herbaceous regrowth (Roundy and others, 2014a; Williams and others, 2017). Where the goal of tree removal is improving sage-grouse habitat, the selected approach should leave no standing tree skeletons. In sites that are susceptible to invasion, ground disturbance should be minimized. Managers should be aware that posttree removal herbicide applications and seeding may be required once niches are opened from mechanical, chemical, or fire treatments (Miller and others, 2014b). There are field guides available for selecting the most appropriate treatments (Miller and others, 2014a) and assessing site recovery potential (Miller and others, 2015).

Frameworks and Tools

The list of frameworks, data, and tools is rapidly expanding to assist managers faced with prioritizing, planning, and implementing projects across large landscapes. Access to data and tools useful for prioritizing areas for restoration at broad, ecoregional scales has also increased dramatically with the development of online web portals. The "Science Framework for Conservation and Restoration of the Sagebrush Biome—Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to Long-Term Strategic Conservation Actions—Part I—Science Basis and Applications" (Chambers and others, 2017a) provides a common starting point and contains information and a number of geospatial resources to inform landscape prioritization. Federal agency implementation of these principles has begun through the BLM's integrated program of work and Forest Service's fire and invasive assessments and through collaborations between the BLM and the NRCS Sage Grouse Initiative; State programs such as the Utah's WRI are using these tools more locally.

Several portals and web-mapping interfaces provide a suite of geospatial information and decision support tools to inform landscape-level decision making. The BLM's Landscape Approach Data Portal is one example providing a curated subset of tools and interagency geospatial data (https://landscape.blm.gov), including the data contained in the "Science Framework for Conservation and Restoration of the Sagebrush Biome—Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to Long-Term Strategic Conservation Actions—Part I—Science Basis and Applications" (Chambers and others, 2017a). Empirical, field-collected data on plant cover and other land health parameters are available through the BLM assessment, inventory, and monitoring (AIM) program (https://aim.landscapetoolbox.org/) and can be analyzed at landscape or smaller spatial scales. The BLM and Forest Service have also classified habitat condition for greater sage-grouse at landscape and smaller spatial scales through the sage-grouse habitat assessment framework (HAF) for many landscapes (Stiver and others, 2015). The U.S. Geological Survey (USGS), BLM, Forest Service, and Western Association of Fish and Wildlife Agencies (WAFWA) have developed SageDAT (https://sagedat.org), which allows for sharing and leveraging of data, maps, and map services and facilitates broad participation and transparency in decision making.

The use of emerging technologies will allow Federal, State, and local agencies, as well as NGOs, industry, and private parties to share relevant data and tools while maintaining and preserving control by data owners. Data and tools characterizing conditions at landscape scales are often used to set broad objectives that will improve the overall quantity, composition, or configuration of sagebrush habitats. However, this information often lacks the accuracy or precision to be used for site or project-scale implementation. Often, finer scale or field-collected information is necessary when planning and implementing projects.

Evaluation of Outcomes

Stakeholders, partners, funding programs, and supervisors usually require some level of reporting on outcomes of some restoration projects. On short timeframes, managers require information about vegetation recovery and response to restoration treatments to make appropriate posttreatment decisions, such as allowing resumption of grazing and the need for re-treatment. At broader time and spatial scales, outcomes are evaluated to address accountability and general interest and enable learning and improvement for future investments. Historically, restoration efforts in sagebrush ecosystems were difficult to evaluate and learn from at broader spatial and temporal scales (that is, multiple fires) owing to inadequate documentation of treatment timing, location, and details, in addition to limited data on vegetation and ecosystem responses. Availability of the USGS Land Treatment Digital Library has improved treatment documentation and the ability to evaluate and learn from treatment outcomes (Pilliod and others, 2017b). Most evaluations of restoration focus on one taxonomic or community type. Few evaluations quantitatively consider the whole ecosystem, from plants and biological soil crusts to the higher trophic levels such as wildlife and forage production, and yet integration across trophic levels may offer robust insight on ecosystem functioning.

Key evaluation parts include clear objectives, devising and implementing appropriate monitoring and measurements, data management and quantitative synthesis, and assessment of the quantitative outcomes, including determining whether objectives were met or not. Most restoration projects are best evaluated with many observation points measured in a short period of time, rather than few observation points within heterogenous landscapes in order to provide adequate inference over the treated area (Applestein and others, 2018b). In many cases, a small number of plots are used to represent the response of large burn areas up to 40,470 hectares (100,000 acres) or more. The likelihood of a few observations representing the average conditions in the landscape is low, and the patches where high or low recovery success are incipient can help inform follow up treatment actions while community recovery is still underway. This need for spatially adequate sampling contrasts the most common measurement approaches and methods described in chapter S (this volume), which tend to entail either a low-density of observation points (few plots per unit area) monitored persistently over time or coarse information obtained wall-to-wall from remote sensing.

Recovery of sagebrush ecosystems from disturbance can require decades (Wambolt and others, 2001; Lesica and others, 2007), such as the approximately 80 years observed for oil-well drill pads in Wyoming (Avirmed and others, 2015), although herbaceous communities may exhibit stable responses over shorter timeframes. Most restoration evaluations occur at much shorter timeframes, so it may be appropriate to evaluate trends in vegetation recovery. Guiding questions may focus on whether invasive annual grasses are decreasing relative to increasing abundances of native perennial grasses, shrubs, and forbs. Most importantly, yet rarely considered, evaluation should consider how recovery compares with the spatial and temporal factors that affect the likelihood of success. Although sagebrush ecosystems are sometimes perceived as being homogenous landscapes, important topographic, edaphic, climatic, and biological variability exists within and among sites and will cause differences in restoration outcomes, such as differences in resistance and resilience and hotspots for sagebrush recovery (Germino and others, 2018) or broader-scale variability in resilience and resistance (Chambers and others, 2014a).

Evaluation of restoration treatments, such as in final reports or assessments of whether success has been achieved, should weigh the extent of restoration success or failure against physical and biological conditions of sites, weather before and after treatments, treatment details such as seed sources, and whether multiple interventions were used and should all be considered. For example, small amounts of perennial establishment might be considered a relative success in low-resistance and low-resilience zones that were invaded, but not in high-resistance and high-resilience sites.

Social and Economic Costs and Opportunities

Successful implementation of restoration treatments depends not only on ecological considerations but also on factors in the social environment (for example, social acceptability of treatments, institutional context, and economic considerations). Federal law requires considering the concerns of citizens about the potential impacts of treatments, and those concerns may vary with the treatment being proposed. For example, public acceptance of treatments often depends on the extent to which the methods for changing vegetation structure seem to mimic natural processes or the potential severity of initial impacts immediately after treatment. Thus, targeted grazing is generally more acceptable than prescribed burning, fire or biological control is preferred over mechanical treatments, and mechanical treatment is preferred over chemical approaches (Tidwell, 2005; Gordon and others, 2014). People are also more likely to believe a treatment is acceptable than they are to have confidence in an agencies' ability to implement them safely and effectively (Shindler and others, 2011). For that reason, it can be valuable for agencies to engage in active consultation with the local citizens to enhance trust before implementing large-scale or highly visible restoration efforts (Shindler and others, 2014).

Other barriers to implementation include financial (for example, costs of implementation and access to those resources), social/political (for example, resistance from a local community, advocacy group, or politician), or institutional (for example, adequacy of staffing, local office customs, legacy of prior bad outcomes, or legal barriers). Managers may be reluctant to implement new or unfamiliar treatments—even if indicated by the best science—if they perceive that innovation will not be rewarded, that public opposition will be significant, or that they lack time and resources to learn how to implement the treatment successfully (Wright, 2010; Hardegree and others, 2018). This can lead to situations where managers rely upon tried-and-true approaches even when the effectiveness of such treatments is in question.

Economics—Costs/Benefits of Treatment

Economic analysis can be useful for identifying and targeting opportunities for restoration. For example, comparison of costs and benefits across locations or ecological conditions can identify where best to invest in restoration (Boyd and others, 2015; Eiswerth and others, 2016). Ecologically intact sagebrush ecosystems provide a range of important benefits, including biodiversity protection, ecosystem service provisioning, and reductions in long-term fire suppression costs (Havstad and others, 2007; Epanchin-Niell and others, 2009). While individual land users may primarily consider the benefits of restoration that directly affect them (for example, forage values), a social cost-benefit or return on investment analysis, which considers both the direct and indirect costs and benefits that accrue to society, including environmental benefits, is appropriate when evaluating public resource investment in restoration (Macleod and Johnston, 1990; Boyd and others, 2015).

Estimation of the socioeconomic benefits of restoration is difficult, especially because benefits often flow from protection or enhancement of ecosystem services that are not traded in markets and are difficult to monetize (Aronson and others, 2010). Similarly, while costs of invasive species control are relatively easy to estimate, damages of not engaging in such control are more difficult to assess (Epanchin-Niell and Hastings, 2010). Without explicit links between restoration and the products of ecological processes, it is difficult to capture and convey the values that may be achieved through restoration (Brown and MacLeod, 2018). If these values are excluded from economic analyses owing to these difficulties, benefits of restoration will be underestimated, leading to underinvestment in restoration and misinformed resource allocation. While the quantification of benefits can be challenging, even qualitative, systematic documentation of the anticipated effects of restoration across a range of values can be useful for informing resource allocation decisions (Epanchin-Niell and others, 2018). Also, return on investment analysis, in which benefits are quantified but not monetized, enables cost-effective targeting of restoration investments when monetization of benefits is not feasible (Boyd and others, 2015).

Existing economic analyses tend to support the adage that "an ounce of prevention is worth a pound of cure." Postfire reseeding prior to a cheatgrass invasion is cost effective in the long run simply by reducing fire suppression costs (Epanchin-Niell and others, 2009). Restoration treatments to prevent

Wyoming big sagebrush (*A. t. wyomingensis*) and mountain big sagebrush (*A. t. vaseyana*) communities from becoming dominated by invasive annual grasses are similarly cost effective when accounting for reduced fire-suppression costs. However, the success rates for restoration and rehabilitation efforts have substantial influence on the expected benefits of treatment. For example, it is estimated that a 52-percent success rate or lower costs of restoration would be needed for a positive benefit to cost ratio for restoration at sites already dominated by invasive annual grasses (Taylor, M.H., and others, 2013). Coordination of exotic grass invasion efforts across ownerships and agencies is one way to improve cost-effectiveness by reducing costs from reinvasion (Epanchin-Niell and Wilen, 2015).

Ecosystem restoration projects can provide meaningful economic contributions to local and regional economies, although the magnitude of impacts varies depending on characteristics of the local economy where restoration takes place and factors in the restoration itself (for example, the degree to which sources of materials and labor are local). Based on analysis of a series of case studies, it is estimated that between 13 and 32 job-years and between $2.2 million and $3.4 million in total economic output are contributed to the U.S. economy for every $1 million invested in ecosystem restoration (Cullinane and others, 2016).

While institutions such as natural resource conservation districts have tended to focus on single-issue restoration efforts (for example, improving riparian function or restoring livestock forage) rather than broader ecosystem-wide goals, it may be possible to achieve broader goals and more effectively define social as well as economic benefits, if projects explicitly define spatial and temporal extent and engage landowners, policy makers, and concerned citizens in restoration planning (Brown and MacLeod, 2018).

Appendix R1. Generalized and Sagebrush-Ecosystem Specific Information Sources

- "Science Framework for Conservation and Restoration of the Sagebrush Biome—Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to Long-Term Strategic Conservation Actions—Part I—Science Basis and Applications" (Chambers and others, 2017a) and "Science Framework for Conservation and Restoration of the Sagebrush Biome—Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to Long-Term Strategic Conservation Actions—Part II—Management Applications (Crist and others, 2019). These documents provide (1) a science basis and approaches for prioritizing areas for management activities and determining the most appropriate treatments across scales and (2) information to help apply the science and approaches, including using the National Seed Strategy (Plant Conservation Alliance, 2015) in restoration efforts (Edwards and others, 2019).
- Field guides provide an approach for assessing the relative resilience and resistance of project areas to select appropriate treatment areas and treatments in juniper (*Juniperus* spp.) and pinyon (*Pinus* spp.) pine woodlands (Miller and others, 2014a) and to make appropriate restoration decisions postwildfire (Miller and others, 2015).
- A three-volume set of manuals that provides concepts, tools, and approaches for restoration from the landscape to site scales with an emphasis on conservation of greater sage-grouse (*Centrocercus urophasianus*; produced by Pyke and others, 2015a, b, 2017).
- Detailed information on plant species selection and project-level treatments (Monsen and others, 2004).
- Other U.S. Geological Survey circulars and U.S. Department of Agriculture (USDA), Forest Service General Technical Reports on specific topics in sagebrush ecosystem restoration are available on websites such as the Great Basin Fire Science Exchange (greatbasinfirescience.org); the U.S. Department of Agriculture (USDA), Natural Resources Conservation Service (NRCS; https://www.nrcs.usda.gov); the USDA Plants database (https://plants.usda.gov/); and the Society for Ecological Restoration Great-Basin chapter website (https://chapter.ser.org/greatbasin/).
- Webinar series or symposia:
  - Society for Ecological Restoration's website (https://www.ser.org/page/NewsandEvents).
  - Great Basin Fire Science Exchange site (https://greatbasinfirescience.org).

Chapter S. Adaptive Management and Monitoring

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Executive Summary

Adaptive management and monitoring efforts focused on vegetation, habitat, and wildlife in the sagebrush (*Artemisia* spp.) biome help inform management of species and habitats, predict ecological responses to conservation practices, and adapt management to improve conservation outcomes. This chapter emphasizes the adaptive resource management framework with its four stages: (1) problem definition, (2) outcomes, (3) decision analysis, and (4) implementation and monitoring. Adaptive resource management is an evolving process involving a sequential cycle of learning (the accumulation of understanding over time) and adaptation (the adjustment of management over time). This framework operationalizes monitoring a necessary component of decision making in the sagebrush biome. Several national and regional monitoring efforts are underway across the sagebrush biome for both vegetation and wildlife. Sustaining these efforts and using the information effectively is an important step towards realizing the full potential of the adaptive management framework in sagebrush ecosystems. Furthermore, coordinating monitoring efforts and information across stakeholders (for example, Federal, State, nongovernmental organizations) will be necessary given the limited resources, diverse ownership/management, and sagebrush biome size.

Introduction

In natural resource management, monitoring provides information about how resources change through time in response to management or whether resource objectives are met following a management action. Well-designed monitoring for specific conservation problems begins with clearly articulated objectives, often with input from multiple stakeholders. There are many conservation challenges facing the sagebrush (*Artemisia* spp.) biome, and thus, there are a myriad of monitoring approaches or programs. This chapter describes monitoring efforts focused on vegetation, habitat, and wildlife. Collectively, the existing natural resource monitoring in the sagebrush biome (and potentially other future monitoring efforts) can help inform management of species and habitats, predict ecological responses to conservation practices, and adapt management to improve conservation outcomes (Nichols and Williams, 2006; Lyons and others, 2008). Monitoring may also help maximize efficiency of conservation spending so that limited resources are spent on the right things, in the right places, and at the right time.

Types of monitoring used in natural resource management include implementation, effectiveness, and validation (Wiechman and others, 2019), all of which can inform adaptive management if implemented within the appropriate framework (fig. S1). Implementation monitoring evaluates the successful execution of a planned management action, such as whether seeded species germinate and emerge in the first growing season. Effectiveness monitoring evaluates changes in condition and progress toward meeting a management objective, such as stabilizing soils following wildfire rehabilitation or increasing bird populations after restoring wildlife habitats. Validation monitoring uses an experimental approach to determine if the observed outcome is caused by a management action. Some view this latter approach as hypothesis-driven research and thus outside the realm of monitoring for adaptive management. This includes most short-term, local research projects conducted by agencies and universities, including those that evaluate alternative management options.

Given the uncertainty in the management of natural resources, monitoring needs to be integrated into all management systems to maximize effective decision making and sustain conservation efforts. Examples of approaches for integrating monitoring data into decision making frameworks include: (1) Systematic conservation planning to answer the "what to do" and "where to do it" questions; (2) Structured decision making (SDM) to integrate stakeholder objectives, alternative management actions, data models and tradeoffs; (3) Adaptive resource management (ARM) that extends SDM processes to include effectiveness monitoring over time; and (4) Strategic habitat conservation that integrates the principles of conservation planning and ARM at the landscape level (Wilson and others, 2009; Marcot and others, 2012; Millard and others, 2012; Williams and Brown, 2012; Drum and others, 2015).

[1]U.S. Geological Survey.

[2]Bird Conservancy of the Rockies.

[3]U.S. Department of Agriculture, Forest Service.

[4]Oregon State University.

[5]Operational Conservation.

[6]U.S. Department of the Interior, Bureau of Land Management.

[7]Colorado State University, in cooperation with the U.S. Geological Survey.

[8]U.S. Fish and Wildlife Service.

Although monitoring is often given considerable attention in conservation and management policies and plans, it is often treated as an afterthought in conservation and management action. Monitoring data are inadequately used in adaptive management because of a lack of consistent understanding among those tasked with addressing all or some of the steps required for effective adaptive management. Adaptive management operationalizes monitoring as a necessary part of decision making, and as such, this chapter outlines the use of vegetation and wildlife monitoring in sagebrush ecosystems within the construct of adaptive management.

Adaptive Management

Adaptive management is a structured approach to decision making. Adaptive management essentially means learning by doing and adapting management strategies based on what has been learned (Williams and others, 2009). In all cases, adaptive management is seen as an evolving process involving a sequential cycle of learning (the accumulation of understanding over time) and adaptation (the adjustment of management over time). This feedback between learning and decision making is the central feature of adaptive management (Williams and others, 2009; Williams and Brown, 2012). It is important to recognize that adaptive management is the actual process of implementing a conservation program, not a part of the program to be initiated upon failure to attain an objective. Although adaptive management is not conceptually complex or operationally intricate, successful implementation of the process requires long-term perspective, commitment, and dedication, and it can be expensive (Williams and others, 2009; Williams and Brown, 2012). However, given the uncertainty surrounding the proactive management of sagebrush habitats coupled with the need to pursue innovative management approaches to achieve landscape-scale conservation goals in these habitats, the process of how conservation programs are implemented may be as important as the actual management and conservation actions pursued. Strictly adhering to adaptive management principles can inherently facilitate the application of this conservation strategy and the ecological principles described herein, thereby increasing the likelihood of attaining conservation success.

Structure of the Adaptive Management Process

The ARM framework proceeds in four stages involving (1) problem definition, (2) outcomes, (3) decision analysis, and (4) implementation and monitoring (Hammond and others, 2002; Marcot and others, 2012). Although monitoring is an essential component of ARM, it must be integrated within the management context to measure progress toward achieving management objectives (Nichols and Williams, 2006; Lyons and others, 2008).

The first stage of ARM is a clear articulation of the conservation problem to be solved and involves framing the problem, defining objectives, and establishing criteria by which alternative solutions can be evaluated (Marcot and others, 2012; Nichols and others, 2012, fig. 1). The articulation of the problem statement is an indispensable aspect of the ARM framework. Problem structuring involves identifying the responsibilities of decision makers, recognizing necessary tools and information, determining appropriate levels of investment, and ensuring the right problem is being solved (Marcot and others, 2012). Problem framing and objective setting stems from the policy, legal, and social dimensions of the management context and reflects the values of decision makers and stakeholders. Because natural resource management often involves multiple and potentially competing objectives, the development of objectives often benefits from workshops involving social scientists and experts in human dimensions to elicit the values of decision makers and stakeholders (Marcot and others, 2012). Objectives play the central role in ARM because they drive the other aspects of the process.

Second, the outcome analysis stage of ARM entails defining the full range of alternative management options, estimating their potential consequences, analyzing tradeoffs, and identifying key uncertainties (Marcot and others, 2012). Defining alternative management options may involve input from stakeholders, but the remainder of the decision analysis involves confronting management alternatives with mutually agreed-upon objectives developed in the problem-definition stage. Evaluating consequences involves predicting the outcomes of each alternative management action in terms of measurable objectives (Marcot and others, 2012). Quantitative modeling of existing data is often used to predict outcomes for each alternative management option. However, existing data may be of little use if not relevant to the objectives. Hence, not all existing monitoring data can be retrofitted or repurposed for new or future objectives.

Methods of addressing uncertainty in an ARM context often involve assessing the value of information relative to the predicted outcomes, thereby establishing the extent that information discriminates between management decisions (Canessa and others, 2015; Maxwell and others, 2015). In cases where the expected value of information is high or important, such as monitoring trends in populations of a species of concern to inform Endangered Species Act of 1973 (ESA; 16 U.S.C. 1531 et seq.) listing decisions, then it may be appropriate to implement research to reduce uncertainty prior to making management decisions. However, this is not always realistic within management timeframes or budgets. There is no advantage in gathering additional information if the expected value of information or the power to reduce uncertainty is likely to be low (Marcot and others, 2012). The concept of uncertainty in decision making differs from uncertainty in a scientific context. In many cases, reducing scientific uncertainty about predicted outcomes may not reduce uncertainty relative to the best course of action. Nevertheless, uncertainty can influence model predictions for the effects of management alternatives (Marcot and others, 2012), and several approaches have been developed to deal with uncertainty concerning resource conditions, consequences of management options, uncontrolled environmental variation, and dynamic processes (Williams and Johnson, 2013).

Third, decision analysis involves the selection of an alternative policy, conservation plan, or management option (Marcot and others, 2012). A decision can be thought of as an irrevocable allocation of resources, and may be a choice between strategic directions, such as land and resource management within a given region or area, or project-level decisions involving specific management actions. Several decision analysis frameworks are available for the transparent ranking of management alternatives using available science, values, and preferences of decision makers, and considerations raised by stakeholders (Marcot and others, 2012).

Fourth, implementation and monitoring describe a process of land and natural resource management where monitoring is integrated with the implementation of the preferred management alternatives (Marcot and others, 2012). Within the ARM process (fig. S1), the learning or adaptive phase is represented by the monitor and model components, whereas the optimization or management phase is represented by the model and decide components (fig. S1; Nichols and others, 2012). The state variables to measure and the scale of monitoring should be directly linked to the management context with a clear understanding of how the information gathered will be used to evaluate the management objectives (Marcot and others, 2012). To ensure the feedback necessary for ARM, the iterative, cyclic nature represented by the arrows in figure S1 is critical for sustainable conservation.

Adaptive resource management is a promising framework for managing sagebrush ecosystems (Kachergis and others, 2013; Hardegree and others, 2018), but the full potential of the adaptive framework has yet to be realized. In many respects, the term “adaptive management” has become a catchall phrase meaning something different to conservation planners, land managers, and research scientists (Williams and Brown, 2012). Despite considerable progress in conservation planning, management, and science in sagebrush ecosystems (Davies and others, 2011; Miller and others, 2011; Christiansen and Belton, 2017), separate frameworks for land management and conservation science developed in isolation may ultimately impede learning (Williams and Brown, 2012). In addition, monitoring to inform management in an informal or indirect way is often assumed sufficient to close the feedback loop in adaptive management (Williams and Brown, 2012). Attempts to develop adaptive frameworks in an ad hoc way often overlook key steps in the process and have been termed “adaptive management lite” (Ruhl and Fischman, 2010). These ad hoc approaches often suffer from the lack of clearly defined objectives, monitoring thresholds, and actions triggered by thresholds, and are better characterized

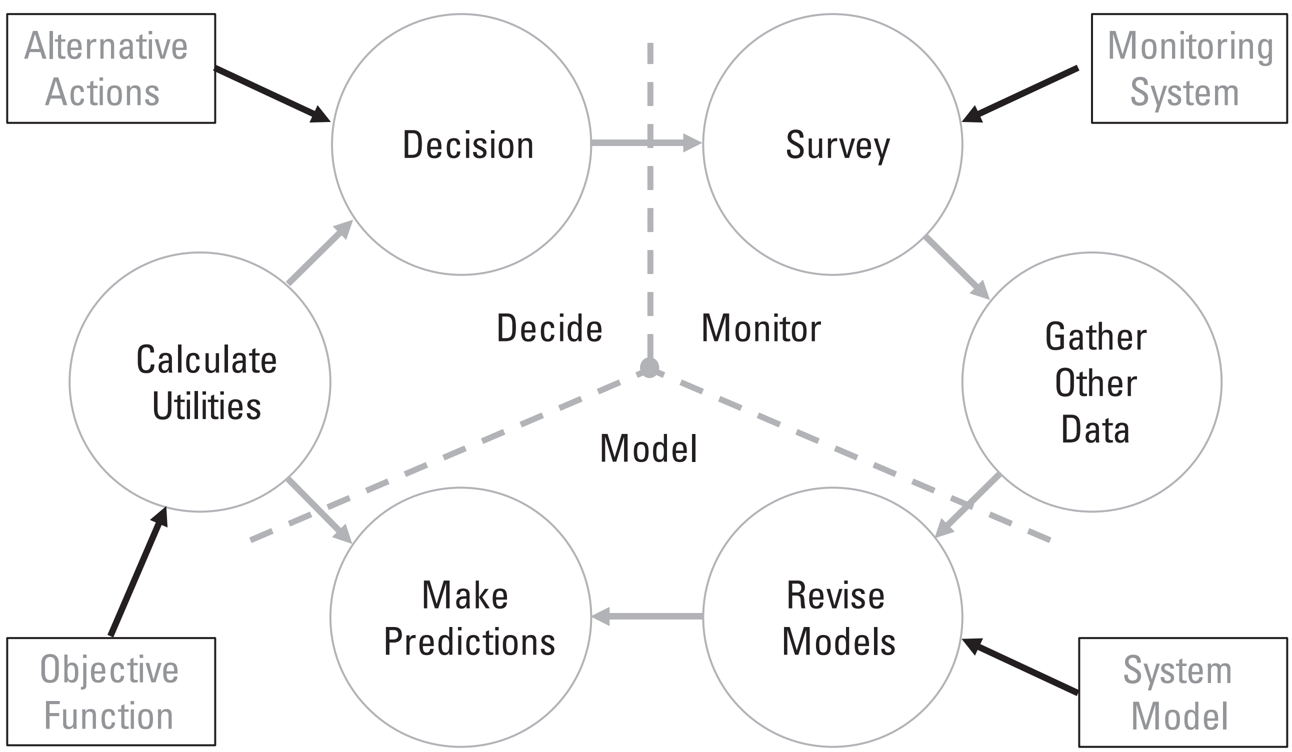

Figure S1. Monitoring in an adaptive resource management framework. Modified from Nichols and others, 2012.

as contingency planning based on monitored conditions than adaptive management (Fischman and Ruhl, 2016). Because legal proceedings have overturned several applications of adaptive management lite, adopting the adaptive management process as defined in the literature (fig. S1) may improve transparency, stakeholder participation, and accountability in the management of resources in the public trust (Fischman and Ruhl, 2016).

Vegetation Monitoring

Vegetation monitoring in sagebrush ecosystems ranges from local efforts on grazing lands to regional efforts designed to understand trends in rangeland health and wildlife habitats. Tracking changes in vegetation parameters of interest can be difficult and sometimes requires specific methods and sampling designs that allow for statistical analyses. The ability to detect changes in habitats over time depends on methods that provide precise estimates at each time interval so that meaningful differences can be detected (Seavy and Reynolds, 2007). In addition, sample sizes need to be large enough to maintain sufficient power—that is, to detect a difference when one exists (Taylor and Gerrodette, 1993).

Most vegetation monitoring methods quantify different measures of abundance. These include cover, biomass, frequency, and density, all of which can be used to derive dominance, and, indirectly, species composition (Bonham, 2013). These methods typically involve sampling using fixed-area plots of varying sizes. Vegetation within plots is sampled with multiple quadrats, belt transects, lines, or points (that is, subsamples). Measurements from these subsamples are then summarized as proportions (for example, percent cover) or as some measure of central tendency (for example, average cover) to represent the vegetation within the plot. Data are often summarized by species, lifeform, or functional group. In addition, ground cover (for example, bare ground, litter, rock) or canopy gap data may be collected. Careful comparisons among methods by researchers (for example, Stohlgren and others, 1998; Seefeldt and Booth, 2006; Godínez-Alvarez and others, 2009; Pilliod and Arkle, 2013) have enabled monitoring data that were collected using different methods to be combined. However, all methods have sampling biases and different levels of precision; these should be considered carefully when combining datasets.

Two common vegetation sampling methods are associated with line transects (Elzinga and others, 2001). The line-point intercept method tallies the number of intercepts ("hits") along a transect, usually at evenly spaced intervals. Multiple transects are usually placed in a plot, often parallel to each other or in a spoke design (for example, Herrick and others, 2009). Alternatively, the line-intercept method measures the length of a line that is intercepted by vegetation.

Several methods are associated with area sampling within fixed-area plots or subplots. The quadrat method uses multiple small sampling frames placed on the ground, typically along multiple transects within a macroplot (Elzinga and others, 2001). Vegetation cover in the quadrats is either visually estimated or counted systematically at intercepts of grid points (that is, grid-point intercept). Biomass is usually quantified in quadrats by clipping and weighing current year's growth. Density (the number of units [individual plants or stems]/sample area) is typically recorded in either quadrats or belt transects. Belt transects are like quadrats but elongated, often along a transect tape (for example, 1 meter [m; 3.3 feet {ft}] x 25 m [82 ft]). Finally, plotless methods or distance measures (for example, point-center quarter, nearest neighbor) can also be used to estimate density of plants that are randomly distributed or occur at low densities, and time- or area-constrained visual searches are useful for detecting rare plants (Elzinga and others, 2001).

Frequency is the presence (or absence) of a species (for example, lifeform, functional group member) rooted within a fixed area plot or quadrat. It is reported as the percentage of all possible plots/quadrats within a sample area in which a species is present. Plot or quadrat size strongly affects the percent frequency; selecting the appropriate size depends on the size and distribution pattern of the vegetation. Frequency has been used to infer abundance, but it is not the same as cover. However, in areas that are grazed, it is commonly used in lieu of cover estimates because, in theory, herbivory should not influence species presence as much as species cover. This holds at least until heavy or repeated herbivory begins to eliminate species when both metrics converge towards zero.

Finally, a well-designed, random (but representative) sample offers the best opportunity for detecting relevant trends in resources with maximum inference for areas of interest (Urquhart and others, 1998). In the sagebrush biome, which is heterogeneous owing to soil, topography, and climate, sampling designs often require spatial stratification to improve meaningful representation of resources or environmental conditions. This approach to rangeland vegetation monitoring is increasingly being implemented across multiple spatial scales and by many agencies and organizations (Herrick and others, 2010; Toevs and others, 2011; Barker and others, 2018). Nonrandom monitoring and convenience sampling still occurs but has limited inference and is difficult to roll up for multiscale assessments.

Examples of Vegetation and Habitat Monitoring Programs

Several monitoring programs have been developed by Federal agencies to address status and trends of resources on public and private lands. The U.S. Department of the Interior, Bureau of Land Management (BLM) Assessment Inventory and Monitoring (AIM) and the U.S. Department of Agriculture, Natural Resources Conservation Service (NRCS), National Resources Inventory (NRI) both use core indicators and standardized protocols. The U.S. Department of Agriculture,

Nested, Hierarchical Adaptive Management

The focus for this Sagebrush Conservation Strategy is on a nested, hierarchical adaptive management construct:

Local Scale

- Build adaptive management construct into local sagebrush conservation strategies;
- Orient around the goals of all relevant stakeholders,
- Predict what is needed or what actions to take (for example, restoration) to meet resource, objectives (for example, forage, security cover) explicitly described;
- Assess progress through onsite monitoring; and
- Model pathways and feedback loops explicitly.

Midscale, Ecoregional

- Focus on major drivers to the system and actions needed to meet ecoregional goals, set ecoregional quantitative goals with respect to major drivers and evaluate through monitoring (for example, trends in annual grass infestation, conifer encroachment, restoration of major fires);
- Evaluate progress toward goals by summing number of projects, acreage treated, success, or other variables of local scale management actions by monitoring (most likely remotely) the extent and coverage of sagebrush, multiyear trends in invasive plant species distribution, fire frequency, acres burned, and more;
- Incorporate ecoregional-level monitoring of sagebrush-dependent species as a metric for assessing the success of sagebrush conservation strategies and efforts; and
- Incorporate explicit metrics into ecoregional models to iteratively evaluate whether and where additional conservation efforts are needed or whether assumptions or goals need to be changed at local scales.

Biome Scale

- Similar to the ecoregional scale but with biome-wide goals and assessed through monitoring at biome-wide levels (for example, remotely monitoring the extent and coverage of sagebrush, multiyear trends in invasive plant species distribution, fire frequency and acres burned, across all ecoregions);
- Incorporate biome-wide trends in sagebrush-dependent species by aggregating ecoregional monitoring as a metric for use assessing the success of sagebrush conservation strategies and efforts; and
- Incorporate explicit metrics into biome-wide models to iteratively evaluate whether and where additional conservation efforts are needed or whether assumptions or goals need to be changed at ecoregional scales.

Example

The ARM theory is well-developed. However, implementation, especially at broader scales, has not paced theoretical development. There are State and State/Federal collaborative adaptive management programs that primarily target game species for which harvest or other removal is potentially a factor limiting populations of these species. Examples include harvest management under the North American Waterfowl Management Plan (U.S. Department of the Interior, Environment Canada, and Environment and Natural Resources Mexico, 2018), big game management programs within State wildlife agencies, and the Mourning Dove Harvest Strategy coordinated by the U.S. Fish and Wildlife Service. These programs all include monitoring of population levels and trends, usually through modeling supported by indices of abundance, and feedback to adjust harvest or removal rates in support of larger population goals. A major weakness of all these adaptive management constructs is that while they provide feedback to regulate harvest, there is little to no monitoring of habitat and no feedback of habitat data to influence land use decisions affecting habitat.

Forest Service, Forest Inventory and Analysis (FIA) program uses a different set of indicators but also uses standardized protocols. Although FIA and NRI/AIM use different sampling methods, their sample designs allow for combined analyses of pooled data so that periodic assessments can be rolled up across spatial scales of interest using a nested hierarchy (Patterson and others, 2014). Each program is described below, with more information in appendix S1 (table S1.1).

NRCS National Resources Inventory Rangeland Resource Assessment

The NRCS NRI rangeland resource assessment provides information on the trends of land soil, water, and related resources on the Nation's non-Federal lands (accessible at https://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/technical/nra/nri/results/?cid=nrcseprd1343025). A spatially balanced, randomly located sampling design provides land area estimates for qualitative and quantitative indicators related to rangeland health. Quantitative indicators include bare ground, plant species cover and composition, gaps between plant canopies, and soil stability according to the standard methods in Herrick and others (2009). The qualitative indicators of rangeland health (Pellant and others, 2005, 2020) are also assessed at each site. Results are reported to Congress as part of the Resource Conservation Act Appraisal (RCA;16 U.S.C. 2001–2009) and are increasingly used for other applications including research (for example, Herrick and others, 2010).

BLM Assessment, Inventory, and Monitoring Strategy

The BLM monitors public rangelands as part of the AIM strategy (Toevs and others, 2011), which provides a consistent and repeatable monitoring methodology to collect detailed quantitative information on rangeland vegetation condition. The AIM Strategy informs the BLM of resource status, condition, and trend at multiple spatial scales ranging from management units (for example, grazing allotments or treatments) to national-level assessments (Karl and others, 2016). Standard indicators (MacKinnon and others, 2011) are measured using the same methods as the NRCS NRI (Herrick and others, 2009). Many AIM efforts employ a stratified, randomized sample design to enable a statistically valid extrapolation across different spatial scales and reporting units, with greater sampling intensity in areas where issues have been detected or treatments are being monitored. Plots are resampled at 5-year intervals to detect trends over time. The AIM data are captured and managed electronically, which helps ensure data quality and facilitates centralized data storage, analysis, and reporting.

USDA Forest Service Forest Inventory and Analysis

The USDA Forest Service FIA is a national program for collecting and reporting information on status and trends in forest ecosystems. Forested vegetation data are collected across all land ownerships. The FIA programs also consistently collect data on nonforested land on National Forest System lands in California, Idaho, Montana, North Dakota, and parts of Oregon, Utah, Washington, and Wyoming. Because this covers most of the sagebrush distribution, the FIA dataset can be useful at broad scales. Canopy cover is estimated on the four most dominant species within a lifeform that are present within each (of the four 169 square meter [m^2] [1/24-acre]) subplots (within the -plot primary sample unit) that have at least 3-percent cover. In addition, line-point intercept is conducted to quantify ground cover (bare soil, rock, basal vegetation, and litter) composition for each of the four subplots.

Habitat Assessment Framework

The Sage-Grouse Habitat Assessment Framework (HAF; Stiver and others, 2015) is a multiscale assessment of sage-grouse (*Centrocercus* spp.) habitat suitability. The HAF is the primary assessment method used to evaluate the wildlife standard in the BLM land health evaluation process and is used by other Federal and State agencies to characterize sage-grouse habitat suitability. National forests and grasslands implementing the joint BLM/Forest Service sage-grouse land use plan amendments also use HAF to assess sage-grouse habitat. The HAF rates sage-grouse habitat suitability across four spatial scales: rangewide (first order), population (second order), subpopulation (third order), and seasonal habitat areas (fourth order). The second and third order HAF assessments evaluate the availability and continuity of sagebrush habitat at a landscape scale. At the seasonal habitat (fourth order) scale, HAF uses standardized monitoring data from AIM plots, as well as supplemental indicators, to rate sage-grouse habitat suitability across seasonal habitat areas based primarily on vegetation composition and structure. These monitoring indicators are summarized into overall suitability ratings at each plot, which are aggregated across seasonal habitat areas to determine sage-grouse habitat suitability. This coordination effort addresses two critical challenges that Federal land management agencies face today: (1) field capacity to complete monitoring data collection and (2) the ability to share and combine data to conduct data analysis across administrative boundaries.

Interpreting Indicators of Rangeland Health

Interpreting Indicators of Rangeland Health (IIRH) is a qualitative assessment protocol for rangelands (Pellant and others, 2005, 2020). The IIRH provides a preliminary evaluation of the current status of three attributes of rangeland health: soil and site stability, hydrologic function, and biotic integrity. This assessment is conducted by an interdisciplinary team observing and rating 17 indicators related to rangeland ecosystem functions. The IIRH is not meant as a stand-alone tool for monitoring rangelands or determining trend, but it is often used either prior to or in conjunction with quantitative monitoring efforts including BLM AIM and NRCS NRI.

Project-Level Monitoring

Project-level monitoring, including measures of condition and of change following disturbance, occurs throughout the sagebrush ecosystem at local scales and for a variety of purposes. Capturing the extent and diversity of those efforts is beyond the scope of this chapter; however, the individual efforts often provide critical guidance to subsequent management actions, in an adaptive management context, at the project scale. A good example is postfire Emergency Stabilization and Rehabilitation (ESR) monitoring associated with the 2015 Soda Fire in southwest Idaho and eastern Oregon. Led by BLM, this ESR monitoring mostly focused on implementation of many treatments, but collaboration with scientists at the U.S. Geological Survey (USGS) has included effectiveness and validation monitoring (for example, Germino and others, 2018; Davidson and others, 2019).

Project-level and postdisturbance monitoring can take many forms, from quantifying vegetation composition at the species or functional group level to photo points taken at certain time intervals. Project-level monitoring also occurs through programs like the NRCS Sage Grouse Initiative (SGI), where ranch-scale monitoring tracks condition, allows the individual producer to see firsthand the benefits of conservation practices, and provides the mechanism for long-term conservation. This monitoring instills in the producer the benefits of sustainable grazing systems in their operation and to sage-grouse conservation. Many other agencies and entities conducting restoration treatments in the sagebrush ecosystem also collect monitoring information, and some agencies require some posttreatment monitoring as part of their reporting to receive grant funds. Project-level monitoring also occurs as part of long-term research programs designed and carried out by scientists to track changes in vegetation and the biological response of sage-grouse populations to conservation practices.

State Agency Vegetation Monitoring Efforts

Many State agencies collaborate with the BLM to apply HAF for assessing sage-grouse habitat in their States. In addition, many States have developed habitat quantification tools (HQTs) that are used for mitigation programs. These tools are used to measure habitat function to quantify gains in sage-grouse habitat resulting from activities that restore, enhance, or preserve habitat, as well as losses resulting from activities that disturb, fragment, or eliminate habitat. Some States have also adopted individual monitoring and assessment protocols to address sagebrush vegetation and sage-grouse habitat quality; two examples follow.

The State of Oregon adopted simplified state-and-transition models, referred to as threat-based models (Johnson and others, 2019), as a framework for identifying and addressing the primary ecosystem threats to upland sagebrush ecosystems. Vegetation condition is described by ecological states that indicate current vegetation composition and the level of risk from major ecological threats like fire, conifer encroachment, and invasive annual grasses. Transitions between categories may be caused by natural disturbances (for example, drought or wildfire) or by management actions (for example, grazing, juniper [*Juniperus* spp.] removal, prescribed burning). Ecological states are described in easily understood terms, from “A” or “B” for relatively good condition with minimal threats expressed, to “C” for moderate conditions that require management changes to address threats, and then to “D” or “E” for poor conditions with high threat levels. Threat-based models are central to the 2015 Oregon Sage-Grouse Action Plan (Sage-Grouse Conservation Partnership, 2015), forming the basis of the State’s HQT and applied at scales from individual mitigation projects and U.S. Fish and Wildlife Service (FWS) candidate conservation agreements to statewide mapping and assessment of state wildlife action plan effectiveness. Although they have been used for sage-grouse planning in the State, they are ecosystem models that are not species-specific and can be used alongside species-specific methods, such as HAF, to paint a fuller picture of the ecosystem threats affecting sagebrush-obligate species.

In Nevada, the Department of Wildlife monitors approximately 2,000 plots across the State and into the California side of the Bi-State sage-grouse priority management units. Monitoring began in 2011 with the goal of evaluating the effectiveness and efficiency of habitat projects for sage-grouse. With validation in mind, most plots are placed in specific projects that allow for comparisons between treated and untreated areas. Monitoring methods mostly follow the AIM protocol, although the State has partnered with the Forest Service to implement HAF in some areas.

Remote Sensing and Geospatial Data for Monitoring

The use of remote sensing and geospatial datasets can provide tools for monitoring at multiple spatial scales. The increasing availability of remote sensing imagery has offered the potential to characterize and monitor conditions of sagebrush-dominated ecosystems at broad spatial and temporal scales (Kennedy and others, 2014). Given that satellite imagery, such as Landsat, dates back to the 1970s and 1980s, these technologies can provide a consistent approach across the sagebrush biome to monitor implementation of management activities and changes to habitat attributes, such as extent and condition of sagebrush and factors that contribute to habitat degradation.

Continuous remote sensing of vegetation has been available through the USGS Landsat Program since 1972 (U.S. Geological Survey, 2016). Landsat and other satellite, aerial, and ground-based sensors provide standardized metrics for evaluating vegetation productivity (Rouse and others, 1974) and other characteristics (Jensen, 2005). Use of these data products enabled the implementation of thematic vegetation mapping and laid the groundwork for many of the current monitoring programs. The Interagency Greater Sage-Grouse Monitoring Framework (Interagency Greater Sage-Grouse Disturbance and Monitoring Subteam, 2014) outlines standardized protocols for using LANDFIRE (U.S. Geological Survey, 2013b) and other map products to track loss and shifts in landscape attributes and vegetation characteristics that are critical for sagebrush-associated wildlife. Multiple remote sensing products are now available to characterize and monitor rangeland vegetation, including continuous cover maps of rangeland vegetation such as trees, shrubs, sagebrush, total herbaceous, and invasive annual herbaceous vegetation (see app. S2). Remote sensing can also be used to monitor other threats to sagebrush ecosystems. Fires are mapped annually through the USGS Monitoring Trends in Burn Severity (MTBS) program (Eidenshink and others, 2007), the Geospatial Multi-Agency Coordination (GeoMAC) wildfire application (https://www.geomac.gov/GeoMACTransition.shtml), and other programs. Sagebrush loss through agricultural conversion and urban development can also be monitored through programs like the USDA National Agricultural Statistics Service and Multi-Resolution Land Characteristics consortium (multiagency).

As technology has advanced, the capabilities and capacity of agencies and organizations to rapidly develop information to track changes in ecosystem condition have increased dramatically. However, applying remotely sensed maps as part of a monitoring program can be challenging. Although all datasets have limitations, a full understanding of the assumptions, error sources, scale, and limitations of each product is especially important for remotely sensed maps. While mapping technology has improved dramatically, localized errors (for example, inability to precisely reproduce spatial patterns at fine scales) and other accuracy issues (for example, overall bias of predicted values such as an inability to predict where a condition is absent) can limit the ability for mapping vegetation condition, particularly at smaller spatial scales. Most applications of remotely sensed products in rangeland monitoring use products at broad scales (for example, rangewide analyses such as the Interagency Greater Sage-Grouse Monitoring Framework [Interagency Greater Sage-Grouse Disturbance and Monitoring Subteam, 2014] or statewide assessment of habitat condition). Remotely sensed maps hold great promise for tracking changes over time across large landscapes, but accounting for map error is needed for robust change detection analysis. Maps can also be difficult to interpret along with other sources of information, including vegetation plot data, other datasets, and expert knowledge, and there is a need for examples of how to apply maps to management applications at finer management-relevant scales such as grazing allotments. However, technology in remote sensing and computational processing is rapidly evolving, and maps should continue to improve over time.

Additional Datasets for Monitoring and Adaptive Management

As agencies collect and compile spatially referenced data in the course of their functions, these datasets could offer opportunities to study and monitor management across landscapes. For example, the Land Treatment Digital Library (LTDL) has compiled thousands of land treatment records dating back to 1940 from BLM field and district offices across the western United States (Pilliod and others, 2017b). As this dataset is developed and maintained, the LTDL could provide a systematic record of land treatments that could serve a variety of applications including adaptive management and retrospective analyses (for example, Pilliod and others, 2017b; Copeland and others, 2018). Another data source is the FWS's Conservation Efforts Database (CED), which maintains records of conservation and restoration actions on private and public lands targeting sagebrush habitat (Heller and others, 2017). Other useful records relate to livestock grazing on public lands (Veblen and others, 2011; Monroe and others, 2017). The BLM maintains records each year of the reported livestock use (billed use animal unit month [AUM]) and the maximum number of AUMs authorized (permitted active use) in each allotment. These data represent one of the most complete and widespread records of livestock across the western United States and may provide insights into the relationships between the timing of grazing and rangeland condition or sage-grouse population trends (for example, Monroe and others, 2017).

Challenges and Opportunities for Vegetation Monitoring

Vegetation monitoring in sagebrush ecosystems has evolved through time and improved as natural resource managers have adopted inferential sampling designs and standardized methods. However, there remain gaps in vegetation monitoring approaches. One area of improvement is the frequency of monitoring and the length of time following restoration or other types of land treatments. Vegetation monitoring programs, such as AIM, frequently struggle to balance the costs of revisit frequency (for example, yearly, every other year, every fifth year) against increased spatial coverage (that is, more plots). Most management actions provide insufficient funding to perform monitoring for more than a few years, and thus, most project-level monitoring falls into implementation monitoring and not effectiveness monitoring. Some restoration outcomes take years to discern, so a commitment to longer term monitoring efforts is often needed. Monitoring programs used by different agencies, and sometimes within the same agency, are rarely integrated. This integration could increase inference and power, but also cost efficiency. Ultimately, monitoring programs, whether distributed across the sagebrush biome or at the project level, are constrained by limited funding. Perhaps the most practical way to alleviate this constraint is to increase efficiency through better partnerships and data sharing. Both approaches require communication, standardization of methods, and a commitment to value monitoring data as a source of information for adaptive management.

Wildlife Monitoring

The use of monitoring data in the conservation and management of wildlife populations requires a foundation of well-articulated monitoring objectives (Sauer and Knutson, 2008; Lindenmayer and Likens, 2010). For example, management and conservation objectives from the U.S. North American Bird Conservation Initiative (NABCI) are to (1) determine the status and trends of populations, (2) inform management and policy to achieve conservation, (3) evaluate conservation efforts, (4) inform conservation planning, (5) set population objectives and management priorities, and (6) determine causes of population change (U.S. North American Bird Conservation Initiative Monitoring Subcommittee, 2007). Monitoring long-term trends in occupancy, abundance, or demography provide some of the most useful data for the conservation planning process to prioritize and assess the vulnerability of wildlife species (Rosenberg and others, 2017). However, population trends without reference to monitoring objectives have limited utility for evaluating species responses to conservation and management (Nichols and Williams, 2006).

Population density or abundance metrics are essential for wildlife conservation and important for estimating the effect of management actions on wildlife species (Nichols and others, 2007; Smith and others, 2013). For example, a conservation objective for sage-grouse is to maintain annual counts of male sage-grouse at leks within a desired range or relative to a baseline. The State lek monitoring programs can be used to estimate sage-grouse abundance, population trends (McCaffery and others, 2016; Coates and others, 2018) and regional population size (Shyvers and others, 2018). These monitoring data can then be used in the adaptive management process to predict sage-grouse population responses to management alternatives and to determine which management alternatives attain the population size objective. Of course, the development of conservation objectives for sage-grouse at multiple scales will require careful deliberation among decision makers and stakeholders in the problem-definition stage of the adaptive management process (Coates and others, 2017d). Another potential objective could be to maintain population size above a threshold (Martin and others, 2009), and this must be considered in tandem with socioeconomic objectives in the region. Abundance may be a useful state variable for other sagebrush species of conservation concern, although occupancy may be more realistic given the challenges of monitoring most species. One exception appears to be population density of sagebrush birds, which can be quantified using point-count methods and evaluated with respect to management alternatives, such as conifer removal (Holmes and others, 2017) and prescribed grazing (Golding and Dreitz, 2017). Estimating abundance, however, requires larger sample sizes than site occupancy (Joseph and others, 2006; Noon and others, 2012) and may be more appropriate for well-studied, abundant, and conspicuous species, such as birds and native ungulates.

Monitoring population parameters, including movement and demographic or vital rates, provide mechanistic explanations for population change in response to management over time (Sandercock, 2006). Demographic or vital rates include the annual estimates of survival, production, and recruitment that are the ultimate cause of population dynamics. These parameter estimates are a powerful way to assess species responses to habitat management actions in sagebrush ecosystems (for example, Zeoli and others, 2008; Taylor and others, 2012; Doherty and others, 2014; Pilliod and Scherer, 2015; Dahlgren and others, 2016b; Coates and others, 2017d). The costs involved with monitoring population parameters with respect to management alternatives can be considerable because they often require mark-recapture methods, telemetry, or direct observation (for example, nest monitoring). The cost of obtaining this level of information needs to be weighed carefully against the value or necessity of the information to determine if the effort is necessary. As previously stated, the value or necessity of the information is determined when objectives are established by stakeholders and assessed relative to the degree of acceptable uncertainty in the population parameters (Canessa and others, 2015; Maxwell and others, 2015).

Site occupancy is an alternate state variable for wildlife conservation involving the extent of occurrence or geographic range of species (MacKenzie and Nichols, 2004; Noon and others, 2012). Multispecies occupancy models provide a community framework for monitoring the responses of individual species to management alternatives, with species richness summarized across the individual species' responses (Zipkin and others, 2010; Sauer and others, 2013). For example, an objective for the adaptive management of multiple sagebrush species of various taxa can be developed to maximize species richness as the cumulative occupancy of the species (Sauer and others, 2013). Objectives for occupancy dynamics include estimating local extinction and colonization to provide greater understanding of range expansion or contraction in response to management actions (Bled and others, 2013). Species richness of sagebrush wildlife may be best evaluated in an umbrella species framework (Nicholson and Possingham, 2006), with the objective of maximizing species richness when population size or population growth of a representative species is above an acceptable threshold. The sage-grouse has been suggested as an umbrella species for sagebrush wildlife species (Rowland and others, 2006), although there is disagreement over the effectiveness of this approach (Hanser and Knick, 2011; Norvell and others, 2014; Carlisle and others, 2018b; Runge and others, 2019; Timmer and others, 2019; chap. Q, this volume). The ability of adaptive management to accommodate multiple objectives will allow an evaluation of individual species' responses to management alternatives, and this will provide a framework for learning about the linkage between objectives for multiple sagebrush wildlife species and sage-grouse. Although adaptive management often involves directly evaluating the effectiveness of management alternatives (Nichols and Williams, 2006; Lyons and others, 2008), objectives based on habitat relationships can be used to indirectly predict species' responses to changes in habitat structure in response to vegetation management (Marcot, 2006; Aldridge and Boyce, 2007). Objectives defined by habitat relationships present an opportunity to monitor the performance of management alternatives in terms of vegetation responses to management. However, because habitat relationships are correlational rather than causal, effectiveness monitoring may be necessary to validate and update the predicted responses to changes in vegetation structure (Marcot, 2006).

State variables for rare and cryptic taxa with limited data can still be developed using a combination of qualitative data and expert opinion (Nyberg and others, 2006; Choy and others, 2009). For example, occurrence objectives for data-deficient species can be developed from range and distribution maps derived from opportunistic data (NatureServe, 2019), and expert opinion can be used to predict species responses to management alternatives (Kuhnert and others, 2010). Objectives initially formulated with qualitative data and expert opinion are justifiable on the basis that, rather than wait for definitive data, it is preferable to start the adaptive management processes with limited data and uncertain responses to management with the understanding that monitoring the relative performance of management alternatives and updating model results will reduce uncertainty over time (Williams and Brown, 2012; Neckles and others, 2015).

Adaptive Management and Monitoring of Nongame Species

There are relatively few national, regional, or State-level adaptive management or monitoring programs for nongame species in sagebrush ecosystems with the exception of songbirds. The distribution and status of most nongame mammals are rarely assessed, although interest in lagomorphs (for example, pygmy rabbits [*Brachylagus idahoensis*]) and bats has increased recently in sagebrush ecosystems. Reptiles and amphibians tend to be data-deficient, even though some species garner attention (chap. I, this volume).

Monitoring programs for songbirds provide our best example of nongame monitoring. The North American Breeding Bird Survey (BBS; Bystrak, 1981) and the Integrated Monitoring in Bird Conservation Regions (IMBCR; Pavlacky and others, 2017) provide data sources for estimating the site-occupancy and population size of sagebrush-obligate bird species. The primary objective of the BBS is to provide an index of abundance that can be used to estimate population trends and relative abundances at various geographic scales. The BBS covers the entire sagebrush biome, but some intermountain regions in Montana and Nevada have low numbers of routes. The IMBCR program provides defensible estimates of avian abundance and occupancy, designed to meet the NABCI goals for improving avian monitoring and is well suited for addressing multiple management and conservation objectives (U.S. North American Bird Conservation Initiative Monitoring Subcommittee, 2007). In addition, IMBCR accommodates a stratification scheme for effectiveness monitoring of habitat restoration, as well as local-scale habitat associations for predicting species responses to vegetation management (Pavlacky and others, 2017). The IMBCR program currently covers the eastern portion of the sagebrush ecosystem and has recently expanded to include Utah and BLM-administered lands in Nevada and Oregon. The BBS and IMBCR programs can both incorporate remotely sensed data to evaluate objectives for multiple species with respect to management alternatives such as conifer removal (Donnelly and others, 2017).

Amphibian monitoring is organized under USGS's Amphibian Research and Monitoring Initiative, although with less emphasis in the sagebrush biome. Most amphibian monitoring in sagebrush ecosystems is used to determine the status and trends of species petitioned for listing under the Endangered Species Act of 1973 (ESA; 16 U.S.C. 1531 et seq.). Examples include the Columbia spotted frog (*Rana luteiventris*) and western toad (*Anaxyrus boreas*). Monitoring these species involves visual encounter surveys of wetlands to document occupancy and evidence of reproduction.

Sometimes a few focal populations are monitored more intensively using mark-recapture methods to estimate population size and vital or demographic rates. Nevada employed all these strategies in a 10-year monitoring effort for the Columbia spotted frog in 2015 ahead of a not-warranted decision by the FWS (McAdoo and Mellison, 2016).

Adaptive Management and Monitoring of Game Species

Game management provides a useful example of adaptive management (see “North American Waterfowl Management Plan” sidebar). However, game management does not include quantification of habitat quality metrics. Game management aims to monitor annual population changes based on abundance data and hunter success data (estimates rely on data collected from surveys of hunters). As an example, upland game harvest success data are based on a random sample of hunters that purchased upland game hunting licenses. General surveys have inherent bias, such as nonresponse bias associated with higher survey return rates from successful hunters. Most State agencies have reduced reporting bias by increasing survey effort via permits, phone interviews, or web-based surveys, producing a random sample of species-specific hunters (for example, sage-grouse hunters).

As an example of partial adaptive management for a game species, sage-grouse harvest monitoring includes abundance monitoring based on lek counts, hunter surveys, and in some States, wing returns. Analysis of grouse wings provides ratios of males to females, ratios of chicks to females, and potentially nest success information. These ratios can provide productivity estimates to assess habitat quality across time but only at large scales. Counts of adult male sage-grouse on leks during the spring are the primary source of information used to assess populations and set appropriate regulations for the following hunting season. Unfortunately, there is a mismatch with the estimated population size to be hunted because productivity occurs in between the population assessment timeframe and when harvest occurs in the fall. Generally, lek trends are used to recommend season regulations by hunting unit, including season start date, season length, bag and possession limits, and areas open for hunting. Public input is also solicited in this process. Hunting season closures may occur in response to major habitat disturbances (for example, wildfire) or following outbreaks of disease (for example, West Nile virus), or when small populations decline to management triggers.

Most western States use some variation of adaptive harvest management (AHM) to manage big game populations. Not unlike vegetation components of the sagebrush ecosystem (of which several State-trust game species intersect), game resources require careful and increasingly intensive management to accommodate the many and varied public demands and growing impacts from people. Ideally, management of big game populations follows a “management by objective” approach. The primary objectives are based on how many animals should exist in a hunting unit and what is the desired sex ratio for the population (for example, the number of males per 100 females). The selection of

Bird Conservancy of the Rockies—Decision Support Tool

In an example of integrating monitoring and management for nongame species, the Bird Conservancy of the Rockies and partners developed a prototype web-based decision making tool (Bird Conservancy of the Rockies, http://rmbo.org/DST) to answer key management questions surrounding livestock grazing on privately owned or leased sagebrush rangelands (Cagney and others, 2010), as well as conservation objectives for greater sage-grouse (*Centrocercus urophasianus*; Manier and others, 2013) and sagebrush-dependent songbirds (Knick and others, 2003). The objectives of the tool are to maximize the (1) occurrence of sagebrush-dependent songbirds, (2) suitability of greater sage-grouse nesting habitat, and (3) forage production for sustainable livestock production. The tool evaluates alternative U.S. Department of Agriculture, Natural Resources Conservation Service (NRCS) practices for prescribed grazing and brush management to improve nesting habitat for greater sage-grouse (U.S. Fish and Wildlife Service, 2010). The tool is based on existing management and planning methods, and includes State wildlife habitat evaluation guides, NRCS state-and-transition models, and important ecological site descriptions for greater sage-grouse. The predicted responses of sagebrush-dependent songbirds to the management actions were based on local-scale habitat relationships and landscape-scale distribution models from the Integrated Monitoring in Bird Conservation Regions (IMBCR) program (Pavlacky and others, 2017). In addition, the tool is compatible with ongoing conservation initiatives in the range of the greater sage-grouse and was designed to input preproject vegetation data collected by NRCS resource inventories. Finally, the tool integrates stakeholder objectives, conservation practices, and data-driven predictions to identify win-win solutions for sustainable livestock production and multispecies conservation of sagebrush birds. The management tool can be easily extended to adaptive management by including data for postproject effectiveness monitoring (Nyberg and others, 2006).

North American Waterfowl Management Plan

As an illustration of adaptive management in action, U.S. Department of the Interior's Adaptive Management Applications Guide (Williams and Brown, 2012) describes how the Harvest Management Working Group uses adaptive management to inform waterfowl harvest regulations. The adaptive harvest model (AHM), better described as a process rather than a model, incorporates waterfowl population data collected annually into population models to inform development of hunting season regulations. Each year, monitoring activities such as aerial surveys and hunter questionnaires provide information on population size, habitat conditions, and harvest levels. Data collected from this monitoring program are analyzed each year, and proposals for duck-hunting regulations are developed by the flyway councils, States, and U.S. Fish and Wildlife Service (FWS). After extensive public review, the FWS announces a regulatory framework within which States can set their hunting seasons.

Adaptive Harvest Model Components

- A limited number of regulatory alternatives that describe Flyway-specific season lengths, bag limits, and framework dates;
- A set of population models describing various hypotheses about the effects of harvest and environmental factors on waterfowl abundance;
- A measure of reliability (probability or "weight") for each population model; and
- A mathematical description of the objective(s) of harvest management (that is, an objective function) by which alternative regulatory strategies can be evaluated.
- Components are used in a stochastic optimization procedure to derive a regulatory strategy that specifies the appropriate regulatory alternative for each possible combination of breeding population size, environmental conditions, and model weights. The setting of annual hunting regulations then involves an iterative process:
- Each year, an optimal regulatory alternative is identified based on resource and environmental conditions and on current model weights;
- After the regulatory decision is made, model-specific predictions for subsequent breeding population size are determined;
- When monitoring data become available, model weights are increased to the extent that observations of population size agree with predictions and decreased to the extent that they disagree; and
- The new model weights are used to start another iteration of the process.

The AHM approach explicitly recognizes that the consequences of hunting regulations cannot be predicted with certainty and provides a framework for making objective decisions in the face of that uncertainty. The process is optimal in the sense that it provides the regulatory choice necessary, each year, to maximize management performance. The process is adaptive in the sense that the harvest strategy "evolves" to account for new knowledge generated by a comparison of predicted and observed population sizes. Inherent in the adaptive approach is awareness that management performance can be maximized only if regulatory effects can be predicted reliably. Thus, the AHM approach relies on an iterative cycle of monitoring, assessment, and decision making to clarify the relationships among hunting regulations, harvests, and waterfowl abundance. Despite its limits, the AHM is considered one of the most successful wildlife management programs in North America (Williams and Johnson, 1995; Johnson and Williams, 1999; Williams and others, 2002).

population and sex ratio objectives drive important decisions in the big game season setting process, namely, how many animals need to be harvested to maintain or move toward the objectives, and how are hunting seasons managed to achieve the harvest objective. Most big game AHM constructs lack any explicit habitat component in their modeling approaches. Consequently, they are limited in their ability to respond when harvest management is not an effective tool, for instance when populations are chronically below objective because of long-term declines in habitat quality, quantity, or both.

In summary, game management programs are good examples of adaptive management because they start with broad strategic and population-level management plans which describe quantitative population and performance (for example, doe/fawn ratios) objectives that are based on scientific underpinnings (ongoing monitoring data and models). Annual objectives (for example, harvest quota) are adjusted in some cases because of other sources of mortality, public involvement, and other factors (for example, to reduce damage to property). Cyclic repetition with annual adjustments and consideration of uncertainty and stochasticity represent an AHM approach. Below is an example for waterfowl management that could easily be applied to sage-grouse, for example (see “North American Waterfowl Management Plan” sidebar). However, game management AHM approaches could be improved by the addition of an explicit habitat component that would illustrate the nature and extent of habitat improvements needed to achieve objectives.

Challenges and Opportunities to Implement Adaptive Management for Wildlife

The gaps in wildlife monitoring approaches are often identified when setting priorities for measurable objectives in the problem definition phase of adaptive management. The objectives must be established ahead of the management interventions and before monitoring designs are developed (Lyons and others, 2008). When setting objectives for wildlife, rather than anchoring on the availability of existing data, it is preferable to develop objectives to solve the most pressing conservation problems in sagebrush ecosystems. However, the development of measurable objectives and monitoring designs are an iterative process that often involves evaluating the cost and feasibility of monitoring. Data gaps for the response of wildlife species to management creates uncertainty about the consequences of the management actions (Williams and Brown, 2012). Although there is often institutional resistance to acknowledging uncertainty, adaptive management provides a framework for addressing and reducing uncertainty through the process of management itself (Williams and Brown, 2012). Adaptive management can increase the cost-effectiveness of management and monitoring, but because the process requires considerable time investments on the front-end and continuity to monitor management alternatives on the back-end (Williams and Brown, 2012), implementation of adaptive management across the sagebrush biome faces obvious funding constraints.

Although this chapter provides several examples of successful implementation of ARM for wildlife species and populations in North America, existing programs and approaches also have several shortcomings. First, these iterations of ARM are largely single-species approaches that are not likely to effectively conserve the full breadth of sagebrush-associated taxa. Second, the programs described are, for the most part, funded through license fees and dedicated Federal programs such as Pittman-Robertson for single-species management. Those kinds of funding sources are not expected to be available for most sagebrush species, guilds, communities, or whatever target/ecological unit is identified. Existing adaptive management programs are not typically based on random survey designs and are not standardized among all harvest units, among populations, or across governing entities; in some cases, known technical and analytical flaws persist because of institutional or capacity limitations. Standardization of survey techniques and implementation of random survey designs would allow for better inference related to population trajectories across time (for example, Robust Design surveys). These concepts would reduce inherent sampling bias present in current surveys. Furthermore, in most cases, few, if any, other critical factors are used to inform decisions (for example, habitat extent, quantity, or quality). Spatially explicit surveys would allow wildlife monitoring (abundance or indices) to be related to habitat quality by comparison to habitat data derived from field plots or geographic information system analysis. Also, the targets of existing programs consistently have economic value and active user-bases, neither of which may be the case for many sagebrush-associated taxa.

Advances in technology, statistical design, model integration, and shared conservation planning methods provide opportunities to consider and initiate ARM for multiple taxa and ecological systems. Monitoring programs are getting stronger and more robust, including integration of habitat and population modeling. Advances in remote sensing and data management processes now provide opportunities not available before. Policy makers, agency leaders, and biologists are now recognizing that data-driven management with appropriate feedback loops (that is, effective ARM) will help prevent species from being petitioned or listed under the ESA, an event that would further constrain management options.

Acknowledgments

We thank the many natural resource managers and scientists that contributed to this chapter through thoughtful, insightful, and provoking conversations about monitoring and adaptive management in the sagebrush biome and elsewhere.

Appendix S1. Comparison of Federal Monitoring Programs in Rangelands

Table S1.1. Comparison of Federal monitoring programs in rangelands.

[<, less than; BIA, Bureau of Indian Affairs; NRCS, U.S. Department of Agriculture, Natural Resources Conservation Service; BLM, U.S. Department of the Interior, Bureau of Land Management, USDA, U.S. Department of Agriculture]

Protocol	National Resources Inventory (NRI)	Assessment Inventory and Monitoring (AIM)	Forest Inventory and Analysis (FIA)
Target population	Private- and BIA-managed rangelands (<25 percent tree canopy cover)	BLM-managed rangelands (<25 percent tree canopy cover)	All nonforested (<10 percent tree cover) National Forest System lands
Sample design	Probabilistic	Probabilistic	Probabilistic
Scale	Broad	Broad to fine	Broad
Attributes	Foliar cover by species	Foliar cover by species	Canopy cover by species (reduced species list)
	Ground cover	Ground cover	Ground cover
	Species richness	Species richness	
	Woody plant height	Woody plant height	
	Herbaceous plant height	Herbaceous plant height	
	Plant canopy gaps	Plant canopy gaps	
	Soil aggregate stability	Soil aggregate stability	
	Production	Others locally collected	
	Sagebrush shape		
Method	Line-point intercept, species inventory, height, canopy gap intercept, soil stability kit, clipping and double sampling, sagebrush shape	Line-point intercept, species inventory, height, canopy gap intercept, soil stability kit, clipping and double sampling, sagebrush shape, others locally collected	Fixed area circular plot (1/24-acre) and canopy cover estimation of top four dominant species within a lifeform that have at least 3 percent canopy cover; line- point intercept for ground cover
Standard plot layout	47 meters (150 feet) diameter circle	30 meters (98 feet) diameter circle	
Data availability	Summary reports available from NRCS; very limited site or database data availability	Calculated indicators by site are public; database available by request	Summary reports are available from USDA Forest Service; site and data unavailable

Appendix S2. Remotely Sensed Maps of Rangeland Vegetation Available Across the Sagebrush Biome

Below we provide information about major remotely sensed maps of rangeland vegetation available across all or most of the sagebrush biome (current as of early 2019). Products specific to smaller geographies (for example, individual States) are not included.

LANDFIRE Existing Vegetation Type and Biophysical Setting Maps

Produced by U.S. Department of Agriculture, Forest Service, and U.S. Department of the Interior

Description.—LANDFIRE delivers geospatial data products for vegetation, fuel, disturbance, and fire regimes that are consistent, comprehensive, and standardized across the entire Nation.

Map product(s) available.—Many LANDFIRE products are available, but most applicable to sagebrush monitoring are existing vegetation type and biophysical setting. Other products include fuel maps, fuel models, and vegetation models.

Timeframe.—Products have been produced for multiple timeframes from 2001 to 2016.

Imagery source.—Landsat satellite imagery.

Plot data source.—The public LANDFIRE reference database (https://www.landfire.gov/lfrdb.php) includes plots from several national vegetation monitoring programs.

Web viewer.—Products available on the LANDFIRE webpage through the Data Distribution Site (https://www.landfire.gov/viewer/).

Data download.—The data access page (https://www.landfire.gov/getdata.php) allows download through the web viewer or ArcMAP tool for an area of interest, download of data mosaics for the entire United States, or streaming of web services.

Documentation.—See LANDFIRE webpage (https://www.landfire.gov/vegetation.php).

Reference.—See list of publications (https://www.landfire.gov/lf_methods.php).

National Land Cover Dataset (NLCD) Characteristics Shrubland Products

Produced by Multi-Resolution Land Characteristics (MRLC) Consortium

Description.—The NLCD shrubland map products characterize shrubland vegetation across the western United States by quantifying the proportion of shrub, sagebrush, herbaceous, annual herbaceous, litter, and bare ground, as well as the height of shrubs and sagebrush.

Map product(s) available.—percent shrub, percent sagebrush, percent big sagebrush, percent herbaceous, percent annual herbaceous, percent bare ground, percent litter, shrub height, sagebrush height.

Timeframe.—current maps represent 2016 conditions. Updates are planned every 5 years, and back in time products are in progress.

Imagery source.—WorldView-2 and Landsat 8 imagery.

Plot data source.—High resolution training data and other sources.

Web viewer.—The MRLC Interactive Viewer (https://www.mrlc.gov/viewer/) allows viewing and download of NLCD data layers.

Data download.—Data are downloadable by ecoregion (https://www.mrlc.gov/data?f%5B0%5D=category%3Ashrubland).

Documentation.—Documentation is provided on the NLCD website (https://www.mrlc.gov/data/type/rangeland-basemap) and product metadata.

Rangeland Analysis Platform

Produced by University of Montana and released in 2018 (https://rangelands.app)

Description.—This product provides continuous cover maps of major rangeland vegetation functional groups at yearly intervals from 1984 to 2017 across the western United States. The mapping process merges machine learning and cloud-based computing with remote sensing and field data to provide continuous vegetation cover maps.

Map product(s) available.—Annual forbs and grasses, Perennial forbs and grasses, Shrubs, Trees, Bare ground.

Timeframe.—Yearly maps for all years from 1984 to 2017. Maps will be updated annually in the future.

Imagery source.—Landsat satellite imagery.

Plot data source.—NRCS NRI plots, BLM AIM plots and Landscape Monitoring Framework (LMF) plots.

Web viewer.—A public web viewer (https://rangelands.app/) allows users to view data layers in an interactive map and generate graphs of average values for each year across a user-defined area of interest.

Data download.—Data download can be requested by the authors, or data can be viewed in ArcGIS as a web map tile service.

Documentation.—Documentation is provided on the web viewer and the reference below.

Near-Real-Time Herbaceous Annual Cover in the Great Basin

Produced by U.S. Geological Survey and released in 2018

Description.—Maps provide near-real-time spatial estimates of herbaceous annual vegetation percent cover across the Great Basin at multiple time points each year (May and June/July). Maps are based on Normalized Difference Vegetation Index (NDVI), which provides an estimate of vegetation greenness. Maps are produced each year by late May to help inform fire suppression activities and other management activities, such as application of weed suppressive bacteria, targeted grazing, and other cheatgrass control measures.

Map product(s) available.—Herbaceous annual cover.

Timeframe.—Multiple timeframes from 2017 to 2018. Maps are produced for multiple months within each spring. Updates are planned in early and late spring each year.

Imagery source.—Enhanced Moderate Resolution Imaging Spectroradiometer (eMODIS) imagery.

Plot data source.—High–resolution training data and other sources.

Web viewer.—None.

Data download.—Data download available from Sciencebase (https://www.sciencebase.gov/catalog/item/5b439bf9e4b060350a127028).

Documentation.—Documentation in the publication and ScienceBase.

Tree Canopy Cover

Produced by Colorado State University and released in 2017

Description.—High resolution maps of tree canopy cover (1-m resolution) were produced from Natural Agricultural Imagery Program (NAIP) imagery by using spatial wavelet analysis.

Map product(s) available.—Tree canopy cover.

Timeframe.—Single timeframe representing 2012–2013.

Imagery source.—National Agriculture Imagery Program (NAIP) air photos.

Plot data source.—None.

Web viewer.—Map is viewable in an interactive map through the Sage Grouse Initiative Data Viewer (https://map.sagegrouseinitiative.com).

Data download.—Data downloadable by State from the data viewer.

Documentation.—Documentation provided on the data download page and in the publication.

Chapter T. Communication and Public Engagement

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Executive Summary

The natural resource management paradigm has evolved, and so has recognition that communication, outreach, and engagement are crucial components of successful conservation strategies. Effective, strategic communication can tap into popular culture and public discourse to create and enhance grassroots conservation movements, identify new generations of conservationists and communicators who care about the sagebrush (*Artemisia* spp.) ecosystem, and stimulate or sustain public participation in sagebrush conservation issues.

The art and science of communication serves as more than a mechanism for sharing stories, more than a loudspeaker for the conservation community to announce its laments and achievements. Effective communication is a form of dialog that builds mutual understanding and serves as the foundation of trusting relationships. When planned strategically, funded appropriately, and executed mindfully, communication serves as a force multiplier. It tangibly advances on-the-ground conservation objectives, creates and nurtures the intergroup and interpersonal relationships necessary for success, tells stories that motivate existing collaborators to take action, inspires new partners to join a cause, increases the American public's level of awareness and engagement, and builds public support for sustainable stewardship of the sagebrush biome.

Integrating strategic communication, outreach, and engagement efforts into sagebrush conservation programs is essential to achieving success. The sagebrush biome is a vast geographic region with many stakeholders, values, land uses, and ecological threats. It is not easily accessible to most Americans and has held a low profile when compared to forests, wild and scenic rivers, and beaches. While scientific research on sensitive species within the sagebrush biome (most notably greater sage-grouse [*Centrocercus urophasianus*]) has proliferated over recent years, support for communication research and implementation remains a challenge. With over 50 percent of the sagebrush ecosystem managed by Federal and State agencies, public support is necessary to ensure a sustainable future for this ecosystem. Effective communication is essential to achieving this goal.

Introduction

The sagebrush (*Artemisia* spp.) biome has a branding problem—the public does not understand the diverse values and ecosystem services that the sagebrush ecosystem provides to American wildlife, western communities, and the Nation at large (Strickland and others, 2016). However, this challenge is a symptom of a larger truth: people have never been more disconnected from the landscapes that provide our fuel, food, and fiber (Cumming and others, 2014; Seto and others, 2014). People are not likely to conserve what they do not understand and value (Hunn, 2014). Thus, increasing the public's perception of the value of sagebrush to humans and wildlife is ultimately a communications challenge. In this chapter, we review why communication is essential to sagebrush conservation, the current communication capacity within the sagebrush community, and key gaps in sagebrush brand identity that are hampering public perception of the importance and need for sagebrush conservation.

Why Communication is Essential to Sagebrush Conservation Success

All of the sagebrush conservation needs outlined in this strategy, "Sagebrush Conservation Strategy—Challenges to Sagebrush Conservation," have one thing in common: successfully and sustainably meeting sagebrush conservation needs requires a change in human behavior. This includes change by entities that engage in sagebrush ecosystem management efforts (for example, those contributing to this strategy), those deriving their income from sagebrush landscapes, extractive industries, outdoor recreationalists, as well as various sectors of the broader American public. Change, of the type and extent needed, is not likely to occur without an effective communication effort that conveys the need, nature, costs, benefits, and tradeoffs associated with that change. In order to affect behavioral change, our communication efforts must not only be strategic and

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[3]Intermountain West Joint Venture.

[4]Audubon Rockies.

[5]U.S. Geological Survey.

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[7]U.S. Department of the Interior, Bureau of Land Management.

[8]Nevada Department of Wildlife.

[9]Montana State University.

measurable, they must be tailored to the various value systems of our target audience groups (see app. T1).

Communication, outreach, and engagement efforts provide us an opportunity to shine a light on the rich culture, emotions, and values connected with sagebrush-associated wildlife, places, and people. Thus, to be successful in achieving sustainable conservation results and building broader awareness of and appreciation and support for sagebrush conservation, the sagebrush conservation community must understand, accept, respect, and reflect the cultural and economic realities of modern times.

Management of greater sage-grouse (*Centrocercus urophasianus*) highlights the conservation challenges faced by the sagebrush ecosystem (chap. D, this volume). Recent efforts to conserve this species required an unprecedented level of collaboration, compromise, and endurance from natural resource managers and stakeholders across the West. Success (as reflected by a not-warranted listing decision; U.S. Department of the Interior, 2015c) in this hard-earned effort was due in large part to improved lines of communication among disparate stakeholders. This consideration of communication paved the way for shared solutions that prioritized the interests of the many, not the few. The resulting partnerships cemented what has evolved into an ecosystem-wide conservation effort as a cornerstone for an American conservation model that is continually evolving to meet 21st century challenges.

Long-term conservation and restoration of the sagebrush ecosystem will require sustained, concerted, and well-coordinated communication efforts across a broad spectrum of stakeholders, each with different goals and perspectives. To be effective, the growing suite of communication tools, tactics, and strategies must be used by diverse partners to amplify our collective conservation impact. Strategies must be designed with more attention paid to the perspectives of target audiences so that messages truly resonate with members of the sagebrush community, network, and eventually, a broader cross-section of the American public (see app. T1). Sagebrush country stories, and the means through which they are told, must capture and hold the imagination of the American people. Ultimately, our communications must convey a sense of shared heritage and a desire for stewardship because public support is critical to ensuring a sustainable future for this ecosystem.

Current Capacity for Communication

There are a handful of sagebrush conservation initiatives within agencies and organizations that leverage communication capacity as a means for advancing sagebrush ecosystem conservation, some but not all of these are described in table T1. One of the challenges currently facing communication professionals is a lack of capacity—no communicator focuses exclusively on sagebrush conservation communication, engagement, and outreach. Instead, collaborative sagebrush projects are often an additional duty that communicators join voluntarily. They juggle this with potentially competing priorities and projects within their respective roles. Additional support is needed from agencies and nongovernmental organizations to (1) include communication, outreach, and engagement as an essential component of all conservation strategies and (2) empower and support communicators to work on sagebrush conservation. For example, in 2016, leaders from various entities responsible for managing components (habitat, wildlife, and more) of the sagebrush ecosystem gathered and subsequently committed to improving internal and external communications around sagebrush management and conservation. They created the SageWest Communications Network (see table T1 for network description and link to website). The continued growth in number and diversity of participants reflects the value the group provides. Maintaining support from the leadership of participating entities will be necessary for collaborative communication efforts to continue and thus, fully advance conservation across the sagebrush biome.

The skill sets and approaches used by communication professionals are often distinct from but complementary to those of researchers, biologists, and land managers. Indeed, funding is a universal problem across the sagebrush conservation community. Capacity constraints, as introduced above, include a lack of stable, adequate funding necessary to support communication priorities. For example, grant funders tend to place communication in direct competition with on-the-ground conservation actions, rather than treating it as an integral component of the broader conservation strategy's success. Long-term success in landscape-scale conservation will require that robust, holistic, and durable communication strategies be incorporated as a central aspect of every step in the planning, funding, implementation, and analysis phases of sagebrush conservation and management actions.

Sagebrush partnership organizations use a variety of communication technologies such as email (including listservs and newsletters), telephone, virtual meetings, websites, social media, and cloud-based file management tools like Google Drive to build and strengthen collaboration.

Online surveys are periodically distributed to assess communication needs and advance individual or group efforts. To connect with other stakeholders and the public, most organizations have at least one social media account (for example, Facebook, Twitter, Instagram); a blog, magazine, or other storytelling medium; and printed flyers, fact sheets, or other handouts. Some entities also have video production or graphic design capacity in-house.

Current technologies will not meet the sagebrush community's future needs for increased information sharing, communication, and collaboration. The sagebrush conservation community will need access to a range of complex communication technologies in order to build a communication infrastructure capable of supporting internal collaboration. Such infrastructure is needed at the scale we wish to operate and must be sustained for the duration of time

Table T1. Sagebrush (*Artemisia* spp.) conservation organizations with communications capacity.

Conservation organization	Description
SageWest Communications Network (https://www.partnersinthesage.com/sagewest)	An assemblage of diverse stakeholders voluntarily working together to advance communication efforts in support of collaborative conservation in the sagebrush ecosystem. SageWest was founded in 2016 and chartered in March 2018.
National Audubon Society's Sagebrush Ecosystem Initiative (https://rockies.audubon.org/sagebrush)	Focuses select staff on engaging Audubon chapters, members, and local partners to find pragmatic solutions that balance the needs of people and birds. Efforts are oriented around communications related to education, science, and policy.
Natural Resources Conservation Service's Working Lands for Wildlife Program (https://www.nrcs.usda.gov/wps/portal/nrcs/detailfull/national/programs/?cid=stelprdb1046975)	Includes the Sage Grouse Initiative, which provides communications capacity to deliver information about a range of resources designed to further collaborative work around sage-grouse and sagebrush rangelands. This includes extensive communications to disseminate the existing and emerging science guiding land management practices.
Partnering to Conserve Sagebrush Rangelands Initiative (https://iwjv.org/partnering-to-conserve-sagebrush/)	A collaboration between the Intermountain West Joint Venture and U.S. Department of the Interior (DOI), Bureau of Land Management (BLM). This partnership facilitates sagebrush habitat conservation across land ownership boundaries through science, conservation implementation capacity, and communications for the benefit of wildlife, working lands, and communities.
U.S. Geological Survey Sagebrush Ecosystem Program (https://www.usgs.gov/ecosystems/sage-grouse-sagebrush-ecosystem)	Provides annual reports on the latest sagebrush ecosystem and sage-grouse science and tools, advanced notification of upcoming science and tools to partners and the DOI and highlights new science and partnerships via the SageWest network, newsletters, websites, and social media.
U.S. Fish and Wildlife Service (https://www.fws.gov/)	Employs three communications professionals with a sagebrush focus to develop key messages for internal and external use. They publish sagebrush imagery and news on social media platforms; write and execute communication strategies within the agency and beyond; generate stories for service-owned communication platforms; participate in national storytelling campaigns by contributing sagebrush related content; and pitch story ideas to regional reporters and publications.
Various State-sponsored conservation and management efforts	Various State-sponsored conservation and management efforts within the sagebrush ecosystem are supported by State communication professionals including websites, news releases, newsletters, videos, social media platforms, and more. Examples include the Utah Watershed Restoration Initiative (https://wri.utah.gov/wri/), the SageCon Conservation Partnership in Oregon (https://orsolutions.org/osproject/sagecon), and university extension efforts in Wyoming and Utah.

required to achieve the conservation objectives outlined in this strategy. For example, the effectiveness of the sagebrush communications community would be enhanced by access to a suite of technologies, such as customer relationship management, or CRM software, as well as by access to the research capacity of, and direct coordination with, social science experts. The latter is critical in order to accurately determine whether the messages we develop are not only reaching the intended audiences but also generating a change in opinion or behavior. Commitment of resources to this type of analysis will also help to build our collective body of knowledge that can be applied to future communication strategies targeting the same audience groups.

Paid media and place-based marketing has shown promise in its ability to expand reach, raise public awareness, and activate public engagement on an issue. However, most sagebrush conservation communication currently operates within the realm of "earned media," meaning coverage is "earned" through effective media pitching, relationships with newspapers and other outlets, social media, and compelling storytelling. Relying strictly on earned media is akin to relying strictly on one's own personal vegetable garden to sustain a family through the year, but to get what is needed, most citizens must supplement what they raise with what they can buy. Similarly, to expand our outreach and activate public engagement, we need to supplement what we earn with strategic, paid media campaigns.

Sagebrush conservation communications can also benefit from tapping into point-of-sale marketing techniques. These strategies leverage what we know about stakeholder lifestyles, behavior patterns, and values to deliver messages that literally meet them where they live. An example of point-of-sale marketing is when a milk company pays to design and place signs promoting its products near the milk aisle in a chain of grocery stores nationwide. These signs are intended to influence the decision-making process of customers who are not only present in the milk aisle but must make a purchase decision at that location. Analogous approaches exist within sagebrush communication, but this tactic has not been utilized as effectively as it could be and doing so would require additional behavioral research, media planning, graphic design, and financial resources.

Point-of-sale paid media tactics have been used by Texas Parks and Wildlife and its partners to combat invasive species. Their “Clean, Drain, and Dry” campaign (https://www.texasinvasives.org/cleandraindry/) won first prize in the Communications Campaign category for the Association for Conservation Information in 2014 (Hammonds, 2014). Realizing they needed public participation to help stop the spread of invasive zebra mussels (*Dreissena polymorpha*) among lakes in Texas, the partners used paid media (for example, billboards, signs, and brochures) near lakes and other water bodies asking citizens to rinse off their boats and gear.

Brand Identity

The development of a brand identity typically follows the creation of a company, a product, or a result that the brand intends to achieve. This intent is clearly articulated on a company website with its tagline, its logo, and its marketing tactics. Taglines such as “Keep Tahoe Blue,” “Only you can prevent forest fires,” and “Got milk?” not only bring to mind the colors, typefaces, icons, and images of their respective campaigns; the combination of these elements also reminds us of the action we were asked to take.

In sagebrush conservation, there is neither a brand strategy nor a unified brand identity that represents a shared interest in preserving the ecosystem. Sagebrush management and conservation efforts are pursued by diverse stakeholders across a vast geographical area encompassing over a dozen States. The size of this landscape adds a layer of complexity that is exacerbated by the generally rural nature of its communities. Reaching these communities will require different communication strategies and tools than those that would best reach individuals living in the Nation’s most dense population centers.

The sagebrush community operates as an unofficial conglomeration of agencies, organizations, industries, and individuals with overlapping and often competing interests. Stakeholders have their own unique agency or organizational brand. While this offers the sagebrush community unique opportunities, messaging and associated products may feature disparate messages, as well as various logos and designs. As such, developing a collaborative, unified communication effort to conserve sagebrush that still acknowledges the individual needs of each entity has proven challenging.

Public Perception

Fortunately, we can now lean on published research, and encourage the development of new research, to inform the development and design of a sagebrush ecosystem brand that can be shared by the various entities collaborating within the biome. There is a growing body of surveys, analyses, and more designed to assess the American public’s perceptions of various conservation issues. Surveys have consistently shown that the average American supports programs that protect fish and wildlife resources, water resources, aquatic habitat, information and education projects, and habitat protection (Duda and others, 1998). Annual “Conservation in the West” surveys also show that conservation has bipartisan support and is a popular issue among Americans nationwide (Weigel and Metz, 2018a). Specifically, 93 percent of westerners view the outdoor recreation economy as important for the economic future of their State, and voters are more likely to identify as a conservationist in 2019 than in 2017 (Weigel and Metz, 2018a).

Based on data collected by the Bureau of Land Management (BLM), a peer-reviewed study prepared by ECONorthwest found that approximately 13.8 million recreational visits to BLM-managed sagebrush lands occurred in 2013. That means 22 percent of 61.7 million recreational visits to all BLM lands that year occurred within the sagebrush biome (Lee and others, 2014). Furthermore, the study found that visits to BLM-managed lands in the 11 western States resulted in an economic output of over $1 billion in 2013 alone. In the West, nearly 9 in 10 people visited public lands annually, and 1 in 5 visited more than 20 times (Weigel and Metz, 2018a).

Further data indicates residents of the region have even deeper ties to sagebrush. A 2015 report (Western Values Project, 2015) examined economic drivers within five priority sagebrush landscapes. As reported, ECONorthwest analyzed data provided by the Bureau of Economic Analysis and two Census Bureau datasets (County Business Patterns and the 2012 Economic Census). They found that, on average

- One out of 5 jobs was related to outdoor recreation and tourism,
- One in 10 jobs was based on farming and ranching, and
- Tourism and recreation across the sagebrush landscape in these five locations supported over 23,000 jobs, or 20 percent of all jobs in these regions.

These studies suggest that the problems facing sagebrush conservationists do not stem from a lack of interest in public land recreation or conservation. Rather, most visitors to public lands within the sagebrush biome likely do not know they are in sagebrush “country” (potential branding opportunity) and have little reason to distinguish its importance compared to other natural places. This represents a lost opportunity.

Regardless of the political landscape, public opinion on conservation issues is favorable. Surveys about the extent of recreational uses of sagebrush landscapes and associated economic impacts can be used in, and to inform, media campaigns and other communications efforts aimed at increasing public perception of the value of sagebrush country. Survey information from economic research, along with messaging polls like “Conservation in the West” and “The Language of Conservation” (Weigel and Metz, 2018b), should also be applied to the development of effective sagebrush communication strategies and campaigns.

Stakeholder Engagement

Many of the actions needed to ensure successful conservation of the sagebrush ecosystem require the support of or active participation from a diversity of stakeholders. For example, to fight the spread of invasive annual grasses, the conservation community will need to engage a broad range of stakeholders when they are entering, exiting, or working within the landscape. These stakeholders range from cattle ranchers to casual hikers, cyclists to avid rock climbers, dog walkers to extractive industries, and more. Each individual is part of one or several of these special interest communities whose values, habits, and media consumption preferences vary widely, but in this example, it is the behavior that we seek to change, and therefore, the behavior is likely the best way to identify the audience. As detailed in appendix T1, effectively reaching and influencing behaviors requires targeted, varied approaches to messaging, media placement, and analysis. Certainly, social and digital platforms offer a variety of helpful analytical tools to monitor online conversations. However, the information we can learn from digital mediums alone is limited in scope and application. In addition, many of our target audience groups are not likely to be reached via digital platforms alone.

Furthermore, research on how information is shared on social media suggests that an individual's relationships have greater influence upon their decisions than a news source (Rosenstiel and others, 2017). Indeed, people are more likely to share and trust information that was shared by a friend or family member, regardless of the perceived credibility of the original source (Horrigan, 2017). Understanding this word-of-mouth behavior enables sagebrush communicators to account for and ideally integrate partnership with trusted messengers in analog and online settings.

Collaboration with Communicators

Integrating Communicators Throughout Project Lifespans

Communications, outreach, and engagement efforts focused on sagebrush conservation must be more than public relations campaigns touting successes. They should be designed to penetrate modern culture, evoke emotion, and inspire the public to become active participants in an aspect of sagebrush conservation. Communication, outreach, and engagement tactics must be treated as strategic conservation tools that tangibly advance outcomes on the ground.

Including communicators in discussions during the planning and development stages of conservation projects is critical for issues that (1) are perceived as controversial, (2) are located in areas where trust between local residents and public land managers is low, and (3) require participation from private citizens to be successful. Without this coordination, opportunities to enhance conservation outcomes will be missed. In addition, research indicates that those on opposite sides of a conservation issue may politicize science by interpreting it solely in support of their viewpoint (Sarewitz, 2004; Bowman, 2010). Thus, we recommend careful coordination with communicators throughout the lifespan of projects in order to anticipate and effectively address attempts to distort science for political means (Sarewitz, 2004; Lövbrand and Öberg, 2005). Communicators can help managers to identify, at all project stages, ways to motivate stakeholders to take specific actions on the landscape.

Need for Enhanced Communication Literacy

A lack of a shared understanding of modern communication tools, tactics, and strategies can result in miscommunication between the natural resource management and natural resource communication disciplines. This in turn reduces efficacy at all project stages and results in missed opportunities to advance or amplify conservation through communication. Managers may ask for different outcomes than what communicators can realistically provide (Lackey, 2007; Coreau and others, 2018). Managers, biologists, and others in conservation need to work with communicators to break down barriers and improve understanding about inherent constraints and opportunities in various communication approaches (Sarewitz, 2004), the art and science of communication, and the role that effective communication plays in conservation efforts.

Need for Increased Coordination with Social Science Field

The application of social science and human dimensions research is increasing in the conservation field. This growing body of work highlights the need to (1) understand how results from human dimensions/social science studies can be applied to the development of conservation communications strategies, (2) how new research can be designed to better understand our stakeholders, and (3) how new research can be designed to aid communicators in evaluating whether strategic communication goals are achieved. Better coordination is needed between the research and outreach disciplines.

Leveraging the Power of Images

Humans have used images to convey representative meaning for at least 20,000 years (Darian, 2001), and imagery continues to be a powerful communications tool. In comparison to written language, imagery provides a shortcut that allows a recipient's brain to reduce the amount of effort required to understand a message. This efficiency may be why only 10 percent of the information an individual hears is remembered after three days, while 65 percent of information observed is remembered over the same time period (Medina, 2014).

Compelling imagery must be a chief consideration when designing communication products. On social media, imagery is prevalent. Users engage with and share visual content at significantly higher rates; in one market-based study, Twitter posts with images received 150 percent more retweets and 89 percent more favorites than posts without images (Cooper, 2016). Similarly, another market study found that web-based articles (for example, blog post, web pages) with an image once every 75–100 words received double the shares compared to articles with fewer images. In the same study, the image-heavy articles received a minimum of 30 more shares than articles with higher image-to-word ratios (Pinantoan, 2015).

Beyond photographs and artistic renderings, sagebrush conservation partners face a barrier in their ability to produce, share, and distribute information-rich images. While some organizations may possess advanced data analytics expertise, fewer have personnel who can create public-friendly visualizations. Fewer still have a structure in place that provides communicators access to these resources for audience-focused communications efforts. As mentioned earlier in this chapter, research into effective messaging strategies is a gap in our current communications toolkit. Such research must also investigate which visual messages are most effective and which may have unintended, alternate interpretations.

Acknowledgments

Additional contributions were provided by San Stiver, Western Association of Fish and Wildlife Agencies and Hannah Hamilton, U.S. Geological Survey. Pat Deibert, U.S. Fish and Wildlife Service and Tom Remington, Western Association of Fish and Wildlife Agenies served as advisors.

Appendix T1. Communication Models

Values-Based Communication

Natural resource communication efforts are often passive dissemination of information. A commonly held incorrect assumption is that if the receiver of the information is provided with a greater quantity and higher quality of information, their opinions and behaviors will change based on rational, logical conclusions. However, considerable research indicates that "passive dissemination of information" (Haines and Donald, 2002) is ineffective, and neuroscientists have shown that it is emotion that sits at the core of human understanding and decision-making processes (Evans and Cruse, 2004; Lerner and others, 2015; Nelson and others, 2016).

A variation of the following model (fig. T1.1) is used by the U.S. Fish and Wildlife Service (FWS) to illustrate the connection between an individual's behavior and their underlying values and beliefs, which are rooted in emotion. An understanding of this model and an application of values-based communications to inform the use of mediums, messages, and receivers, will enhance the sagebrush (*Artemisia* spp.) conservation community's efforts to achieve communication outcomes.

1. Values are long-lasting, deeply rooted beliefs that govern how a person makes decisions and structures their lifestyle. They are the closest to the heart and are the least likely to change over time (Vaske and Donnelly, 1999; Manfredo and others, 2015). For example, an individual may identify the environment as one of their core values.

2. Beliefs are ideas that a person holds true. They emerge from values and may only change gradually over time. A person who holds the environment as a core value may hold a belief that efforts that protect clean air and water are ways to uphold that value.

3. Attitudes are the emotional relationships or mental dispositions that an individual associates with a particular behavior. Empirical research has documented that attitudes can predict behavioral intentions, which may correlate with actual behavior 53–62 percent of the time (Vaske and Donnelly, 1999). For example, the same individual may express a generally negative disposition toward oil and gas drilling activities because they believe these to be universally harmful to clean air, water, and by extension, the environment.

4. Behaviors are the actions an individual takes in physical space. This includes the patterns of their day that constitute their habits, the items they purchase, the way they manage their property, or the candidates for whom they vote. The same individual may participate in river cleanup activities in their community because that is how they embody their environmentally based value system. Behaviors are the most visible manifestation of a person's value system and are the easiest component to influence or change through strategic communications and measures.

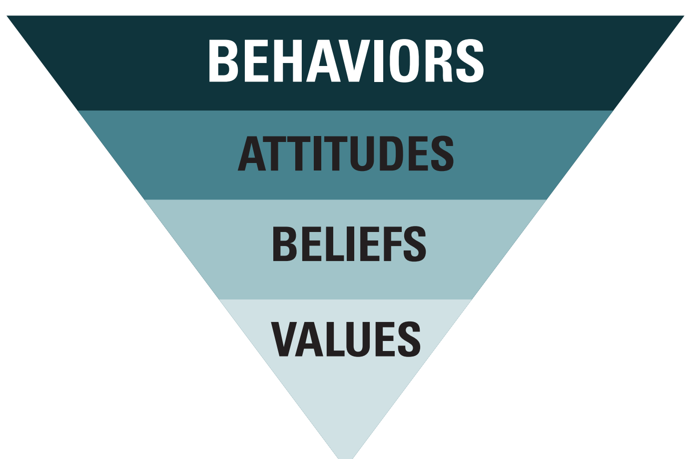

Figure T1.1. Cognitive hierarchy model depicting the connection between an individual's behavior and their underlying attitudes, values, and beliefs (modified from Fulton and others, 1996, and illustrated by Jennifer Strickland).

Segmenting Target Audiences by Their Closeness to an Issue

This model has been adapted (from Kapin and Ward, 2013) for sagebrush communications. The model segments audiences into four groups based on how close an individual is to the sagebrush conservation cause (fig. T1.2). These groups are the crowd, the network, the sagebrush community, and the core.

The core consists of individuals who demonstrate constant presence in the sagebrush conservation community and a consistent focus on achieving shared goals. As time passes, these individuals remain a grounding force in these efforts and are largely responsible for its sustained momentum. This group includes individuals representing agency and other stakeholder partners who convene conservation discussions, fund and facilitate conservation actions and research, and promote conservation of the ecosystem.

The sagebrush community can be defined as those who are already actively engaged in affecting the conservation of sagebrush country. This audience is a type of internal audience, individuals are already aware of and engaged in some aspect of sagebrush conservation. This group includes decision-makers, landowners, resource managers, scientists, storytellers, and others who not only have a seat at the table but keep that seat warm.

The network includes individuals and organizations who are not focused on sagebrush conservation specifically, but share a common value, interest, or cause. This group includes county commissioners, local business owners, professors, and school teachers in sagebrush country, those who value open spaces and public access but have never been to the sagebrush, and public health officials who know that clean air and water are integral to human health and make policy recommendations or decisions to preserve these resources. This group also includes conservation organizations with a broader focus than sagebrush with a shared interest in biodiversity, researchers conducting research in sagebrush ecosystems, and media outlets located in sagebrush country who provide information to people in the core, community, and network. In many cases, colleagues in the same organization as members of the community are actually part of the network. Most importantly, someone in the community knows or can connect with someone in the network.

The crowd is everyone else: extractive industries who may impact sagebrush ecosystems, the mainstream media, influencers of popular culture, east coast residents, national media outlets operating outside the sagebrush ecosystem, and more. Members of the crowd do not wake up thinking about the threats to sagebrush country; in fact, they may have never heard of the place. Alternatively, they may be motivated by factors which place them at odds with sagebrush conservation efforts. They are most distant from the core and thus the most difficult to reach.

These four groups can be used to establish and maintain a shared vocabulary between communicators and natural resource managers, resulting in clearer, more effective communication and broader conservation strategies.

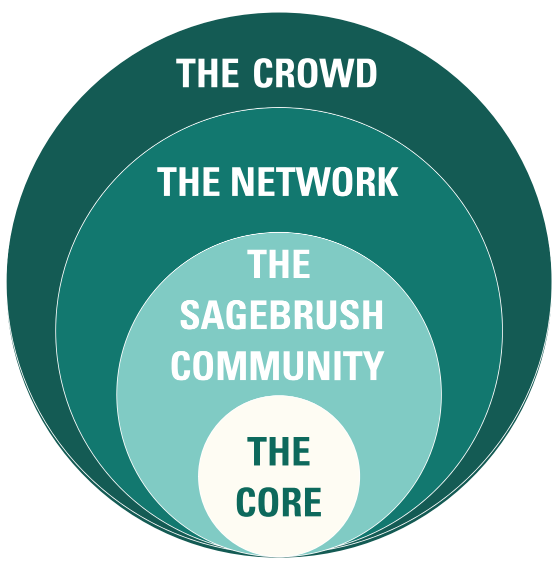

Figure T1.2. Four groups illustrating relationships of individuals to sagebrush (*Artemisia* spp.) conservation (modified from Kapin and Ward, 2013; illustrated by Jennifer Strickland).

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Remington and others—**Sagebrush Conservation Strategy—Challenges to Sagebrush Conservation**—Open-File Report 2020–1125

ISSN 0196-1497 (print)
ISSN 2331-1258 (online)
https://doi.org/10.3133/ofr20201125