

CHIHUAHUAN DESERT RAPID ECOREGIONAL ASSESSMENT

PRE-ASSESSMENT REPORT



U.S. Department of the Interior
Bureau of Land Management
Rapid Ecoregional Assessments

October, 2017



It is the mission of the Bureau of Land Management to sustain the health, diversity, and productivity of the public lands for the use and enjoyment of present and future generations.

SUBMITTED TO

David Herrell, Contracting Officers Representative
(COR) Department of the Interior
Bureau of Land Management
Carlsbad Field Office
620 East Greene Street
Carlsbad, NM 88220-6292

SUBMITTED BY



Sound Science LLC
PO Box 9721
Boise, ID 83707



U.S. Department of Agriculture
Agricultural Research Service
Jornada Experimental Range
MSC 3JER,
NMSU, Box 30003
Las Cruces, NM 88003-8003

**CHIHUAHUAN DESERT
RAPID ECOREGIONAL ASSESSMENT
Final Report
U.S. Department of the Interior
Bureau of Land Management
Rapid Ecoregional Assessments
October 2017**

Authorship

Cite as: Unnasch, R., D. Braun, and K. Young. 2017. Chihuahuan Desert Rapid Ecoregional Assessment Pre-Assessment Report. With contributions by M. Batchner, F. Fogarty, J. Marty, C. Salo, V. Seamster, N. Welch, and T. Whittier. Sound Science technical report to the U.S. Department of the Interior Bureau of Land Management, Rapid Ecoregional Assessment Program.

Contents

Authorship	3
1 Executive Summary.....	1
1.1 Purpose and Structure of Rapid Ecoregional Assessments	1
1.2 Chihuahuan Desert REA Geographic Extent	2
1.3 Overview of the Chihuahuan Desert Ecoregion.....	2
1.4 Chihuahuan Desert REA Conservation Elements and Change Agents.....	4
1.5 Chihuahuan Desert REA Management Questions	5
1.6 Chihuahuan Desert REA Conceptual Ecological Models	7
1.7 Chihuahuan Desert REA Pre-Assessment Report Structure	8
2 Introduction to the Chihuahuan Desert Rapid Ecoregional Assessment.....	9
2.1 Purpose and Structure of Rapid Ecoregional Assessments	9
2.2 Purpose and Structure of the Pre-Assessment Report	10
2.3 Chihuahuan Desert Ecoregion and Rapid Ecoregional Assessment Boundaries	10
2.4 Chihuahuan Desert Biophysical Setting	13
2.4.1 Climate	14
2.4.2 Geology	18
2.4.3 Soils	20
2.4.4 Hydrology	21
2.4.5 Wildfire.....	22
2.4.6 Large-Scale Biogeography	24
2.5 Chihuahuan Desert Biodiversity.....	25
2.5.1 Species Richness.....	25
2.5.2 Endemism.....	25
2.5.3 Characteristic and Keystone Species	26
2.5.4 Ecological Systems	28
2.6 Chihuahuan Desert Human Landscape	32

2.6.1	Demography.....	32
2.6.2	Land Ownership	33
2.6.3	Land Use.....	33
2.6.4	Water Use	34
3	Conservation Elements, Change Agents, and Management Questions	37
3.1	Conservation Elements	37
3.1.1	Dry System Types	38
3.1.2	Wet System Types.....	39
3.1.3	Species and Species Assemblages.....	43
3.2	Change Agents	47
3.2.1	Climate Change	48
3.2.2	Uncharacteristic Wildfire	49
3.2.3	Invasive Species.....	49
3.2.4	Development.....	49
3.2.5	Excessive Domestic Grazing	51
3.2.6	Landscape Restoration.....	52
3.3	Management Questions	52
4	Conceptual Models and Modeling Methods	55
4.1	Conceptual Modeling Framework.....	55
4.2	Chihuahuan Desert Conceptual Model Hierarchy	56
4.3	Chihuahuan Desert Dry System Model.....	57
4.4	Chihuahuan Desert Wet System Model.....	60
4.5	Dry-System Conservation Element Conceptual Model Methodology.....	63
4.6	Wet System, Species, and Assemblage Conservation Element Conceptual Model Methodology	64
5	Chihuahuan Desert Grasslands Conceptual Model	67
5.1	Sources of Information	67
5.2	Chihuahuan Desert Grassland Control Model	67
5.2.1	Landscape Configuration, and Elevation and Soil-Geomorphic Setting	68
5.2.2	Precipitation and Temperature.....	68
5.2.3	Natural Disturbances	69
5.2.4	Vegetation.....	69

5.2.5	Wildlife and Habitat	71
5.3	Chihuahuan Desert Grasslands State-and-Transition Model.....	72
5.3.1	Climate Change	73
5.3.2	Uncharacteristic Wildfire	74
5.3.3	Invasive Species.....	77
5.3.4	Development.....	78
5.3.5	Excessive Domestic Grazing	78
5.3.6	Landscape Restoration.....	79
5.4	Chihuahuan Desert Grassland Key Ecological Attributes.....	82
6	Chihuahuan Desert Scrub Conceptual Model.....	86
6.1	Sources of Conceptual Models and Vegetation Classification.....	86
6.2	Chihuahuan Desert Scrub Control Model	86
6.2.1	Landscape Configuration, and Elevation and Soil-Geomorphic Setting	86
6.2.2	Precipitation and Temperature.....	87
6.2.3	Natural Disturbances	87
6.2.4	Vegetation.....	87
6.2.5	Wildlife and Habitat	88
6.3	Chihuahuan Desert Scrub State-and-Transition Model	89
6.3.1	Climate Change	91
6.3.2	Uncharacteristic Wildfire	92
6.3.3	Invasive Species.....	95
6.3.4	Development.....	96
6.3.5	Excessive Domestic Grazing	96
6.3.6	Landscape Restoration.....	97
6.4	Chihuahuan Desert Scrub Key Ecological Attributes	99
7	Pinyon-Juniper Woodlands Conceptual Model	100
7.1	Sources of Conceptual Models and Vegetation Classification.....	100
7.2	Pinyon-Juniper Woodlands Control Model.....	100
7.2.1	Landscape Configuration, and Elevation and Soil-Geomorphic Setting	100
7.2.2	Precipitation and Temperature.....	102
7.2.3	Natural Disturbances	102

7.2.4	Vegetation.....	103
7.2.5	Wildlife and Habitat	104
7.3	Pinyon-Juniper Woodlands State-and-Transition Model.....	104
7.3.1	Climate Change	107
7.3.2	Uncharacteristic Wildfire	108
7.3.3	Invasive Species.....	112
7.3.4	Development.....	113
7.3.5	Excessive Domestic Grazing	113
7.3.6	Landscape Restoration.....	114
7.4	Pinyon-Juniper Woodlands Key Ecological Attributes	116
8	Perennial Streams Conceptual Model	122
8.1	Montane- vs Lowland-Headwater Perennial Streams	122
8.1.1	Perennial Stream System Type Descriptions and Examples	122
8.1.2	Fishes of the U.S. Portion of the Chihuahuan Desert Ecoregion	124
8.2	Sources of Information	126
8.3	Perennial Stream Systems Control Model	127
8.4	Perennial Stream Systems Stressor Model	130
8.4.1	Critical Ecological Processes and Ecological Outcomes	135
8.4.2	Climate Change	140
8.4.3	Uncharacteristic Wildfire	144
8.4.4	Invasive Species.....	146
8.4.5	Development.....	149
8.4.6	Excessive Domestic Grazing	153
8.4.7	Landscape Restoration.....	154
8.5	Perennial Stream Systems Key Ecological Attributes.....	155
9	Large River-Floodplain Systems Conceptual Model.....	156
9.1	Sources of Information	156
9.2	Large River-Floodplain System Control Model	157
9.3	Large River-Floodplain System Stressor Model	162
9.3.1	Critical Ecological Processes and Ecological Outcomes	170
9.3.2	Climate Change	175

9.3.3	Uncharacteristic Wildfire	180
9.3.4	Invasive Species.....	183
9.3.5	Development.....	187
9.3.6	Excessive Domestic Grazing	192
9.3.7	Landscape Restoration.....	194
9.4	Large River-Floodplain System Key Ecological Attributes.....	194
10	Springs-Emergent Wetlands Conceptual Model.....	196
10.1	Sources of Information	197
10.2	Springs-Emergent Wetlands System Control Model	198
10.3	Springs-Emergent Wetlands System Stressor Model	202
10.3.1	Critical Ecological Processes and Ecological Outcomes	207
10.3.2	Climate Change	212
10.3.3	Uncharacteristic Wildfire	215
10.3.4	Invasive Species.....	217
10.3.5	Development.....	219
10.3.6	Excessive Domestic Grazing	222
10.3.7	Landscape Restoration.....	223
10.4	Springs-Emergent Wetlands System Key Ecological Attributes.....	223
11	Playas and Playa Lakes Conceptual Model	225
11.1	Sources of Information	227
11.2	Playa and Playa Lakes Control Model	227
11.3	Playa and Playa Lakes System Stressor Model	231
11.3.1	Critical Ecological Processes and Ecological Outcomes	234
11.3.2	Climate Change	240
11.3.3	Uncharacteristic Wildfire	244
11.3.4	Invasive Species.....	246
11.3.5	Development.....	250
11.3.6	Excessive Domestic Grazing	253
11.3.7	Landscape Restoration.....	254
11.4	Playa and Playa Lakes Key Ecological Attributes.....	255
12	Pronghorn Conceptual Model.....	256

12.1	Sources of Information	256
12.2	Pronghorn Overview	256
12.2.1	Distribution	256
12.2.2	Habitat.....	257
12.3	Pronghorn Stressor Model.....	259
12.3.1	Climate Change	263
12.3.2	Uncharacteristic Wildfire	267
12.3.3	Invasive Species.....	271
12.3.4	Development.....	273
12.3.5	Excessive Domestic Grazing	276
12.3.6	Landscape Restoration	279
12.4	Pronghorn Key Ecological Attributes	283
13	Mule Deer Conceptual Model.....	285
13.1	Sources of Information	285
13.2	Mule Deer Overview	285
13.2.1	Distribution	285
13.2.2	Habitat.....	286
13.3	Mule Deer Stressor Model.....	289
13.3.1	Climate Change	293
13.3.2	Uncharacteristic Wildfire	296
13.3.3	Invasive Species.....	300
13.3.4	Development.....	303
13.3.5	Excessive Domestic Grazing	306
13.3.6	Landscape Restoration.....	309
13.4	Mule Deer Key Ecological Attributes	313
14	Banner-tailed Kangaroo Rat Conceptual Model	314
14.1	Sources of Information	314
14.2	Banner-tailed Kangaroo Rat Overview.....	314
14.2.1	Distribution	314
14.2.2	Habitat.....	315
14.3	Banner-tailed Kangaroo Rat Stressor Model	316

14.3.1	Climate Change	320
14.3.2	Uncharacteristic Wildfire	324
14.3.3	Invasive Species.....	326
14.3.4	Development.....	330
14.3.5	Excessive Domestic Grazing	333
14.3.6	Landscape Restoration.....	337
14.4	Banner-tailed Kangaroo Rat Key Ecological Attributes.....	339
15	Black-tailed Prairie Dog Conceptual Model	341
15.1	Sources of Information	341
15.2	Black-tailed Prairie Dog Overview.....	341
15.2.1	Distribution	341
15.2.2	Habitat.....	342
15.3	Black-tailed Prairie Dog Stressor Model	343
15.3.1	Climate Change and Uncharacteristic Wildfire	347
15.3.2	Invasive Species and Landscape Restoration.....	349
15.3.3	Development.....	350
15.3.4	Excessive Domestic Grazing	353
15.4	Black-tailed Prairie Dog Key Ecological Attributes.....	356
16	Grassland Bird Assemblage Conceptual Model	357
16.1	Sources of Information	357
16.2	Grassland Bird Assemblage Overview.....	357
16.2.1	Distribution	357
16.2.2	Habitat.....	359
16.3	Grassland Bird Assemblage Stressor Model	361
16.3.1	Climate Change	366
16.3.2	Uncharacteristic Wildfire	369
16.3.3	Invasive Species and Landscape Restoration.....	371
16.3.4	Development.....	373
16.3.5	Excessive Domestic Grazing	377
16.4	Grassland Bird Assemblage Key Ecological Attributes.....	381
17	Grassland Small Mammal Assemblage Conceptual Model.....	382

17.1	Sources of Information	382
17.2	Grassland Small Mammal Assemblage Overview	382
17.2.1	Distribution	382
17.2.2	Habitat.....	383
17.2.2.1	Chihuahua deer mouse	383
17.2.2.2	Hispid cotton rat	383
17.2.2.3	Southern Plains woodrat.....	384
17.2.2.4	Tawny-bellied cotton rat.....	384
17.2.2.5	Yellow-nosed cotton rat.....	385
17.3	Grassland Small Mammal Assemblage Stressor Model.....	386
17.3.1	Climate Change	390
17.3.2	Uncharacteristic Wildfire	394
17.3.3	Invasive Species and Landscape Restoration.....	397
17.3.3.1	Grass Assemblage	400
17.3.3.2	Shrub Density.....	402
17.3.3.3	Restoration Management.....	403
17.3.4	Development.....	404
17.3.5	Excessive Domestic Grazing	406
17.4	Grassland Small Mammal Assemblage Key Ecological Attributes	409
18	Synthesis of Assessment Priorities.....	411
18.1	Chihuahuan Desert REA Conservation Elements.....	411
18.2	Chihuahuan Desert REA Change Agents	412
18.3	Chihuahuan Desert REA Management Questions	412
18.4	Chihuahuan Desert REA Conceptual Ecological Models.....	413
19	Literature Cited	414
20	Glossary.....	481
21	Acronyms	484

Tables

Table 1-1. Chihuahuan Desert REA Management Questions	6
Table 2-1. Terrestrial ecological systems of the U.S. portion of the ecoregion (NatureServe 2009)	29
Table 3-1. Chihuahuan Desert REA Conservation Elements	37
Table 3-2. Amphibians of the U.S. portions of the Chihuahuan desert ecoregion.....	47
Table 3-3. Chihuahuan Desert REA Management Questions.	53
Table 8-1. Fishes native to the U.S. waters of the Chihuahuan desert ecoregion, by drainage and type of water body.....	124
Table 8-2. Chihuahuan desert perennial stream systems stressor model drivers, critical environmental elements, critical ecological processes, and ecological outcomes.	130
Table 9-1. Chihuahuan desert large river-floodplain system stressor model drivers, critical environmental elements, critical ecological processes, and ecological outcomes	163
Table 10-1. Chihuahuan desert springs-emergent wetlands system stressor model drivers, critical environmental elements, critical ecological processes, and ecological outcomes.	202
Table 11-1. Playa and playa lakes stressor model components.	231
Table 12-1. Pronghorn stressor model components.	259
Table 13-1. Definitions of stressor model components for mule deer.....	289
Table 14-1. Banner-tailed kangaroo rat definitions of stressor model components.	316
Table 15-1. Black-tailed prairie dog definitions of stressor model components.....	343
Table 16-1. Grassland bird assemblage definitions of stressor model components.	361
Table 17-1. Grassland small mammal assemblage species, habitat preferences, and food preferences.....	385
Table 17-2. Grassland small mammal assemblage definitions of stressor model components.....	386

Figures

Figure 2-1. The Chihuahuan Desert ecoregion and REA analysis extent.	11
Figure 2-2. U.S. portion of the Chihuahuan Desert ecoregion, analysis extent, BLM districts, and adjacent REAs.	12
Figure 2-3. Topographic relief and major landforms.	14
Figure 2-4. Monthly average maximum (solid line) and minimum (dashed line) temperatures (°C), left axis, and precipitation (gray bars) (mm), right axis, at the Jornada Experimental Range, New Mexico, (left; WRCC 2016), and Rio Grande Village in Big Bend National Park, Texas (right; NCEI 2010).....	15
Figure 2-5. Spatial patterns of six climatic variable: Mean temperature of the a) warmest and b) coldest quarters; c) annual mean temperature; d) annual precipitation; and precipitation of the e) wettest and f) driest quarters.	16
Figure 2-6. Landcover types within the Level III boundaries. (NatureServe 2009).....	30
Figure 4-1. Chihuahuan desert dry system conceptual ecological model.	58

Figure 4-2. Chihuahuan desert dry system conceptual ecological model with change agents.....	59
Figure 4-3. Chihuahuan desert wet system conceptual ecological model.	62
Figure 4-4. Chihuahuan desert wet system conceptual ecological model with change agents.	63
Figure 5-1. Chihuahuan Desert Grasslands control model.	68
Figure 5-2. Chihuahuan Desert Grassland-Sandy Plains Landform state-and-transition model.	83
Figure 5-3. Chihuahuan Desert Grassland-Piedmont Landform state-and-transition model.	84
Figure 5-4. Chihuahuan Desert Grasslands Subsystem-Foothill and Lowland Landforms state-and-transition model.....	85
Figure 6-1. Chihuahuan Desert Scrub control model.....	87
Figure 6-2. Combined Chihuahuan Desert Scrub and Salt Desert Scrub state-and-transition model.	91
Figure 7-1. Pinyon-Juniper Woodlands control model.	101
Figure 7-2. Historic Pinyon-Juniper Savanna state-and-transition model.	117
Figure 7-3. Current Pinyon-Juniper Savanna state-and-transition model.	118
Figure 7-4. Historic Pinyon-Juniper Shrub Woodland state-and-transition model.	119
Figure 7-5. Current Pinyon-Juniper Shrub Woodland state-and-transition model.	120
Figure 7-6. Historic and Current Pinyon-Juniper Persistent Woodland state-and-transition model.	121
Figure 8-1. Chihuahuan desert perennial stream systems control model.	128
Figure 8-2. Chihuahuan desert perennial stream systems stressor model.	134
Figure 8-3. Chihuahuan desert perennial stream systems stressor model: Potential impacts of climate change.....	142
Figure 8-4. Chihuahuan desert perennial stream stressor model: Potential impacts of uncharacteristic wildfire and excessive domestic grazing.....	145
Figure 8-5. Chihuahuan desert perennial stream stressor model: Potential impacts of invasive species and landscape restoration.	147
Figure 8-6. Chihuahuan desert perennial stream stressor model: Impacts of land and water development.....	151
Figure 9-1. Chihuahuan desert large river-floodplain system control model.....	159
Figure 9-2. Chihuahuan desert large river-floodplain system stressor model.	168
Figure 9-3. Chihuahuan desert large river-floodplain systems stressor model: Potential impacts of climate change.	178
Figure 9-4. Chihuahuan desert large river-floodplain systems stressor model: Potential impacts of uncharacteristic wildfire and excessive domestic grazing.	182
Figure 9-5. Chihuahuan desert larger river-floodplain systems stressor model: Potential impacts of invasive species and landscape restoration.....	185
Figure 9-6. Chihuahuan desert large river-floodplain systems stressor model: Impacts of land and water development.	189
Figure 10-1. Chihuahuan desert springs-emergent wetlands system control model.	199
Figure 10-2. Chihuahuan desert springs-emergent wetlands system stressor model.	206
Figure 10-3. Chihuahuan desert perennial stream systems stressor model: Potential impacts of climate change.....	213
Figure 10-4. Chihuahuan desert perennial stream stressor model: Potential impacts of uncharacteristic	

wildfire and excessive domestic grazing.....	216
Figure 10-5. Chihuahuan desert perennial stream stressor model: Potential impacts of invasive species and landscape restoration.	218
Figure 10-6. Chihuahuan desert perennial stream stressor model: Impacts of land and water development.....	221
Figure 11-1. Chihuahuan desert playa and playa lakes system control model.	228
Figure 11-2. Chihuahuan desert playa and playa lakes full stressor model.	236
Figure 11-3. Chihuahuan desert playa and playa lakes stressor model: Potential impacts of climate change.....	242
Figure 11-4. Chihuahuan desert perennial stream stressor model: Potential impacts of uncharacteristic wildfire and excessive domestic grazing.....	245
Figure 11-5. Chihuahuan desert perennial stream stressor model: Potential impacts of invasive species and landscape restoration.	248
Figure 11-6. Chihuahuan desert perennial stream stressor model: Impacts of land and water development.....	251
Figure 12-1. Pronghorn stressor model.	262
Figure 12-2. Pronghorn stressor model: Potential impacts of climate change.	265
Figure 12-3. Pronghorn stress or model: Potential impacts of uncharacteristic wildfire.....	270
Figure 12-4. Pronghorn stressor model: Potential impacts of invasive species and associated management.....
Figure 12-5. Pronghorn stressor model: Potential impacts of development.	275
Figure 12-6. Pronghorn stressor model: Potential impacts of excessive domestic grazing.	278
Figure 12-7. Pronghorn stressor model: Potential impacts of landscape restoration.....	282
Figure 13-1. Mule deer stressor model.....	292
Figure 13-2. Mule deer stressor model: Potential impacts of climate change.....	295
Figure 13-3. Mule deer stressor model: Potential impacts of uncharacteristic wildfire.	299
Figure 13-4. Mule deer stressor model: Potential impacts of invasive species and associated management.....	302
Figure 13-5. Mule deer stressor model: Potential impacts of development.....	305
Figure 13-6. Mule deer stressor model: Potential impacts of excessive domestic grazing.....	307
Figure 13-7. Mule deer stressor submodel for wildlife management and landscape restoration (See Appendix 1 for link details).	312
Figure 14-1. Banner-tailed kangaroo rat stressor model.....	319
Figure 14-2. Banner-tailed kangaroo rat stressor model: Potential impacts of climate change.....	322
Figure 14-3. Banner-tailed kangaroo rat stressor model: Potential impacts of uncharacteristic wildfire.	325
Figure 14-4. Banner-tailed kangaroo rat stressor model: Potential impacts of invasive species and landscape restoration.	328
Figure 14-5. Banner-tailed kangaroo rat stressor submodel for human uses and urban & industrial growth.....	332
Figure 14-6. Banner-tailed kangaroo rat stressor submodel for legacy effects of historic grazing and	

grazing management.	336
Figure 15-1. Black-tailed prairie dog stressor model.	346
Figure 15-2. Black-tailed prairie dog stressor model: Potential impacts of climate change and uncharacteristic wildfire.	348
Figure 15-3. Black-tailed prairie dog stressor model: Potential impacts of development.	352
Figure 15-4. Black-tailed prairie dog stressor model: Potential impacts of excessive domestic grazing.	355
Figure 16-1. Grassland bird assemblage stressor model.	365
Figure 16-2. Grassland bird assemblage stressor model: Potential impacts of climate change.	368
Figure 16-3. Grassland bird assemblage stressor model: Potential impacts of uncharacteristic wildfire.	370
Figure 16-4. Grassland bird assemblage stressor model: Potential impacts of invasive species and landscape restoration.	372
Figure 16-5. Grassland bird assemblage stressor model: Potential impacts of development.	376
Figure 16-6. Grassland bird assemblage stressor model: Potential impacts of excessive domestic grazing.	379
Figure 17-1. Grassland small mammal assemblage full stressor model.	389
Figure 17-2. Grassland small mammal assemblage stressor model: Potential impacts of climate change.	391
Figure 17-3. Grassland small mammal assemblage stressor model: Potential impacts of uncharacteristic wildfire.	396
Figure 17-4. Grassland small mammal assemblage stressor model: Potential impacts of invasive species, their management, and landscape restoration.	399
Figure 17-5. Grassland small mammal assemblage stressor model: Potential impacts of development.	405
Figure 17-6. Grassland small mammal assemblage stressor model: Potential impacts of excessive domestic grazing.	408

1 Executive Summary

This report constitutes the Phase I or Pre-Assessment report for the Chihuahuan Desert Rapid Ecoregional Assessment, prepared for the U.S. Department of the Interior, Bureau of Land Management. This chapter summarizes the Pre-Assessment report.

1.1 Purpose and Structure of Rapid Ecoregional Assessments

Bureau of Land Management (BLM) Rapid Ecoregional Assessments (REAs) seek to provide information to natural resource managers concerning (a) ecoregional-scale ecological conditions and trends, (b) the major factors that shape these conditions and trends, and (c) opportunities to conserve ecological resources across management boundaries. The REA approach integrates diverse sources of information needed to support conservation, restoration, and the development of ecological management programs in a cohesive manner.

REAs provide a foundation for adaptive ecosystem management by summarizing current ecological understanding, and provide a baseline for comparisons with future data and understanding. These comparisons can examine ecological trends and the effectiveness of management practices, for example, to help ecological resource managers assess which practices are working and where practices need to be modified. REAs do not make management decisions. They provide information to help natural resource managers make good management decisions.

REAs do not address all ecological resources in an ecoregion, as this is an impossible task. Instead, they focus on a limited set of key resources, termed Conservation Elements (CEs), consisting of regionally-significant terrestrial and aquatic habitats and species of management concern. Additionally, REAs do not attempt to assess all threats to the CEs in an ecoregion, another impossible task. Instead, REAs focus on a limited set of key stressors, termed Change Agents (CAs). REAs then develop and prioritize a set of specific questions to answer concerning the CEs and CAs, termed Management Questions (MQs), which the REA seeks to address using geospatial data.

Each REA has two phases. During Phase I, the Pre-Assessment phase, the REA team identifies the Conservation Elements, Change Agents, and Management Questions on which to focus the REA. The REA team then develops conceptual models for three purposes: to identify potentially measurable key ecological attributes for each Conservation Element; to document present understanding of how each Change Agent may affect each Conservation Element; and to provide a means for translating the Management Questions into terms specific to each individual Conservation Element and/or Change Agent. The present report constitutes the Phase I report for the Chihuahuan Desert REA. Phase II, the Assessment phase, uses existing geospatial data and publications to map the distribution of the Conservation Elements and, where feasible, assess the condition of these Conservation Elements, assess the impacts of the Change Agents, assess possible future impacts of Change Agents where appropriate, and address the key Management Questions identified during Phase I.

The word “Rapid” in the term, “Rapid Ecoregional Assessment” bears emphasis. REAs collect no new data. They are built on existing data and published reports. REAs address large-scale conditions and concerns that cut across managerial boundaries, and work with conceptual models of conservation elements distributed across the entire ecoregion. REAs necessarily focus more on the kinds of broad characteristics of these large areas that can be captured in large-scale geospatial data. As a result, field offices always need to compare the results of REAs against finer-scale, local information before taking actions based on REA findings. REAs do not replace fine-scale data or the expertise of local managers. Rather, they place local concerns in an ecoregional context and provide framework for considering integrated responses that address large scale change agents and resource issues. At the same time, REAs provide crucial information on gaps in existing large-scale data or knowledge, to help ecological resource managers identify needs for future monitoring and/or research.

1.2 Chihuahuan Desert REA Geographic Extent

Each REA focuses on an individual Level-III ecoregion or a group of adjacent Level-III ecoregions (USEPA 2013). The Chihuahuan Desert Level-III ecoregion (see Chapter 2, Figure 2-1) covers portions of both the U.S. and Mexico (Wiken et al. 2011), with approximately three quarters lying within Mexico (Dinerstein et al. 2001, Monger 2006). However, the Chihuahuan Desert REA addresses only lands within the U.S. The U.S. portion of the Chihuahuan Desert ecoregion covers significant portions of western Texas and southern New Mexico approximately from the eastern margins of the Pecos River valley westward to the Arizona border, with a small extension into southeastern Arizona. The Madrean Archipelago and Southern Great Plains REAs address the landscapes immediately to the west and northeast, respectively.

Every REA addresses an area slightly larger than its Level-III ecoregion(s), termed the “analysis extent,” that includes all watersheds that overlap the Level-III boundaries. Consequently, the analysis extent for the Chihuahuan Desert REA overlaps with the analysis extents for the Madrean Archipelago and Southern Great Plains REAs.

The analysis extent for the Chihuahuan Desert REA includes parts of the BLM Albuquerque, Las Cruces, and Pecos Districts in New Mexico and part of the BLM Gila District in Arizona (see Chapter 2, Figure 2-2). The BLM does not operate district or field offices in west Texas. The analysis extent includes approximately 201,000 km² (approx. 77,500 mi²). This includes approximately 2,000 km² (approx. 750 mi²) in Arizona, 97,000 km² (approx. 37,400 mi²) in New Mexico, and approximately 102,000 km² (approx. 39,350 mi²) in Texas. The BLM manages 36,488 km² (14,088 mi²) of these lands, including 688 km² (266 mi²) in Arizona and 35,799 km² (13,822 mi²) in New Mexico.

1.3 Overview of the Chihuahuan Desert Ecoregion

The Chihuahuan desert is the largest desert in North America and the southernmost desert in the United States, stretching from the Southwest U.S. to the Central Highlands of Mexico (see Chapter 2, Figure 2-1). The two dominant ground cover types in the U.S. portion of the ecoregion consist of grasslands and scrub, the relative areas of which have fluctuated back and forth at least three times in the past 3,000 years alone (Havstad and Schlesinger 2006). The perennial grasses in the U.S. portion of the ecoregion

today, including black grama grass (*Bouteloua eriopoda*), may be relicts from wetter conditions during the mid-Holocene (Havstad and Schlesinger 2006). Since the mid-1800s, shrub-dominated systems have expanded at the expense of grassland cover (NMDGF 2006, Ruhlman et al. 2012). The relative importance of the possible causes for this latter transition, including excessive livestock grazing, climate change, and altered fire regimes, remain a matter of debate (Ruhlman et al. 2012).

The western two thirds of the ecoregion in the U.S. consists of a basin and range province of mostly north-south trending mountain ranges separated by broad desert basins (Monger et al. 2006) (See Chapter 2, Figure 2-3). Additional mountain ranges and high country occur in the region of the Big Bend of the Rio Grande, spanning the southernmost portion of the ecoregion within the U.S. The easternmost lands of the analysis extent are less mountainous, including the western margins of the Llano Estacado, a high tableland that extends across much of northwestern Texas. Much of the eastern third of the Chihuahuan Desert REA analysis extent overlies the western formations of the Permian Basin, a distinctive geologic region with unique karst and cave features around its western margins that also constitutes the most productive and heavily developed oil and gas region in North America.

Peak elevations within the Chihuahuan Desert REA analysis extent (See Chapter 2, Figure 2-3) reach to over 2,700 m above sea level in the San Andres Mountains in New Mexico, over 2,600 m in the Guadalupe Mountains straddling the Texas-New Mexico border, and over 2,500 m in the Davis Mountains in Texas. Other high mountain ranges within the U.S. portion of the ecoregion include the Oscura and Organ Mountains in New Mexico and the Sierra Diablo and Hueco, Eagle, Chinati, Del Norte, Chisos, and Glass Mountains in Texas. Several ranges with high peaks above 3,000 m straddle the boundaries of the analysis extent, including the Capitan and Sacramento Mountains between the Pecos and the central closed basins, and the Black Range and Magdalena Mountains along the west side of the Rio Grande basin in New Mexico. The lowest point in the analysis extent lies at 350 m above sea level where the Rio Grande enters Amistad Reservoir. Other low points, within closed basins, include Lordsburg Playa, 1,266 m above sea level, Playas Lake, 1,305 m, Lake Lucero, 1,188 m, and the Salt Basin near Dell City, TX, 1,102 m. This elevation range, combined with variations in topographic aspect, create a wide variety of macro- and micro-climates.

The analysis extent includes portions of three major rivers and their associated basins (see Chapter 2, Figures 2-3 and 2-4): the Gila River basin in the far west, the Rio Grande basin through the center and south, and the Pecos River basin in the east. It also includes a large portion of the closed Guzmán (aka Mimbres River) basin in the far southwest; a group of closed basins roughly in the center of the analysis extent, consisting of the Jornada del Muerto, Jornada Draw, Tularosa, and Salt basins; and a small portion of the Devil's River basin in the far southeast. The Gila River, Rio Grande, and Pecos River all originate outside the boundaries of the analysis extent; the Gila River and Rio Grande also continue flowing past the boundaries of the analysis extent. Flows along the Pecos River and Rio Grande are highly altered through the operations of numerous dams and diversions, both within and upstream from the ecoregion. The analysis extent also contains numerous perennial streams and rivers that originate and terminate entirely within its boundaries, springs, cenotes, seeps, playa lakes, and reservoirs, all with

associated wetlands. Intermittently wetted runoff channels and playas also contribute to the diversity of wetted habitats in the ecoregion.

The U.S. portion of the Chihuahuan desert ecoregion is hot and dry, and experiences a wide range of variation in temperature across seasons and elevation (see Chapter 2, Figures 2-5 and 2-6). Maximum summer temperatures in the U.S. portion of the ecoregion, at an elevation of 1,300 meters, average 34°C (93°F) and minimum temperatures in the coldest months average -5°C (23°F) (Wainwright 2006). Precipitation is low and highly variable (see Chapter 2, Figures 2-5 and 2-6). The northern portion of the ecoregion, at an elevation of 1,300 meters, receives an average of 245 mm (9.65 in) yr⁻¹ of precipitation, accompanied by an average 220 cm (86.7 in) yr⁻¹ of potential evaporation (Wainwright 2006). About half of the precipitation arrives in the form of convective storms during the late summer monsoon, supplied by moisture circulated from over the Gulf of Mexico. The remainder arrives in winter storms carrying moisture from over the Pacific Ocean. May and June are typically the driest months (Havstad and Schlesinger 2006). El Niño years typically bring 1.5 times the average winter (October to May) precipitation to the northern portion of the extent, while La Niña years typically bring half the average winter precipitation (Wainwright 2006). Neither El Niño nor La Niña conditions significantly affect summer moisture.

The U.S. portion of the Chihuahuan Desert ecoregion is mostly sparsely populated (see Chapter 2, Figure 2-7) with an economy based on ranching, irrigated farming, manufacturing, mining, oil and gas production, and military testing and training (Anderson and Gerber 2008). The Borderplex Region of Las Cruces, New Mexico, El Paso, Texas, and Ciudad Juarez, Chihuahua, Mexico, is the seventh largest manufacturing area in North America (MVEDA undated). Its combined population of approximately 2.5 million people, nearly 1.2 million of whom live in the U.S., shares use of the Rio Grande and local aquifers (Hogan 2013, Borderplex Alliance 2016, TWDB 2016). Other populous (populations > 20,000) urban areas in the U.S. portion of the ecoregion include Roswell, Alamogordo, and Carlsbad, New Mexico. Smaller urban areas include Artesia, Socorro, and Truth or Consequences, New Mexico, and Fort Stockton, Texas. Floodplain development for irrigated agriculture and other uses is widespread. As noted above, oil and gas production, with its associated access roads, pipelines, and waste management and pumping facilities, dominates the economy of southeastern New Mexico and a large adjacent portion of Texas.

1.4 Chihuahuan Desert REA Conservation Elements and Change Agents

The Chihuahuan Desert REA selected fourteen (14) Conservation Elements for assessment. These consist of three dry (terrestrial) ecological system types, five wet (aquatic-wetland) ecological system types, and four individual species and two assemblages of species of management concern associated with terrestrial ecological systems. Chapter 3 describes the process that led to the selection of these fourteen CEs. One of the aquatic-wetland CEs, “Playas and Playa Lakes,” has both wet (inundated) and dry phases, and thus shares features with both wet and dry system types. The term, “ecological system” here refers to “... recurring groups of biological communities that are found in similar physical environments and are influenced by similar dynamic ecological processes, such as fire or flooding” (Comer et al. 2003).

Chihuahuan Desert REA Conservation Elements are as follows:

Dry-System Conservation Elements

- Chihuahuan Desert Grasslands
- Chihuahuan Desert Scrub
- Pinyon-Juniper Woodlands

Wet-System Conservation Elements

- Montane-Headwater Perennial Streams
- Lowland-Headwater Perennial Streams
- Large River-Floodplain Systems
- Springs-Emergent Wetlands
- Playas and Playa Lakes

Species and Species Assemblage Conservation Elements

- Pronghorn
- Mule Deer
- Banner-tailed Kangaroo Rat
- Black-tailed Prairie Dog
- Grassland Bird Assemblage
- Grassland Small Mammal Assemblage

The Chihuahuan Desert REA addresses six Change Agents. These include the four overarching Change Agents addressed by all REAs: climate change, wildfire, invasive species, and development. Wildfire *per se* is a natural disturbance that historically affected most – if not all – of the fourteen CEs selected for the Chihuahuan Desert REA. However, alterations to the natural fire regime that result in *unusual* fire patterns do constitute a Change Agent. The present REA therefore includes “uncharacteristic wildfire” as a Change Agent. The “development” CA for the present REA includes crop production, various types of industrial development including oil and gas production, and urban and suburban growth. The two additional Change Agents addressed by the present REA concern excessive domestic grazing and landscape restoration. Landscape restoration is not a stressor but an intentional counter-measure against some stressors that can bring about significant changes in this ecoregion of interest to the BLM. Chapter 3 describes the process that led to the selection of these six CAs.

1.5 Chihuahuan Desert REA Management Questions

All REAs, including the Chihuahuan Desert REA, address four basic Management Questions concerning the geographic distribution of each CE, how the condition of each CE varies across its geographic distribution, the geographic distribution of each CA, and the forecasted future geographic distributions of impacts of those CAs for which forecasts are available. Table 1-1 lists these four core MQs, designated MQ A – MQ D, and indicates the CE(s) and CA(s) to which each question applies.

Table 1-1. Chihuahuan Desert REA Management Questions.

MQ #	Question	CE(s)	CA(s)
A	What is the geographic distribution of each CE?	All	n/a
B	What is the current condition of each CE across its geographic distribution?	All	n/a
C	What is the current geographic distribution of the impacts of each CA, both in general and in relation to each CE?	All	All except Climate Change, for which “current distribution” is the baseline for MQ #D.
D	What are the forecasted geographic distributions of development and climate change impacts in relation to each CE?	All	Climate Change, Development
1	Where have restoration treatments been applied to dry-system CEs, and what is the status (e.g., success rate) of those treatments?	All Dry-System CEs	Landscape Restoration
2	What is the geographic distribution of the Chihuahuan desert amphibian assemblage?	All Dry- and Wet-System CEs	n/a
3	Where would uncharacteristic wildfire likely increase sedimentation and loss of habitat among the wet systems?	All Wet Systems	Uncharacteristic Wildfire
4	What areas of potential black-tailed prairie dog habitat would support restoration?	Black-tailed Prairie Dog	Landscape Restoration
5	Where are the areas of greatest faunal species biodiversity among the species and species-assemblage CEs taken together?	All Species and Species Assemblage CEs	n/a
6	Where will urban and industrial growth impact intact grasslands or impede their recovery?	Chihuahuan Desert Grasslands CE	Development, Landscape Restoration
7	How do the current and historic geographic distributions of the dry-system CEs differ?	All Dry-System CEs	n/a
8	How will urban and industrial growth alter the geographic distribution of the grassland bird assemblage?	Grassland Bird Assemblage CE	Development
9	What and where are the aquifers and their recharge zones that support the wet systems?	All Wet-System CEs	Development
10	How do the current and historic geographic distributions of the Pecos River and Gila River fish assemblages differ?	All Wet-System CEs except Playas	n/a
11	Where are the breeding, winter, and year-around habitats for pronghorn and mule deer?	Pronghorn; Mule Deer	n/a
12	Are there areas where invasive plants are being killed on a broad scale (e.g., by the tamarisk leaf-eating beetle) where managers need to focus on restoration or controlling succession?	All Wet-System CEs	Invasive Species; Landscape Restoration
13	What is the current geographic distribution of the impacts of gypsum in the soil and water, in general and in relation to each CE and CA?	All	All except Climate Change

REAs also addresses additional MQs, focused on management concerns that cannot be resolved by individual offices alone and have regional importance. The Chihuahuan Desert REA addressed thirteen

such additional MQs, which concern: (1) interactions between specific CAs and specific CEs; (2) specific attributes or indicators of individual CEs, such as particular habitat types or particular groups of species within an ecosystem; or (3) additional environmental conditions that can affect some CEs or CAs. Table 1-1 lists these thirteen additional MQs, designated MQ 1 – MQ 13, and indicates the CE(s) and CA(s) to which each question applies.

1.6 Chihuahuan Desert REA Conceptual Ecological Models

The Chihuahuan Desert REA also developed conceptual ecological models for all fourteen Conservation Elements. These models show how the Change Agents may affect each Conservation Element and provide a means for translating Management Questions into terms specific to each individual Conservation Element and/or Change Agent. Chapter 4 briefly summarizes the purposes of conceptual models in ecological resource management and describes the methods used to develop the conceptual models for the Chihuahuan Desert REA CEs. Overarching “dry system” and “wet system” conceptual models provide a hierarchical framework for organizing and integrating the conceptual models for the individual Conservation Elements, following the recommendations of Miller et al. (2010). The conceptual models for the individual Conservation Elements differ in their methodologies.

Conceptual models for ecological resources summarize scientific understanding about (1) how and why the condition of the resource varies in response to natural variation in driver conditions, and (2) how and why it would be expected to change in response to changes in driver conditions beyond natural ranges of variation. At the same time, the conceptual model for an ecological resource necessarily also identifies the sources of information available concerning the resource and the drivers of its condition, and the certainty of this information. In effect, all statements in the conceptual model for an ecological resource constitute hypotheses about how characteristics of the resource are likely to vary or change as a result of changes in its drivers, including changes due to management actions. These hypotheses can then guide management action, including actions to test hypotheses to improve the model.

Conceptual models for ecological resources also identify key attributes of each resource that managers can use to monitor resource condition and test hypotheses about the possible effects of change agents and management actions. The Chihuahuan Desert REA refers to such key attributes as the “key ecological attributes” for each CE, on which to focus the assessment when appropriate geospatial data are available. Key ecological attributes include defining physical, biological, and ecological characteristics of a CE, along with its abundance and/or spatial distribution. The defining physical, biological, and ecological characteristics of a CE may include characteristic biological and ecological processes. When one or more key ecological attributes of a CE become stressed in a specific setting, i.e., are altered so that they depart significantly from long-term historic conditions, the entire CE in that setting is degraded or, in extreme circumstances, will disappear. A well-constructed conceptual model for a CE necessarily identifies a limited set of key ecological attributes to represent the overall condition of the CE. Ecosystem complexity, the limits of scientific knowledge, and the constraints of budgets prevent evaluation of all possible characteristics and processes of any single resource.

The conceptual models for the three dry-system CEs – Chihuahuan Desert Grasslands, Chihuahuan Desert Scrub, and Pinyon-Juniper Woodlands – are state-transition models (STMs) based on the work of the Integrated Landscape Assessment Project (ILAP; Gaines et al. 2013). Natural Heritage New Mexico (NHNM), a division of the Museum of Southwestern Biology at the University of New Mexico, developed the original conceptual models for ILAP using the Vegetation Dynamics Development Tool (VDDT, ESSA Technologies Ltd.). The resulting STMs are not spatially highly explicit. They identify large areas of land with the same potential natural vegetation type (PNVT). PNVT is largely determined by biogeochemical and disturbance processes. Land managers use data on vegetation state, soils, and disturbance dynamics to monitor system conditions and trends. The resulting STMs identify key drivers of ecological variation and change, such as fire frequency and severity, precipitation and the frequency and severity of precipitation extremes, and temperature and the frequency and severity of temperature extremes. The resulting STMs also identify key soil characteristics and key characteristics of plant community composition and structure that characterize each CE and its different vegetative states.

The conceptual models for the five wet-system, four species, and two species assemblage CEs – Montane-Headwater Perennial Streams, Lowland-Headwater Perennial Streams, Large River-Floodplain Systems, Springs-Emergent Wetlands, Playas and Playa Lakes, Species and Species Assemblage Conservation Elements, Pronghorn, Mule Deer, Banner-tailed Kangaroo Rat, Black-tailed Prairie Dog, Grassland Bird Assemblage, and Grassland Small Mammal Assemblage – are “Driver-Linkage-Outcome” (DLO) models. The DLO models presented in the present follow a methodology adapted from the methodology of the Sacramento-San Joaquin Delta (California) Ecosystem Restoration Program (ERP; https://www.dfg.ca.gov/ERP/conceptual_models.asp). DLO models identify drivers and constraints that act on an ecological resource, and the effects of those actions on characteristics of the resource such as its taxonomic and functional composition, abundance, spatial and temporal distributions, or other qualities. These effects are termed “outcomes.” The “linkages” in the model are the cause-effect relationships between drivers (or constraints) and outcomes (DiGennaro et al. 2012).

1.7 Chihuahuan Desert REA Pre-Assessment Report Structure

The Chihuahuan Desert REA Pre-Assessment report consists of 18 chapters, including the present chapter. Chapter 2 introduces the ecoregion and presents a narrative model of how the ecoregion “works” as a set of interconnected ecological systems. Chapter 3 presents the Conservation Elements, Change Agents, and Management Questions for the REA, and the processes followed to select these crucial REA building blocks. Chapter 4 describes the methods used to develop the conceptual models for the Conservation Elements. Chapters 5-17 present the conceptual models. Chapter 18 simply reviews the key management and assessment needs identified in the preceding chapters. An Appendix provides detailed tables of supporting information for the conceptual (DLO) models presented in Chapters 8-17.

2 Introduction to the Chihuahuan Desert Rapid Ecoregional Assessment

2.1 Purpose and Structure of Rapid Ecoregional Assessments

The Bureau of Land Management’s Rapid Ecoregional Assessments (REAs) seek to provide information to natural resource managers concerning (a) ecoregional-scale ecological conditions and trends, (b) the major factors that shape these conditions and trends, and (c) opportunities to conserve ecological resources across management boundaries. The REA approach integrates diverse sources of information needed to support conservation, restoration, and the development of ecological management programs in a cohesive manner.

REAs provide a foundation for adaptive ecosystem management, by summarizing current ecological understanding and providing a baseline for comparisons with future data and understanding. These comparisons can examine ecological trends and the effectiveness of management practices, for example, to help ecological resource managers assess which practices are working and where practices need to be modified. REAs do not make management decisions. They provide information to help natural resource managers make good management decisions.

REAs do not address all ecological resources in an ecoregion, as this is an impossible task. Instead, they focus on a limited set of key resources, termed Conservation Elements (CEs), consisting of regionally-significant terrestrial and aquatic habitats and species of management concern. Additionally, REAs do not attempt to assess all threats to the CEs in an ecoregion, as this is another impossible task. Instead, REAs focus on a limited set of key stressors, termed Change Agents (CAs). REAs then develop and prioritize a set of specific questions to answer concerning the CEs and CAs, termed Management Questions (MQs), which the REA seeks to address using geospatial data.

REA development occurs in two phases. During Phase I (aka the Pre-Assessment phase), the REA team identifies the Conservation Elements, Change Agents, and Management Questions on which to focus the REA. The REA team then develops conceptual models for three purposes: to identify potentially measurable key ecological attributes for each Conservation Element; to document present understanding of how each Change Agent may affect each Conservation Element; and to provide a means for translating the Management Questions into terms specific to each individual Conservation Element and/or Change Agent. The present report constitutes the Phase I or Pre-Assessment report for the Chihuahuan Desert REA. Phase II uses existing geospatial and data and publications to map the distribution of the Conservation Elements and, where feasible using existing data, assess the condition of these Conservation Elements, assess the impacts of the Change Agents, assess possible future impacts of Change Agents where appropriate, and address the key Management Questions raised during Phase I.

The word “Rapid” in the term, “Rapid Ecoregional Assessment” bears emphasis. REAs collect no new data and do not exhaustively review the literature. REAs address large-scale conditions and concerns

that cut across managerial boundaries, and work with conceptual models for conservation elements distributed across thousands of acres. REAs necessarily focus more on the kinds of broad characteristics of these large areas that can be captured in large-scale geospatial data. As a result, field offices always need to compare the results of REAs against finer-scale, local information before taking actions based on REA findings. REAs do not replace fine-scale data or the expertise of local managers. Rather, they place local concerns in a regional context and provide framework for considering integrated responses that address large scale change agents and resource issues. At the same time, REAs provide crucial information on gaps in existing large-scale data or knowledge, to help ecological resource managers identify needs for future monitoring and/or research.

2.2 Purpose and Structure of the Pre-Assessment Report

The present report is the first of two reports on the Chihuahuan Desert REA, as noted above. This Phase I or Pre-Assessment report (1) presents an overview of the boundaries, biophysical setting, biological diversity, and human landscape of the ecoregion; (2) presents the “Conservation Elements,” “Change Agents,” and “Management Questions” on which the REA focuses; and (3) presents the conceptual ecological models for the Conservation Elements. Some REA Phase I reports also discuss the identification of the geospatial data to be used in the Phase II assessment. In the present instance, however, the discussion of data identification occurs in the Phase II report.

The present report consists of 18 chapters, including the present chapter. Chapter 2 introduces the ecoregion and presents, in narrative form, a conceptual model of how the ecoregion “works” as a set of interconnected ecological systems. Chapter 3 presents the Conservation Elements, Change Agents, and Management Questions for the REA, and the processes followed to select these crucial REA building blocks. Chapter 4 describes the methods used to develop the conceptual models for the Conservation Elements. Chapters 5-17 present the conceptual models. Finally, Chapter 18 reviews the key management and assessment needs identified in the preceding chapters.

2.3 Chihuahuan Desert Ecoregion and Rapid Ecoregional Assessment Boundaries

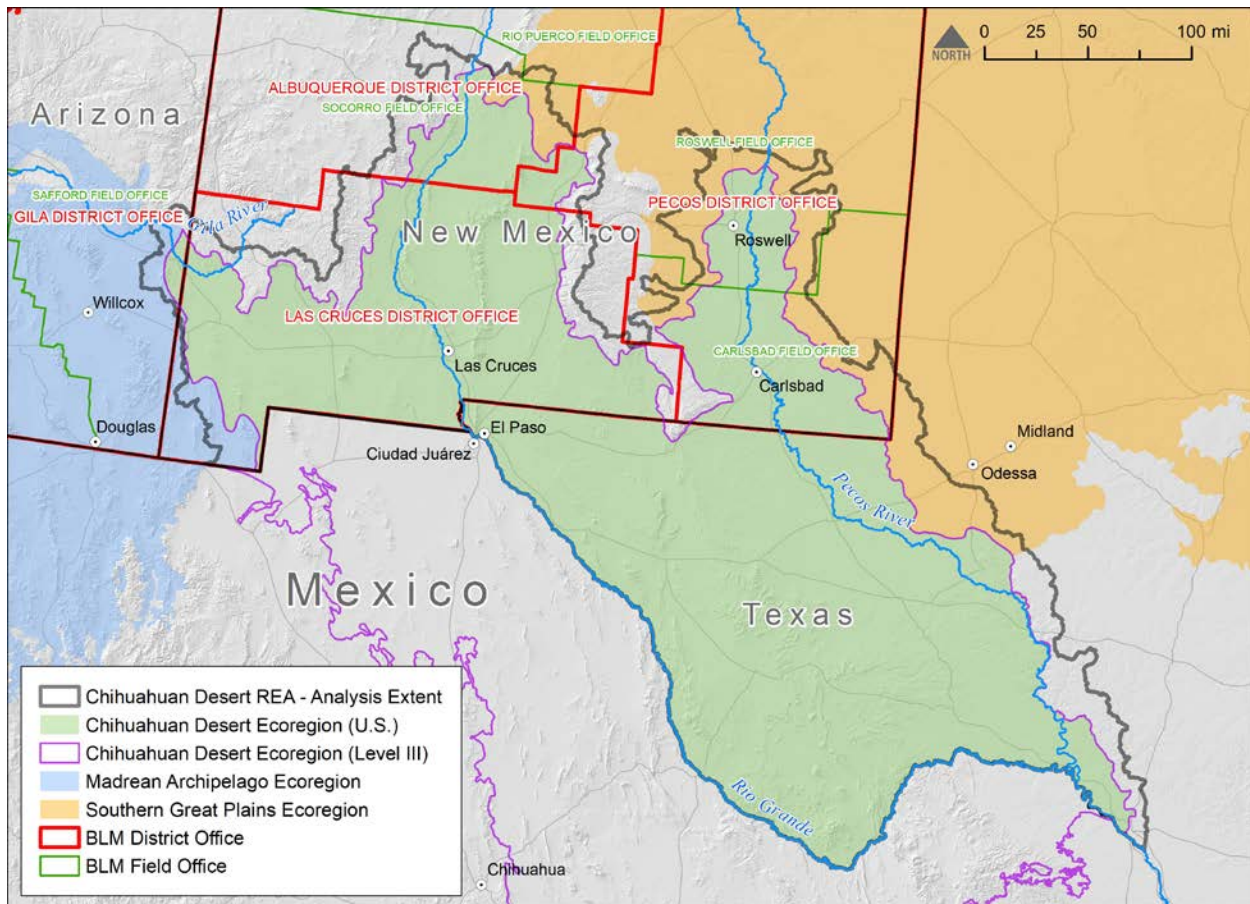
Each BLM REA focuses on an individual Level-III ecoregion or a group of adjacent Level-III ecoregions. The Chihuahuan Desert ecoregion is designated as Level-III ecoregion No. 24 (USEPA 2013) (Figure 2-1). It covers portions of both the U.S. and Mexico (Wiken et al. 2011). In fact, approximately three quarters of the Chihuahuan Desert Level-III ecoregion lies in Mexico (Dinerstein et al. 2001; Monger 2006). However, as with all REAs for ecoregions that straddle the borders of the U.S. with Canada or Mexico, the Chihuahuan Desert REAs addresses only lands within the U.S.

Figure 2-1. The Chihuahuan Desert ecoregion and REA analysis extent.



The U.S. portion of the Chihuahuan Desert ecoregion – the subject of this REA – covers significant portions of western Texas and southern New Mexico approximately from the Pecos River valley westward to the Arizona border, with a small extension into southeastern Arizona (Figure 2-2). It contains several Level-IV ecoregions, including (24a) Chihuahuan Basins and Playas, (24b) Chihuahuan Desert Grasslands, (24c) Low Mountains and Bajadas, (24d) Chihuahuan Montane Woodlands, (24e) Stockton Plateau, (24f) Rio Grande Floodplain, (24g) Gypsiferous Dunes, and (24h) Lava Malpais (Griffith et al. 2004, 2006).

Figure 2-2. U.S. portion of the Chihuahuan Desert ecoregion, analysis extent, BLM districts, and adjacent REAs.



The Chihuahuan Desert REA, as with all REAs, addresses resources not only within its target Level-III ecoregion, but also within immediately adjacent portions of the neighboring Level-III ecoregions. Specifically, using watersheds identified by a fifth-level (aka “10-digit”) Hydrologic Unit Code (HUC; Seaber et al. 1987), each REA assesses the ecological resources that occur within all *watersheds that lie within or overlap* the boundaries of the Level-III ecoregion of interest. The resulting watershed-based “analysis extent” for the Chihuahuan Desert REA includes small portions of two adjacent Level-III ecoregions: The Arizona/New Mexico Mountains and Southwestern Tablelands ecoregions (Level-III ecoregions No. 23 and 26, respectively) (Figure 2-2).

The U.S. portion of the ecoregion includes parts of the Albuquerque, Las Cruces, and Pecos BLM districts in New Mexico, which include the Roswell, Socorro, and Carlsbad Field Offices, and also includes part of the Gila District, Safford Field Office in Arizona (Figure 2-2). The BLM does not operate district offices in west Texas. The analysis extent includes approximately 201,000 km² (approx. 77,500 mi²). This includes approximately 2,000 km² (approx. 750 mi²) in Arizona, 97,000 km² (approx. 37,400 mi²) in New Mexico, and approximately 102,000 km² (approx. 39,350 mi²) in Texas. The BLM manages 36,488 km² (14,088 mi²) of these lands, including 688 km² (266 mi²) in Arizona and 35,799 km² (13,822 mi²) in New Mexico.

2.4 Chihuahuan Desert Biophysical Setting

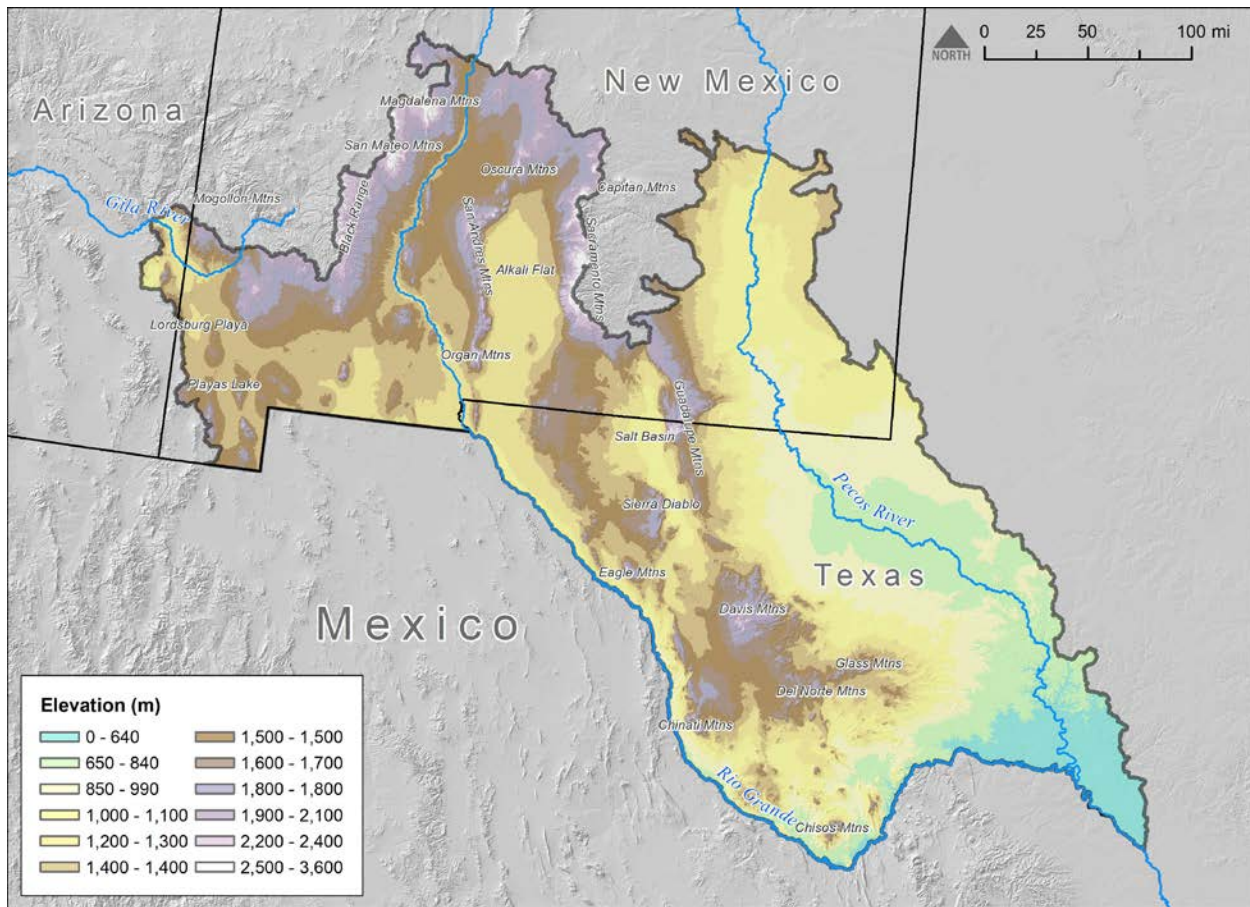
This section of Chapter 2, along with the following two sections—Chihuahuan Desert Biodiversity, and Chihuahuan Desert Human Landscape—together comprise a narrative conceptual model of how the ecoregion “works” as a set of interconnected ecological systems. Together, these three sections describe the biophysical setting that shapes the biodiversity of the ecoregion, the resulting broad patterns of biodiversity across the ecoregion, and the patterns of human land- and water-use across the ecoregion that further shape the present distribution and condition of the ecological resources of the region.

The Chihuahuan desert is the largest desert in North America, stretching from the Southwest U.S. to the Central Highlands of Mexico. It is the southernmost desert in the U.S. and the only one located east of the Continental Divide (Havstad and Schlesinger 2006). It is in some respects a relatively young ecoregion, having developed its major ecological characteristics only within the past 9,000 years, as a result of changes in climate following the end of the Pleistocene (Havstad and Schlesinger 2006). In fact, the Chihuahuan desert in the U.S. may have transitioned from grassland to shrubland and back three times in the past 3,000 years alone (Havstad and Schlesinger 2006). The perennial grasses in the northern Chihuahuan desert today, including black grama grass (*Bouteloua eriopoda*), may be relicts from wetter conditions during the mid-Holocene (Havstad and Schlesinger 2006). Since the mid-1800s, grasslands have been transitioning to native shrub-dominated desert scrubland (NMDGF 2006, Ruhlman et al. 2012). The relative importance of the possible causes for this change, including excessive livestock grazing, climate change, and altered fire regimes, remain a matter of debate (Ruhlman et al. 2012), as discussed in greater detail in Chapters 5 and 6.

In other respects, however, the Chihuahuan desert is an ancient ecoregion, having developed millions of years ago following the formation of the mountains that today comprise the Continental Divide and their resulting rain shadow. The Chihuahuan desert has experienced volcanic activity, the rising of mountains and sinking of portions of the Earth’s crust, and flooding beneath lakes and oceans for millions of years (Dick-Peddie 1993). The geologic history of the ecoregion has produced great topographic diversity, which has set the stage for the wide habitat diversity that supports the ecoregion’s high biodiversity.

The western two thirds of the ecoregion in the U.S. consist of a basin and range province of mostly north-south trending mountain ranges separated by broad desert basins (Monger et al. 2006) (Figure 2-3). Additional mountain ranges and high country occur in the Big Bend region spanning the southernmost portion of the ecoregion within the U.S. The easternmost lands of the ecoregion within the U.S. are less mountainous, consisting of portions of the Llano Estacado, a high tableland that extends across much of northwestern Texas.

Figure 2-3. Topographic relief and major landforms.



Elevation within the U.S. portion of the ecoregion (Figure 2-3) varies from about 350 meters above sea level where the Rio Grande enters Amistad Reservoir to over 2,700 meters in the San Andres Mountains in New Mexico, over 2,600 meters in the Guadalupe Mountains straddling the Texas-New Mexico border, and over 2,500 meters in the Davis Mountains in Texas. Other high mountain ranges within the U.S. portion of the ecoregion include the Organ Mountains in New Mexico and the Chinati, Chisos, and Glass Mountains in Texas. There are also numerous ranges with high peaks above 3,000 meters immediately surrounding the ecoregion, including the Sacramento, and Magdalena Mountains in New Mexico. This vast elevation range, combined with variations in topographic aspect, create a tremendous variety of macro- and micro-climates.

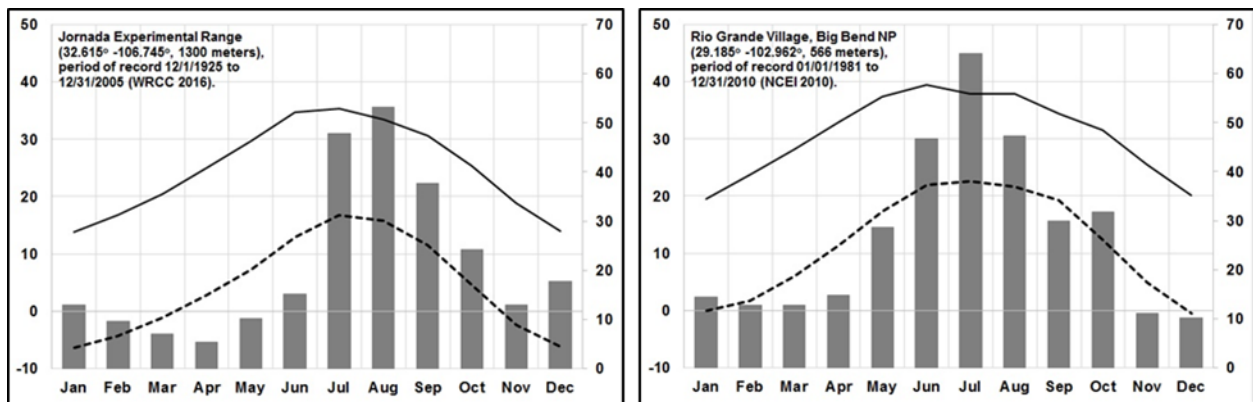
2.4.1 Climate

A wide array of flora and fauna thrive within this ecoregion, from cold-sensitive species in the Big Bend region of Texas to cold-hardy types in the mountains of New Mexico. Maximum summer temperatures in the northern Chihuahuan desert, at an elevation of 1,300 meters, average 34°C and minimum temperatures in the coldest months average -5°C (Figure 2-4) (Wainwright 2006).

Precipitation in the Chihuahuan desert is low and highly variable. The northern portion of the ecoregion in the U.S., at an elevation of 1,300 meters, receives an average of 245 mm yr⁻¹ of precipitation,

accompanied by an average 220 cm yr⁻¹ of potential evaporation (Wainwright 2006). About half of the precipitation arrives in the form of convective storms during the late summer monsoon, supplied by moisture circulated from over the Gulf of Mexico. The remainder arrives in winter synoptic storms from the Pacific Ocean. May and June are typically the driest months (Havstad and Schlesinger 2006). El Niño years typically bring 1.5 times average winter (October to May) precipitation to the northern portion of this Chihuahuan Desert REA and La Niña years see about half the average winter precipitation (Wainwright 2006). Neither El Niño nor La Niña conditions significantly affect summer moisture.

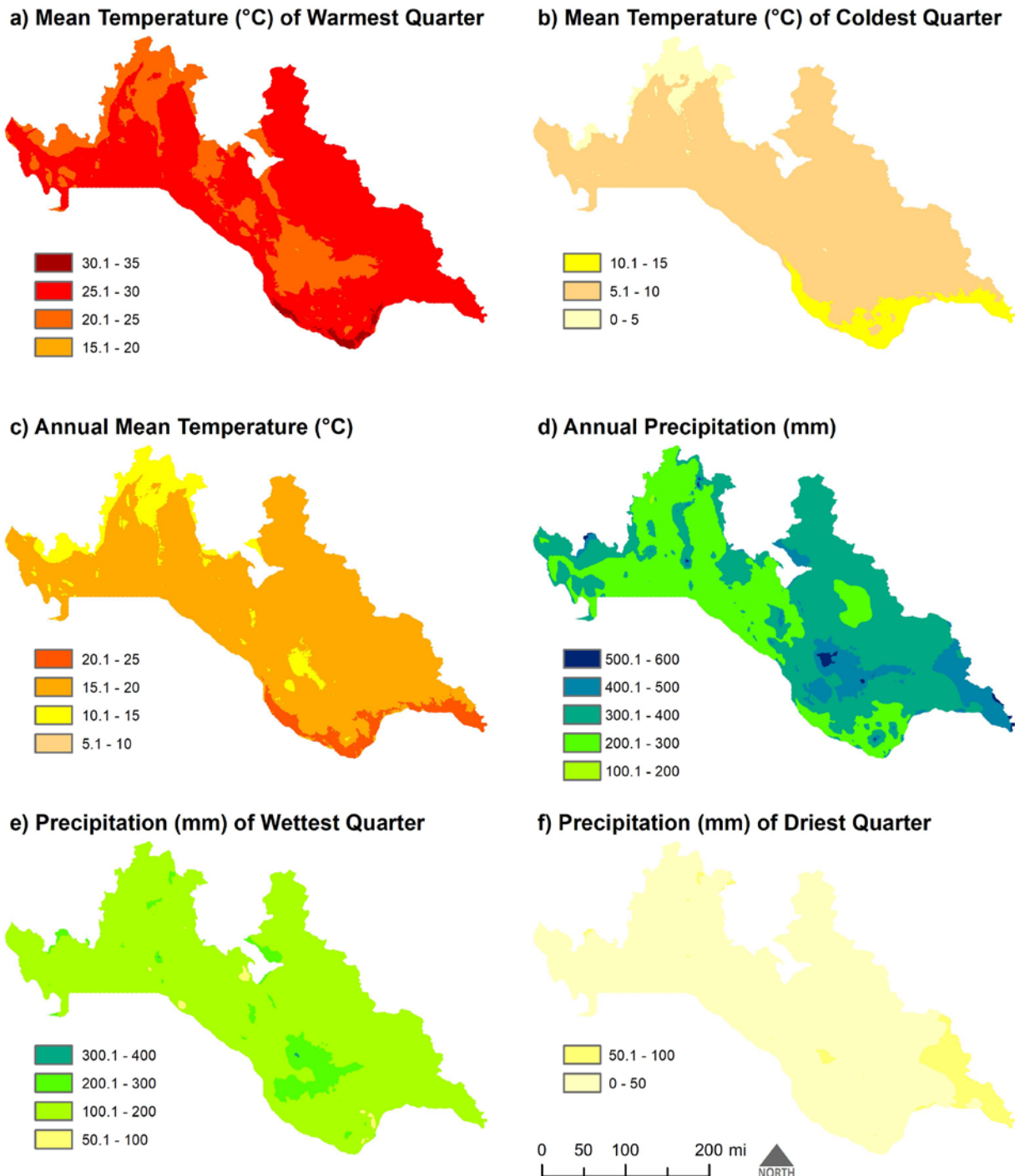
Figure 2-4. Monthly average maximum (solid line) and minimum (dashed line) temperatures (°C), left axis, and precipitation (gray bars) (mm), right axis, at the Jornada Experimental Range, New Mexico, (left; WRCC 2016), and Rio Grande Village in Big Bend National Park, Texas (right; NCEI 2010).



Several factors are responsible for the overall aridity of the ecoregion. The ecoregion is located far away from oceans, in a band of dry subtropical high pressure produced by global circulation. Additionally, it lies in the rain shadows of the Sierra Madre Occidental and Sierra Madre Oriental in Mexico and the mountain ranges along and west of the Continental Divide in the U.S. These ranges prevent most moisture-laden winds from the south and west from reaching southern New Mexico and western Texas (Schmidt 1986).

Monthly climate data for the ecoregion, averaged over the 30-year time period 1981-2010, with a resolution of 800 m were obtained from the PRISM (Parameter-elevation Relationships on Independent Slopes Model) Climate Group at Oregon State University (Daly et al. 2008 ; <http://prism.oregonstate.edu/normals/>) to examine the patterns of monthly, quarterly, and annual variation in climate variables across the ecoregion. Quarterly variables are roughly equivalent to different seasons; they are defined in terms of three consecutive months with hottest or coldest temperatures or largest or smallest amounts of precipitation (e.g., mean temperature of warmest quarter or precipitation of wettest quarter). Figure 2-5 summarizes six seasonal and annual climate variables for the U.S. portion of the ecoregion: the mean temperatures of the warmest and coldest quarters, annual mean temperature and mean annual precipitation, and the mean precipitation of the wettest and driest quarters. Variation in these six climate variables is shaped by global, continental, and regional atmospheric systems, rain-shadow effects, and elevation.

Figure 2-5. Spatial patterns of six climatic variable: Mean temperature of the a) warmest and b) coldest quarters; c) annual mean temperature; d) annual precipitation; and precipitation of the e) wettest and f) driest quarters.



The temperature variables, especially annual temperature and mean temperature of the warmest quarter, show similar spatial patterns across the ecoregion (Figure 2-5(a), 2-5(c)). In particular, temperatures are higher in the southeastern-most portion of the region, including the area around the

Rio Grande on the border between Texas and Mexico and Big Bend Ranch State Park in west Texas. Temperatures are coolest in the north near Socorro, New Mexico and in multiple mountain regions including the Chinati, Chisos, Davis, and Eagle Mountains and Sierra Diablo in Texas and Organ, Oscura, and San Andres Mountains in New Mexico (Figure 2-3). There are also cooler temperatures in the western portion of the region near Silver City, New Mexico. The mountain ranges in Texas do not appear to be as cool relative to their surroundings during the coldest quarter as do the San Andres Mountains and other mountains in New Mexico relative to their surroundings (Figure 2-5(b)).

Annual precipitation and precipitation during the wettest quarter show fairly similar patterns, with areas of higher rainfall roughly corresponding to areas with cooler temperatures as described previously (Figure 2-5-(d), 2-5(e)). There are however some differences between areas of higher rainfall and relatively cooler temperatures. The area of higher precipitation in the northern-most portion of the ecoregion is more limited to mountain ranges; the area around Socorro, New Mexico does not receive higher levels of rainfall. The region of higher precipitation in western Texas is larger than the area of cooler temperatures and encompasses some additional mountain ranges, including the Del Norte and Glass Mountains, both east of the Davis Mountains. There is a region of higher precipitation north of Carlsbad Caverns National Park in the northeastern portion of the region, and another region of higher precipitation in the southeast along the Rio Grande valley, extending up to near McCamey, Texas. The latter region receives higher rainfall during the driest quarter (Figure 2-5(f)) but has higher temperatures and lower rainfall during the wettest quarter.

These six climate variables provide the ecologically most useful summary information on seasonal and annual climate variation in the U.S. portion of the ecoregion among the many variables available in the PRISM dataset. For example, black-tailed prairie dogs (*Cynomys ludovicianus*) may have lower survival in years with lower precipitation (Facka et al. 2010). Black-tailed prairie dog is Conservation Element of the REA (see Chapters 3 and 15). For another example, annual precipitation is correlated with scaled quail (*Callipepla squamata*) abundance in the southern portion of the Chihuahuan Desert REA region (Bridges et al. 2001). Scaled quail is one of the key species of the Grassland Bird Assemblage, another Conservation Element of the REA (see Chapters 3 and 16). Further, scaled quail may not tolerate extremely high temperatures (~45 °C) well (Henderson 1971).

Similarly, the three large-scale terrestrial ecological system types selected as Conservation Elements for the REA (see Chapters 3 and 5-7) have distributions strongly affected by climate and its interactions with topographic elevation across the ecoregion. Chihuahuan Desert Grasslands (Chapter 5) occur on piedmonts, foothills, and lowlands mainly between 1,100 and 1,700 m in elevation. These topographic zones tend to experience slightly less precipitation and slightly cooler temperatures than do areas at lower elevations in the U.S. portion of the ecoregion, which in turn are dominated by the Chihuahuan Desert Scrub ecological system type (Chapter 6). Conversely, the Pinyon-Juniper Woodlands ecological system type (Chapter 7) generally at moderate elevations between 1,400 and 2,200 m in elevation. These elevations experience slightly greater precipitation and slightly cooler temperatures compared to areas dominated by Chihuahuan Desert Grasslands. Pinyon-juniper woodlands in fact are often bordered by grasslands at their lower elevations. The separation of grasslands from woodlands in the

ecoregion reflects the impacts of precipitation patterns and long-term climate variations on fundamental aspects of plant physiology: The success of C₄ perennial grasses at low elevations depends on summer precipitation (Havstad and Schlesinger 2006), while C₃ shrubs in grasslands and woody species at higher elevation rely more on winter precipitation (see also Chapters 5-7).

2.4.2 Geology

The geologic history of the U.S. portion of the ecoregion affects the ecological dynamics and biological diversity of the region in several ways, through its effects on topography, hydrography, and geochemistry. The following paragraphs summarize relevant aspects of the geologic history of the ecoregion and their consequences for the ecology of the region, beginning with the late Pennsylvanian and early and middle Permian geological periods, ca. 323-260 million years ago.

The Chihuahuan desert ecoregion during the late Pennsylvanian and early and middle Permian geological periods lay within the Pangea supercontinent. An inland sea developed within this supercontinent, filling a geologic basin, now called the Permian Basin, which spanned most of what is now west Texas and southeastern New Mexico. The inland sea connected to the ocean only through a narrow inlet. A lobe of the Permian Basin, called the Delaware Basin, spanned what is today most of the U.S. portion of the ecoregion, roughly from the San Andres Mountains eastward in New Mexico and across all but the Big Bend region and adjacent westernmost portions of the ecoregion in Texas. A massive system of reefs—the Capitan Reef complex—formed around the periphery of the Delaware Basin during the middle to late Permian period and later fossilized as limestone. Erosion from the surrounding uplands and evaporation from the inland sea filled the Permian Basin and all its lobes with vast deposits of halite and sulfate mineral salts interlayered with silicates (clay, silt, and sand) and organic matter (Urbanczyk et al. 2001, Monger et al. 2006, George et al. 2011).

Tectonic dynamics following the Permian period raised the Delaware Basin reef limestone and basin sedimentary layers – including their mineral salt layers – high above sea level. The uplift also affected sedimentary rocks that formed after the Permian during another period of marine inundation in the Cretaceous period across the entire portion of the ecoregion in Texas, including what is now the Big Bend Region. Subsequent tectonic extension, faulting, and downward displacement of grabens between faults (approximately 36-20 million years ago) created most of the mountain ranges and intervening basins of the U.S. portion of the ecoregion today. Additional topographic relief within the U.S. portion of the ecoregion comes from intrusive volcanic activity during the Tertiary period, particularly in the westernmost portions of what is now Texas. The Rio Grande valley in the ecoregion consists of a series of large basins created during the period of formation of the basin and range province.

The valleys of the basin and range province may contain thousands of feet of sedimentary “basin fill” from the erosion of the surrounding ranges, which now store substantial volumes of groundwater of critical importance to the people of the ecoregion (Kelley 1971, Bachman 1980, Hill 2000, George et al. 2005, Huff and Chace 2006, Monger et al. 2006, George et al. 2011). Some of the basins created by the pattern of faulting of the Permian and Cretaceous sedimentary rocks of the region are closed valleys or “bolsons” – valleys with no surface drainage outlet. Heavy precipitation and runoff during the

Pleistocene created lakes in these valleys, some quite large, which subsequently dried, leaving playa lakes and playas scattered throughout the ecoregion (Hawley 1993, Wilkins 1997, Langford 2003, Allen 2005). The lakes were a crucial resource for early Native Americans in the region, and numerous archaeological sites occur along their ancient shorelines (e.g., Monger 1993, Hill and Holliday 2011).

The Capitan Reef complex today forms the most resistant rocks of several mountain ranges in the ecoregion, including the Guadalupe Mountains that straddle the New Mexico-Texas border and the Glass Mountains in Texas. Erosion of salts—both halites and sulfates—from the exposed Delaware Basin sedimentary rocks has resulted in the re-accumulation of these evaporites in the bottoms of all the closed valleys that lie within the original extent of these rock formations. The evaporites include both salt and gypsum. The heavy precipitation of the Pleistocene resulted specifically in the formation of large, *salty* lakes in these particular valleys, most of which have now also evaporated leaving new re-accumulations of salts, for example in the Tularosa Basin and in the Salt Basin near Dell, Texas (Bachman 1981; 1987, Hussain and Warren 1989, Hawley 1993, Angle 2001, Monger et al. 2006, NPS-CDIMN 2010). Evaporation from gypsum-rich lakes in these closed valleys has also sometimes given rise to dune fields of gypsum crystals, such as at White Sands in the Tularosa Basin (NPS-CDIMN 2010, Szykiewicz et al. 2010).

Even outside these closed valleys, exposures of Delaware Basin sedimentary deposits contribute salt and gypsum to the overlying soils (El Hage and Moulton 1998, Monger et al. 2006, McCraw et al. 2007, Moore and Jansen 2007, McCraw 2008, Hudnall and Boxell 2010). Watershed runoff from these areas results in elevated salt and gypsum concentrations in the receiving streams, including the Pecos River (El Hage and Moulton 1998, Cowley et al. 2003, Miyamoto et al. 2005, Hoagstrom 2009, Stafford et al. 2009, Hogan 2013, Szykiewicz et al. 2015a; 2015b). Subsurface leaching of mineral salt deposits from Delaware Basin sedimentary deposits also results in the formation of karst landscapes, through the collapse of caves created by the dissolution of mineral salts. This subsurface leaching produces elevated concentrations of halites and gypsum in the groundwater that passes through these deposits – groundwater that in turn may emerge in sinkholes, at springs, or along seepage faces beneath streams (e.g., recently, Stafford et al. 2008a; 2008b; 2008c; 2009, Land and Huff 2010, Partey et al. 2011, Land and Veni 2012, Szykiewicz et al. 2012, Stafford 2013, Sigstedt et al. 2016).

Salts in soils and water, in general, and gypsum in soils and water, in particular, place severe physiological demands on any plants or animals that may live there, selecting for biota with unique adaptations (e.g., Moore et al. 2015). The geologic history of the U.S. portion of the Chihuahuan desert ecoregion therefore has selected for an array of plant and animal species with unique physiological adaptations to the saline and/or gypsiferous soils, playas and playa lakes, springs, and streams of much of the area (e.g., Waterfall 1946, Miller 1977, Blinn 1993, Edwards 1997, Hoagstrom and Brooks 1999, Propst 1999, MacRae et al. 2001, Lang and Rogers 2002, Cowley et al. 2003, Howells 2003, Lang et al. 2003, White et al. 2006, Grunstra and Van Auken 2007, Moore and Jansen 2007, Hoagstrom 2009, Turner et al. 2010, USFWS 2010, Turner and Edwards 2012, USBR 2012, USFWS 2013).

The geology of the Permian Basin also has shaped the human history of the ecoregion. The organic matter trapped in the deposits of the basin has become oil and natural gas. The Permian Basin geologic

formations comprise the most productive oil and gas fields in North America (e.g., Ruhlman et al. 2012). The industrial activities associated with the exploitation of oil and gas – production, transport, waste disposal, and so forth – consequently have a large footprint in the ecoregion, most notably in the Pecos River valley (see below and Chapter 3). Employment associated with these activities in turn has and continues to significantly affect the demography and economy of the ecoregion. Oil and gas production, brine production, and other industrial activities such as at the nuclear Waste Isolation Pilot Plant (WIPP) east of Carlsbad, New Mexico, can produce briny wastes that must be controlled or treated to prevent contamination of surface waters (Bachman 1980, 1981, 1987; Siegel et al. 1991; Meyer et al. 2012; Klise et al. 2013; Land 2013; Sullivan et al. 2015). Brine production can also result in the creation of anthropogenic sinkholes (Land and Veni 2012; Land 2013). Irrigation of soils with elevated salt concentrations results in return flows with elevated salinity, raising salinity in the receiving rivers and thereby affecting downstream usability of the water (e.g., El Hage and Moulton 1998; Cowley et al. 2003; Miyamoto et al. 2005; Hoagstrom 2009; Stafford et al. 2009; Hogan 2013; Szykiewicz et al. 2015a, 2015b). Salinity in groundwater can also limit its usability (Mace et al. 2001; Huff 2004a, 2004b; Mills 2005; George et al. 2011; Meyer et al. 2012; Klise et al. 2013), rendering nearly uninhabitable lands with no other source of potable water.

Finally, as already noted, and discussed further, below, the Chihuahuan desert itself exists specifically because of the positioning of several mountain ranges relative to prevailing atmospheric circulation patterns. The rise of these mountains has also shaped the history of aquatic ecosystems by determining which river basins are connected to or isolated from each other (Miller 1977). The Continental Divide separates the biota of the Colorado River Basin to the west from the rest of the U.S. portion of the ecoregion; and also isolates numerous closed basins from the rest of the ecoregion. Not all of these aquatically isolated basins occur geologically within the area of the former Delaware Basin: others occur in the far western extension of the ecoregion in New Mexico, including the Guzmán and Lordsburg basins in the western extension of the ecoregion. However, mountains alone do not set the aquatic ecological boundaries of the ecoregion. On the east side of the ecoregion, the northern half of the present-day Pecos River basin formerly constituted the headwaters of the Portales River, which flowed eastward across the southern Great Plains. The headwaters of the original Pecos River, to the south, eroded northward to capture the waters of the northern Portales River Basin during the Pleistocene (Bachman 1987), introducing fishes from the Great Plains into the Pecos River ecosystem.

2.4.3 Soils

The complex geologic and climatic history of the Chihuahuan Desert ecoregion has produced a range of parent materials, on which myriad soils have developed. Soils at higher elevations receive more precipitation and are generally acidic, leached, and well-developed (Maker et al. 1974). Mollisols and Entisols occur in mountain ranges of this REA area. The former contain a surface layer of high organic matter and the latter include shallow soils over bedrock (NPS CDIMN 2010). Soils in lower and drier areas are less developed and typically neutral to alkaline (Maker et al. 1974). These soils are often Aridisols, which contain accumulations of calcium carbonate (CaCO₃) (NPS CDIMN 2010). As noted above, gypsiferous soils and saline soils develop in closed basins and in areas with near-surface bedrock containing mineral salts, creating conditions in which endemic plant species have evolved (Hendrickson

1979, Powell and Turner 1979, Moore and Jansen 2007, Hudnall and Boxell 2010).

Soil nutrient cycling in the northern Chihuahuan desert depends in large part on invertebrates. Subterranean termites (order Isoptera) are keystone organisms and important recyclers of dead plant material and animal dung (Dinerstein et al. 2001, Schlesinger et al. 2006). The ability of these invertebrates to function with limited moisture makes nutrient cycling in this ecoregion less reliant on timing of precipitation than is the case elsewhere (Schlesinger et al. 2006). Specialized soil mites in the ecoregion are also important to nutrient cycling in the dry climate (Dinerstein et al. 2001).

2.4.4 Hydrology

Perennial streams and rivers, springs, cenotes, seeps, playa lakes, and reservoirs create corridors, oases, and expanses of aquatic habitat across the U.S. portion of the Chihuahuan desert, with associated wetlands (e.g., riparian wetlands, ciénegas, and emergent wetlands). Intermittently wetted runoff channels and playas also contribute to the diversity of wetted habitats in the ecoregion. (The REA does not address subterranean biota, and addresses intermittent streams only as components of the terrestrial ecological systems in which they occur). Chapter 3 describes how the REA divides the “wet” ecological systems of the ecoregion into Conservation Elements, and Chapters 8-11 discuss these key ecological systems in detail.

With three notable exceptions, the natural water bodies (streams, springs, cenotes, seeps, playas and playa lakes) of the northern Chihuahuan desert receive their water ultimately from the rainfall and snowfall that occurs within the ecoregion itself. Some of this “local” water reaches its destination simply as runoff, but much arrives only after infiltrating to a groundwater flow system that may take years to millennia to deliver the water to a natural surface outflow. The three notable exceptions to this pattern are the Rio Grande, the Pecos River, and the Gila River.

The Rio Grande is the largest river system flowing through the U.S. portion of the ecoregion, and one of the ten longest rivers in North America. It originates as snow melt and rainfall in the mountains of southern Colorado and northern New Mexico, flows south through New Mexico, and serves as the border between Texas and Mexico. Within the U.S. portion of the ecoregion, it receives additional inputs from local runoff and from groundwater discharge, including scattered springs in the Big Bend region (NPS CDIMN 2010). It has no large tributaries within the U.S. portion of the ecoregion. However, a major tributary, the Rio Conchos, enters the Rio Grande a few miles south of El Paso, Texas, and most of the flow of the river below this confluence – including through the Big Bend region – consists of water from this tributary.

The Pecos River originates in the Sangre de Cristo Mountains of north-central New Mexico and flows southward along the eastern edge of the basin and range province and the western edge of the Great Plains. As noted above, its course north of the ecoregional boundary reflects its history of capturing the former upstream reach of the Portales River. Within the Chihuahuan desert ecoregion, the Pecos River flows south- and eventually southeastward to join the Rio Grande near the southeastern edge of the U.S. portion of the ecoregion. It receives inflows from several substantial perennial tributaries within the ecoregion, including the Rio Hondo, Rio Peñasco originating in mountains along the west side of its

watershed, and the Black River, which also arises on the west side of the watershed but primarily from groundwater outflows rather than from montane rainfall and snowmelt.

The analysis extent for the Chihuahuan Desert REA includes a small portion of the upper Gila River basin and the Gila River mainstem along the Arizona-New Mexico border. The Gila River originates in the high elevations of the Arizona-New Mexico Mountains ecoregion, adjacent to the Chihuahuan Desert ecoregion but on the west side of the Continental Divide, and flows westward through Arizona to the Colorado River.

Verdant riparian areas once occurred extensively along the Rio Grande, Pecos River, and Gila River, their perennial tributaries, and the Mimbres River, which flows out of the Mogollon Mountains into the closed Guzmán Basin. Where they still occur, these riparian areas provide diverse mesic and wetland habitats, including in the Bosque del Apache and Bitter Lake National Wildlife Refuges along the Rio Grande and Pecos Rivers, respectively, and along the Rio Grande through the Big Bend region. High water tables in alluvial aquifers along some reaches of these perennial rivers and streams once helped support their riparian woodlands and wetlands. The Rio Grande and Pecos Rivers today are highly regulated, impounded behind numerous dams, and extensively diverted for human use in the ecoregion as discussed further below (see Water Use, this Chapter). Pumping from the alluvial aquifers also has significantly lowered their storage, as has pumping from other aquifers in the ecoregion (see Water Use, this Chapter). Climate change is expected to reduce the ability of the Rio Grande system to support ecosystems, agriculture, and cities. A recent study found that “[t]he Rio Grande offers the best example of how climate-change-induced flow declines might sink a major system into permanent drought” (Dettinger et al. 2015).

The U.S. portion of the ecoregion contains numerous perennial springs, as noted above (e.g., Brune 1975, Heitmuller and Williams 2006). The largest of these originate in aquifers within Cretaceous sedimentary rock formations in Texas. These springs include the well-known San Solomon Springs complex in the Pecos River valley near Balmorhea, Texas, a popular recreation site, as well as the nearby Phantom Lake, Diamond Y, and Comanche Springs; and the numerous springs of Big Bend National Park. Further north, Bitter Springs National Wildlife Refuge near Roswell, New Mexico, also has numerous springs and cenotes, as does the nearby Bottomless Lakes State Park, New Mexico. The aquifers within the Cretaceous sedimentary rock formations in Texas are important water resources for the ecoregion, as are aquifers in the basin fill deposits of the basin and range province.

The U.S. portion of the ecoregion also contains numerous ephemeral water bodies (Dinerstein et al. 2001). Specialized organisms, including many endemic invertebrates, live in playas and pools across the ecoregion, many of which have highly brackish chemistry, as noted above (e.g., Lang and Rogers 2002; Lang et al. 2003). The disjunctive nature of these water bodies has created high beta diversity among their biota. Assemblages of freshwater invertebrates in playas are important food for migrating waterfowl (Dinerstein et al. 2001).

2.4.5 Wildfire

Wildfire historically played a significant role in the ecological dynamics of the northern Chihuahuan

desert, episodically resetting and directing plant succession across burn areas of varying size. The frequency and effects of fire varied among the different major terrestrial ecological systems (see detailed discussions in Chapters 3 and 5-7). Furthermore, the frequency, spatial extent and severity of wildfire in the U.S. portion of the ecoregion have shifted since the late 1800s. Livestock introduced by European settlers removed the grasses that carry fire and settlers suppressed fires that did ignite (Gebow and Halvorson 2005). Heavy, year-round grazing that resulted in less frequent fires are thought to have allowed shrubs to expand into grasslands (Ruhlman et al. 2012). Climate change likely exacerbates both shrub expansion and changes in fire regimes (Ruhlman et al. 2012).

Historic fire return intervals in Chihuahuan grasslands were short (~10 years for low severity fires) (USDA 2012). Prior to the changes that began in the late 1800s, the patchy distribution of grassland vegetation may not have been highly conducive to the development of large wildfires (Dick-Peddie 1993). However, prior to the late 1800s, some fires burned more than a hundred square miles (Bahre 1991, Humphrey 1949, McPherson 1995).

Wildfire historically helped maintain Chihuahuan desert grasslands by limiting shrub dominance (Bahre and Shelton 1993; Humphrey 1958; McPherson 1995). However, when grazed, black grama grass, the dominant species in Chihuahuan desert grasslands loses vigor and does not provide sufficient fine fuels to carry fire very well enabling shrubs to recruit into, and eventually dominate, grassland communities (Cable 1965, Drewa and Havstad 2001, Peters and Gibbens 2006).

In contrast to desert grasslands, desert scrub systems have sparse fine fuels making them largely fire-resistant. Fire return intervals for these scrub systems is thought to range up to 350 years (USDA, 2012).

Other factors than fire alone—particularly drought dynamics—appear to have played a greater role in shaping where different plant species grow in the kinds of desert grasslands found in the northern Chihuahuan desert (Burgess 1995) It is likely that the interaction of large numbers of cattle foraging on the region's perennial grasses, and dispersing mesquite seed, combined with periodic severe drought and subsequent reduction in fire frequency and severity has led to the landscapes observed today.

At higher elevations, in turn, pinyon-juniper woodlands burn readily. The distribution, composition, and condition of pinyon-juniper woodlands in the Chihuahuan desert are highly sensitive to the seasonal timing, frequency, and severity of wildfire (see Chapter 7). Wildfires of low to mixed severity in pinyon-juniper woodlands in the ecoregion have become rare because livestock grazing has reduced fuel loads, fire suppression has limited fire spread, and droughts have reduced production of fuels. These changes have also promoted soil erosion in some areas, leaving insufficient understory fuel loads to carry fires (Gori and Bate 2007). These changes have allowed pinyon-juniper trees to spread and become increasingly abundant in adjacent terrain. This expansion often reduces herbaceous cover, especially on shallow soil, reducing the potential for low-intensity fires carried by understory fuels and increasing the potential for severe fires that harm all vegetation.

Burn intervals prior to the late 1800s may have been as short as 6–7 years in mountain shrub communities and 4–9 years in grasslands (Gebow and Halvorson 2005). Chihuahuan Desert grasses

generally recover well after fire, given adequate moisture, and most mature trees and shrubs resprout after infrequent, low-intensity fire (Gebow and Halvorson 2005). Succulents, on the other hand, are damaged by fire, especially during dry periods (Gebow and Halvorson 2005). Gebow and Halvorson (2005) concluded from their literature review that fire is a natural part of northern Chihuahuan Desert communities, but that the fire regime at the landscape scale is complex with fire frequency and severity controlled by climatic, geographic and ecological drivers. They suggest that a “mixed” fire regime was typical before European settlement, with patchy, variable fire histories across landscapes.

2.4.6 Large-Scale Biogeography

The U.S. portion of the Chihuahuan desert does not exist in biological isolation. Not only is it a portion of a much larger ecoregion that extends deep into Mexico, but it is surrounded by many other ecoregions (Figure 2-1 and Figure 2-2, above). The neighboring ecoregions include the Sonoran Basin and Range, Madrean Archipelago, Arizona/New Mexico Plateau, Southwestern Tablelands, Arizona/New Mexico Mountains, High Plains, Edwards Plateau, and South Texas Plains. The northern Chihuahuan desert shares biota with this larger biogeographic region, within which no substantial barriers prevent the movement of terrestrial plants, vertebrates, and many invertebrates. This does not mean that there are no opportunities for endemism among terrestrial species within the U.S. portion of the ecoregion: the northern Chihuahuan desert contains many unique habitat conditions that have selected for unique terrestrial biota, as noted above and discussed further below and in subsequent chapters.

The native aquatic and wetland-obligate plants and animals of the northern Chihuahuan desert present a somewhat more complicated situation. A tiny portion of the ecoregion lies west of the Continental Divide. The fish fauna largely differ across the divide, but some mixing has occurred between headwater streams that arise along the divide (see detailed discussion and citations in Chapter 8). The closed basins within the northern Chihuahuan desert are hydrologically isolated from each other today. However, the Mimbres River has its headwaters along the Continental Divide and appears to have been accessible from the east at some time(s) in its history as well – and consequently shares fish fauna with the rivers of the ecoregion on both sides of the divide (see detailed discussion and citations in Chapter 8 and 9). Fish uniquely adapted to springs and other hydro-geo-chemically unique settings in the northern Chihuahuan desert have limited distributions. However, their current habitats occur in settings that at least episodically may become connected to larger drainage networks, and they appear to have evolved through local selection from ancient members of larger, more widespread taxonomic groups (see detailed discussion and citations in Chapters 8-10). Otherwise, the taxonomic groups of aquatic fauna native to the northern Chihuahuan desert are all members of groups found across the entire Chihuahuan desert ecoregion of the Mexico and the U.S. (Miller 1977). This includes native taxa also found in other river systems tributary to the northwestern Gulf of Mexico. Finally, as a result of the capture of the Portales River of the southern Great Plains by the Pecos River, the latter river also contains aquatic fauna native to the southern Plains as well (see detailed discussion and citations in Chapters 8 and 9). The discussions of endemism, below (see next section of this chapter, Chihuahuan Desert Biodiversity), provides additional information on endemism among the aquatic fauna of the ecoregion.

2.5 Chihuahuan Desert Biodiversity

The Chihuahuan desert is one of the three biologically most diverse desert regions in the world (Dinerstein et al. 2001). Only the Great Sandy-Tanami Desert of Australia and the Namib-Karoo of southern Africa are believed to contain greater biological diversity (Dinerstein et al. 2001). Of these three, only the Chihuahuan desert supports both rich terrestrial and rich freshwater biotas (Dinerstein et al. 2001). The ecoregion is the center of diversity of Cactaceae, which includes many regional endemics. Many other plants and reptiles exhibit high endemism and rapid species turnover across Chihuahuan desert landscapes (Dinerstein et al. 2001). The complexity of the ecoregion's freshwater fish assemblages completes the remarkable level of ecosystem biodiversity in the ecoregion (Dinerstein et al. 2001). The following sections of this chapter briefly summarize various aspects of this unique diversity.

2.5.1 Species Richness

This Chihuahuan desert overall—both the U.S. and Mexican portions together—contains over 2,000 known species of vascular plants, over 100 species of mammals, over 100 species of reptiles, over 200 species of birds, over 200 species of butterflies, and roughly 20 amphibian species (Dinerstein et al. 2001). This remarkable richness reflects the diverse topography and climates of the ecoregion, the connectedness of the ecoregion to surrounding ecoregions and biogeographic provinces, and its mesic past.

The basin and range landscape of the ecoregion creates islands of disjunctive terrestrial communities, the dry climate of the ecoregion overall creates innumerable isolated water bodies, and the unique geochemistry of the ecoregion creates settings that have selected for unique terrestrial and aquatic adaptations. Additionally, wide *variation* in climate over the last 10,000 years produced a range of conditions that supported a wide array of flora and fauna (Dinerstein et al. 2001), some of which have persisted in the ecoregion after subsequent climate changes have set in. For example, regional drying 9,000 years ago (Havstad and Schlesinger 2006) isolated mesic-adapted species in pockets of suitable habitat. As a result, familiar mesic-adapted birds, including blue jay (*Cyanocitta cristata*) and yellow-throated vireo (*Vireo flavifrons*) inhabit riparian forests along the Pecos River (Dinerstein et al. 2001). Eastern U.S. invertebrates, such as fireflies (Lampyridae), occur in the Davis Mountains (Dinerstein et al. 2001). And among the Chihuahuan desert herpetofauna, six species are considered relict species from forested regions (Milstead 1960).

2.5.2 Endemism

The Chihuahuan desert is renowned as the global center of diversity for Cactaceae (Dinerstein et al. 2001), as noted above. Over 300 of the world's 1,500 species of Cactaceae occur in this ecoregion and many of these are endemic (Dinerstein et al. 2001). Five other plant families—Poaceae, Euphorbiaceae, Asteraceae, Fabaceae, and Nyctaginaceae—reach high levels of species richness and endemism in the basin and range province of the ecoregion (Dinerstein et al. 2001).

Prior to the formation of the present day Chihuahuan desert, the ecoregion supported a diverse aquatic

biota in plentiful freshwater habitats. Tectonic activity and regional drying over millions of years reduced freshwater habitats, isolating populations from each other (Dinerstein et al. 2001). Pluvial lakes occasionally covered much of the ecoregion, connecting many currently isolated basins hydrologically and allowing species exchanges (Dinerstein et al. 2001). Today, many of these basins are now disjunct with strikingly high rates of endemism, especially among the cichlid (Cichlidae) and cyprinid (Cyprinidae) fishes (Dinerstein et al. 2001). Even where fish fauna have disappeared in these closed basins, distinct suites of endemic aquatic macroinvertebrates still persist (Dinerstein et al. 2001; Lang and Rogers 2002; Lang et al. 2003).

Springs provide the source of many perennial streams in the Chihuahuan desert and individual springs are often the sole habitats of highly endemic species (Dinerstein et al. 2001). Pupfish of the genus *Cyprinodon* and livebearing members of the genus *Gambusia* dominate the piscifauna of springs in the ecoregion (Dinerstein et al. 2001). Springs and their outflow in the Tularosa Basin are the only known habitat for the White Sands pupfish (*Cyprinodon tularosa*). Similarly, springs and their outflow streams in the Pecos River valley are the only known habitat for the Leon Spring pupfish (*Cyprinodon bovinus*), although the species may once have been present in the mainstem Pecos River as well (see Chapter 8, below). Chapters 8 and 10 provide additional information on endemic fishes associated with springs in the ecoregion.

The largest gypsum dune field in the world occurs in the northern Chihuahuan desert, in the dry, closed Tularosa Basin of southeast New Mexico. Endemic gypsophilic plants and white variants of some animals have adapted to conditions in the 71,000-hectare dune field (NPS 2005). Gypsophilic plants occur not only in the Tularosa Basin but across much of the ecoregion, are most numerous in Nyctaginaceae, Brassicaceae, Boraginaceae, Caryophyllaceae, Fouquieriaceae, Papaveraceae, Loasaceae, Onagraceae, Asteraceae, Poaceae, and Scrophulariaceae (Moore and Jansen 2007, Moore et al. 2015, Moore 2015). The richness and diversity of gypsophilic flora in the ecoregion suggest this is a relatively old assemblage (Moore and Jansen 2007, Moore et al. 2015, Moore 2015). This inference of long-term persistence and evolution in the Chihuahuan desert gypsophilous plant assemblage is further supported by the existence of several genera within the ecoregion containing multiple endemic gypsophilous plant species (e.g., *Gaillardia*, *Nama*, and *Tiquilia*) and the similarity among endemic gypsophilous plant species across the entire ecoregion (Moore and Jansen 2007, Moore et al. 2015, Moore 2015).

The Tularosa Basin also contains areas of saline soils, as do countless other large and small closed basins in the ecoregion (Hendrickson 1979) for the geologic reasons described above. Halophytic plant species are most numerous in Chenopodiaceae and Poaceae and the former includes about a dozen halophytic plant species endemic to the ecoregion (Hendrickson 1979).

2.5.3 Characteristic and Keystone Species

Creosote bush (*Larrea tridentata*) is the characteristic (*aka* defining) plant species of the Chihuahuan Desert. This aromatic shrub is often accompanied by tarbush (*Florensia cernua*), mesquite (*Prosopis glandulosa*), and acacias (*Acacia* spp.). Lechugilla (*Agave lechuguilla*) is the defining succulent species, often joined by yuccas (*Yucca* spp.) and cacti, especially prickly pear (*Opuntia* spp.).

Creosote is especially common on the bajada slopes and alluvial fans between mountain ranges and basin floors (Peters and Gibbens 2006). Mesquite is most common on sandy soils and often dominates sites with deep sands and a subsurface calcium carbonate layer (Peters and Gibbens 2006). Black grama grass (*Bouteloua eriopoda*) dominates grassland sites on sandy or gravelly upland sites, especially those with deep, loamy soils (Peters and Gibbens 2006). Tobosa grass (*Pleuraphis mutica*) typically dominates lowland sites with heavy, clayey soils and abundant water. Side oats grama (*Bouteloua curtipendula*) and alkali sacaton (*Sporobolus airoides*) grasses are also common on these sites (Peters and Gibbens 2006).

The Chihuahuan desert has few characteristic native mammal species, due to its relatively recent origin and open connection to neighboring ecoregions (Dinerstein et al. 2001). Native mammals occurring in the U.S. portion of the ecoregion include mule deer (*Odocoileus hemionus*), pronghorn (*Antilocapra Americana*), javelina (*Dicotyles tajacu*), kangaroo rats (*Dipodomys* spp.), woodrats (*Neotoma* spp.), and deer mice (*Peromyscus* spp.). Common bird species include greater roadrunner *Geococcyx californianus*, and scaled quail (*Callipepla squamata*). The now-rare aplomado falcon (*Falco ferorialis*) once roamed the region (Dinerstein et al. 2001).

The native herpetofauna of the Chihuahuan desert is more distinctive of the area than the native mammals and birds. Reptile diversity of the area is among the highest of any desert ecoregion (Dinerstein et al. 2001). Endemic lizards in the region include the Texas banded gecko (*Coleonyx brevis*), reticulated gecko (*C. reticulatus*), greater earless lizard (*Cophosaurus texanums*), and several species of spiny lizards (*Scheloporos* spp.) (Dinerstein et al. 2001). Amphibians strongly (but none exclusively) associated with the ecoregion include the Arizona toad (*Anaxyrus microscaphus*), Great Plains Narrowmouth Toad (*Gastrophryne olivacea*), Texas Toad (*Anaxyrus speciosus*), Barking Frog (*Craugastor augusti*), Rio Grande Chirping Frog (*Eleutherodactylus cystignathoides*), Spotted Chirping Frog (*Eleutherodactylus guttillatus*), Great Plains Narrowmouth Toad (*Gastrophryne olivacea*), Mountain or Arizona Treefrog (*Hyla wrightorum*), the Rio Grande, Plains, Chiricahua, and Northern Leopard Frogs (*Lithobates berlandieri*, *L. blairi*, *L. chiricahuaensis*, and *L. pipiens*, respectively), and Mexican Treefrog (*Smilisca baudinii*) (see Chapter 3).

The highly diverse fish assemblage of the Chihuahuan desert – including watersheds within the REA analysis extent – includes species adapted to cold-water mountain streams, such as the Rio Grande cutthroat trout (*Oncorhynchus clarki virginalis*) and the Gila trout (*Oncorhynchus gilae*); and species adapted to warm alluvial rivers, such as the freshwater drum (*Aplodinotus grunniens*), Rio Grande sucker (*Catostomus plebeius*), Plains killifish (*Fundulus zebrinus*), Rio Grande chub (*Gila pandora*), Rio Grande silvery minnow (*Hybognathus amarus*), and Rio Grande speckled chub (*Macrhybopsis aestivalis*). The assemblage also includes fishes adapted to springs with various unusual hydro-geo-chemical conditions and varying connections to rivers, such as the Big Bend gambusia (*Gambusia gaigei*), Pecos gambusia (*G. nobilis*), San Felipe gambusia (*G. clarkhubbsi*), Tex-Mex gambusia (*G. speciosa*), Leon Springs pupfish (*Cyprinodon bovinus*), Comanche Springs pupfish (*C. elegans*), and Pecos pupfish (*Cyprinodon pecosensis*); and one species adapted to the hydro-geo-chemically unique springs of the closed Tularosa Basin, the White Sands pupfish (*Cyprinodon tularosa*). Chapter 8 discusses the fishes of the northern Chihuahuan desert in detail.

The former keystone predatory mammals of the northern Chihuahuan desert are currently missing or greatly reduced in number. These included the Mexican gray wolf (*Canis lupus baileyi*), grizzly bear (*Ursus horribilis*), mountain lion (*Puma concolor*), coyote (*Canis latrans*), and badger (*Taxidea taxus*). Black-tailed prairie dogs (*Cynomys ludovicianus*), a keystone species in grasslands, are greatly reduced in range and abundance (Dinerstein et al. 2001) (see Chapter 15). Keystone subterranean termites (order Isoptera) continue to thrive in desert grasslands, where these invertebrates play vital roles in nutrient recycling (Whitford and Bestelmeyer 2006).

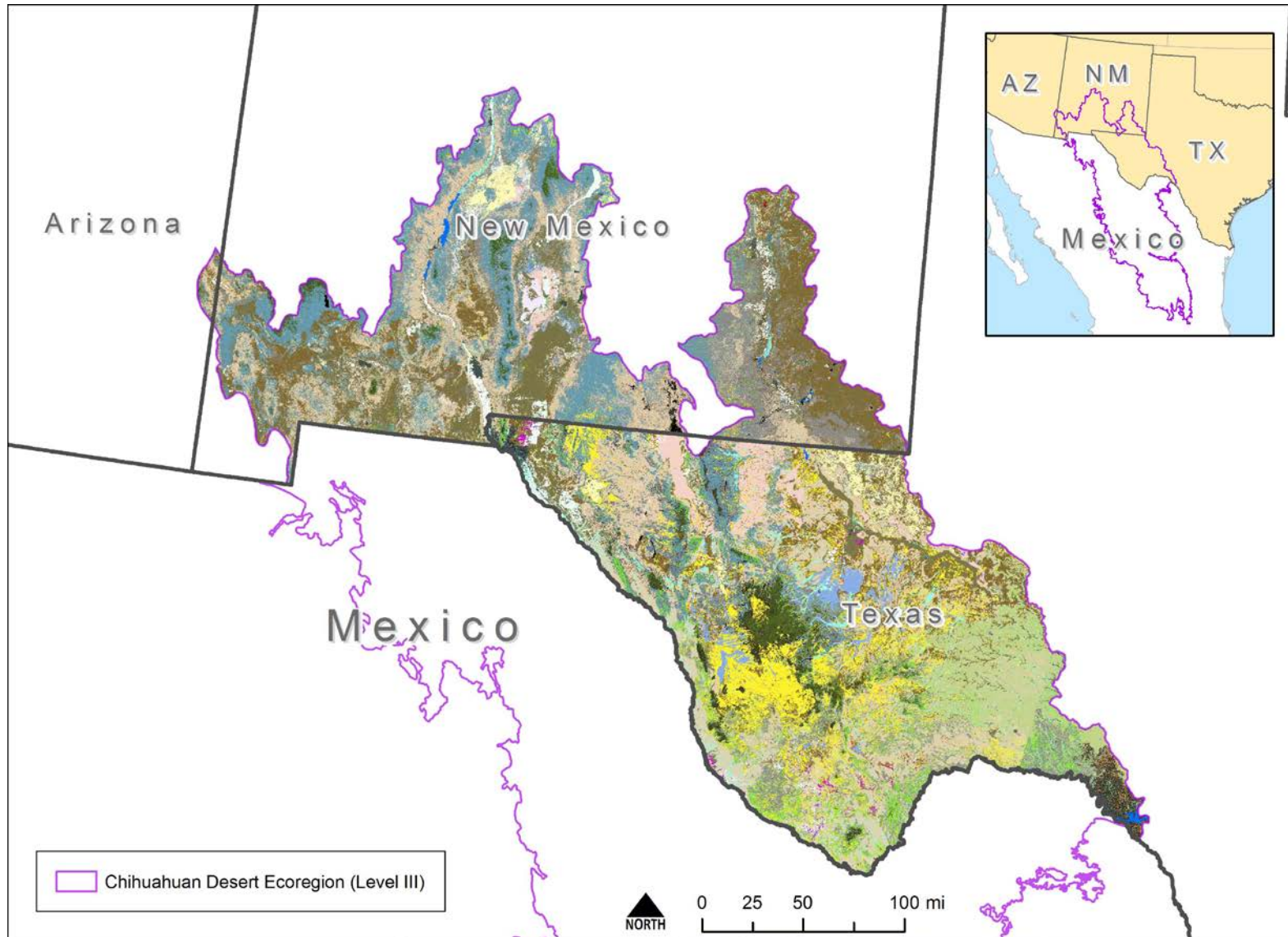
2.5.4 Ecological Systems

Table 2-1 lists the ecological systems recognized in the northern Chihuahuan desert in the U.S. and shows their spatial distribution (NatureServe 2009). The term, “ecological system” here refers to “... recurring groups of biological communities that are found in similar physical environments and are influenced by similar dynamic ecological processes, such as fire or flooding” (Comer et al. 2003). The three terrestrial ecological system Conservation Elements for the present REA (see Chapter 3) each include several of the specific ecological systems listed in Table 2-1, as explained in full in Chapters 5-7. The three stream and river ecological system Conservation Elements for the present REA (see Chapter 3) also each include individual ecological systems listed in this table, as explained in full in Chapters 8 and 9.

Table 2-1. Terrestrial ecological systems of the U.S. portion of the ecoregion (NatureServe 2009)

Forest & Woodland	Sparsely Vegetated
Madrean Encinal	North American Warm Desert Badland
Madrean Mesic Canyon Forest & Woodland	North American Warm Desert Pavement
Madrean Upper Montane Conifer-Oak Forest & Woodland	North American Warm Desert Volcanic Rockland
	Southwestern Great Plains Canyon
Rocky Mountain Bigtooth Maple Ravine Woodland	Edwards Plateau Limestone Savanna & Woodland
Upland Shrub	Madrean Lower Montane Pine-Oak Forest & Woodland
Chihuahuan Creosotebush Desert Scrub	Madrean Pinyon-Juniper Woodland
Chihuahuan Mixed Salt Desert Scrub	Rocky Mountain Aspen Forest & Woodland
Chihuahuan Succulent Desert Scrub Madrean Oriental Chaparral	Southern Rocky Mountain Ponderosa Pine Woodland
	Apacherian-Chihuahuan Mesquite Upland Scrub
Rocky Mountain Gambel Oak-Mixed Montane Shrubland	Chihuahuan Mixed Desert & Thornscrub
	Chihuahuan Stabilized Coppice Dune & Sand Flat Scrub
Tamaulipan Calcareous Thornscrub	Edwards Plateau Limestone Shrubland
Western Great Plains Sandhill Steppe	Mogollon Chaparral
Apacherian-Chihuahuan Semi-Desert Grassland & Steppe	Rocky Mountain Lower Montane-Foothill Shrubland
	Western Great Plains Mesquite Woodland & Shrubland
Llano Estacado Caprock Escarpment & Breaks Shrubland & Steppe	Savanna & Shrub-Steppe
	Chihuahuan Gypsophilous Grassland & Steppe
Southern Rocky Mountain Juniper Woodland & Savanna	Madrean Juniper Savanna
	Southern Rocky Mountain Ponderosa Pine Savanna
Upland Grassland & Herbaceous	Chihuahuan Loamy Plains Desert Grassland
Chihuahuan Sandy Plains Semi-Desert Grassland	Inter-Mountain Basins Semi-Desert Grassland
Southern Rocky Mountain Montane-Subalpine Grassland	Western Great Plains Foothill & Piedmont Grassland
	Woody Wetlands & Riparian
Western Great Plains Shortgrass Prairie	North American Warm Desert Riparian Mesquite
North American Warm Desert Lower Montane Riparian Woodland & Shrubland	Bosque
	North American Warm Desert Wash
North American Warm Desert Riparian Woodland & Shrubland	Rocky Mountain Subalpine-Montane Riparian Shrubland
Rocky Mountain Lower Montane-Foothill Riparian Woodland & Shrubland	Western Great Plains Floodplain
	North American Arid West Emergent Marsh
Rocky Mountain Subalpine-Montane Riparian Woodland	North American Warm Desert Interdunal Swale Wetland
Herbaceous Wetland	Western Great Plains Open Freshwater Depression Wetland
North American Warm Desert Ciénega	
Western Great Plains Closed Depression Wetland	Mixed Upland & Wetland
Western Great Plains Saline Depression Wetland	Western Great Plains Riparian
Chihuahuan-Sonoran Desert Bottomland & Swale Grassland	North American Warm Desert Active & Stabilized Dune
	North American Warm Desert Bedrock
	North American Warm Desert Playa
	Rocky Mountain Cliff, Canyon & Massive Bedrock
	Western Great Plains Cliff & Outcrop

Figure 2-6. Landcover types within the Level III boundaries. (NatureServe 2009)



Agriculture - Cultivated Crops and Irrigated Agriculture	Inter-Mountain Basins Montane Sagebrush Steppe	Orchards/Vineyards
Agriculture - General	Inter-Mountain Basins Semi-Desert Grassland	Recently Burned
Agriculture - Pasture/Hay	Inter-Mountain Basins Semi-Desert Shrub-Steppe	Rocky Mountain Alpine Bedrock and Scree
Apacherian-Chihuahuan Mesquite Upland Scrub	Inter-Mountain Basins Volcanic Rock and Cinder Land	Rocky Mountain Cliff, Canyon and Massive Bedrock
Apacherian-Chihuahuan Semi-Desert Grassland and Steppe	Introduced Riparian Vegetation	Rocky Mountain Gambel Oak-Mixed Montane Shrubland
Central Mixedgrass Prairie	Introduced Upland Vegetation - Annual Grassland	Rocky Mountain Lower Montane-Foothill Riparian Woodland and Shrubland
Chihuahuan Creosotebush Desert Scrub	Introduced Upland Vegetation - Perennial Grassland and Forbland	Rocky Mountain Lower Montane-Foothill Shrubland
Chihuahuan Gypsophilous Grassland and Steppe	Introduced Wetland Vegetation	Rocky Mountain Subalpine Dry-Mesic Spruce-Fir Forest and Woodland
Chihuahuan Loamy Plains Desert Grassland	Madrean Encinal	Rocky Mountain Subalpine-Montane Riparian Shrubland
Chihuahuan Mixed Desert and Thornscrub	Madrean Juniper Savanna	Ruderal Forest
Chihuahuan Mixed Salt Desert Scrub	Madrean Lower Montane Pine-Oak Forest and Woodland	Ruderal Upland - Old Field
Chihuahuan Sandy Plains Semi-Desert Grassland	Madrean Mesic Canyon Forest and Woodland	Sonora-Mojave Creosotebush-White Bursage Desert Scrub
Chihuahuan Stabilized Coppice Dune and Sand Flat Scrub	Madrean Oriental Chaparral	Sonora-Mojave Mixed Salt Desert Scrub
Chihuahuan Succulent Desert Scrub	Madrean Pinyon-Juniper Woodland	Sonoran Mid-Elevation Desert Scrub
Chihuahuan-Sonoran Desert Bottomland and Swale Grassland	Madrean Upper Montane Conifer-Oak Forest and Woodland	Sonoran Paloverde-Mixed Cacti Desert Scrub
Colorado Plateau Mixed Bedrock Canyon and Tableland	Mogollon Chaparral	Southern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest and Woodland
Colorado Plateau Pinyon-Juniper Woodland	Non-Specific Disturbed	Southern Rocky Mountain Juniper Woodland and Savanna
Developed-High Intensity	North American Arid West Emergent Marsh	Southern Rocky Mountain Mesic Montane Mixed Conifer Forest and Woodland
Developed-Low Intensity	North American Warm Desert Active and Stabilized Dune	Southern Rocky Mountain Montane-Subalpine Grassland
Developed-Medium Intensity	North American Warm Desert Badland	Southern Rocky Mountain Pinyon-Juniper Woodland
Developed-Open Space	North American Warm Desert Bedrock Cliff and Outcrop	Southern Rocky Mountain Ponderosa Pine Savanna
Edwards Plateau Cliff	North American Warm Desert Cienega	Southern Rocky Mountain Ponderosa Pine Woodland
Edwards Plateau Dry-Mesic Slope Forest and Woodland	North American Warm Desert Lower Montane Riparian Woodland and Shrubland	Southwestern Great Plains Canyon
Edwards Plateau Floodplain Terrace	North American Warm Desert Pavement	Tamaulipan Calcareous Thornscrub
Edwards Plateau Limestone Savanna and Woodland	North American Warm Desert Playa	Tamaulipan Mixed Deciduous Thornscrub
Edwards Plateau Limestone Shrubland	North American Warm Desert Riparian Mesquite Bosque	Western Great Plains Cliff and Outcrop
Edwards Plateau Riparian	North American Warm Desert Riparian Woodland and Shrubland	Western Great Plains Mesquite Woodland and Shrubland
Inter-Mountain Basins Active and Stabilized Dune	North American Warm Desert Volcanic Rockland	Western Great Plains Riparian
Inter-Mountain Basins Greasewood Flat	North American Warm Desert Wash	Western Great Plains Saline Depression Wetland
Inter-Mountain Basins Juniper Savanna	Northwestern Great Plains Riparian	Western Great Plains Sand Prairie
Inter-Mountain Basins Mixed Salt Desert Scrub	Open Water	Western Great Plains Sandhill Steppe
		Western Great Plains Shortgrass Prairie

2.6 Chihuahuan Desert Human Landscape

The U.S. portion of the Chihuahuan Desert ecoregion is a mostly sparsely populated area with historical roots in ranching, farming, mining, oil and gas production, and military testing and training. The details of these activities have changed over time, but the activities are still economically important in the region (Anderson and Gerber, 2008). Changes in these activities have led to shifts in land use and land cover. This section of the chapter summarizes the major features of the human landscape of the ecoregion in the U.S. that affect ecological resources.

2.6.1 Demography

The human population of the U.S. portion of the ecoregion increased by an estimated 38.4% from 1980 to 2000. However, outside of a few large cities, the land is still mostly sparsely populated grasslands and shrublands used mainly for livestock grazing (Ruhlman et al. 2012, Texas Land Trends 2015). Fewer than approximately 1.5 million people lived in the area in 2015. Sixty percent of these lived in the El Paso area (Texas Demographic Center 2015) and another 15 percent lived in the Las Cruces area (New Mexico Demographics 2016), the second largest city in New Mexico. The Borderplex Region of Las Cruces, New Mexico, El Paso, Texas, and Ciudad Juarez, Chihuahua, Mexico, is the seventh largest manufacturing area in North America (MVEDA undated). It has a combined population of approximately 2.5 million people, over 1.3 million of whom live in Mexico and share use of the Rio Grande and local aquifers with the U.S. population of the Borderplex (Hogan 2013, Borderplex Alliance 2016, TWDB 2016).

The five most populous (populations > 20,000) urban areas in the U.S. portion of the Chihuahuan Desert Ecoregion are El Paso, Texas, and Las Cruces, Roswell, Alamogordo, and Carlsbad, New Mexico. Smaller urban areas include Artesia, Socorro, and Truth or Consequences, New Mexico, and Fort Stockton, Texas. Populations in and around these cities are forecasted to grow. For example, El Paso County, Texas, is projected to grow from more than 670,000 people in 2000 to more than 1.14 million by 2040 (Borderplex Alliance 2016). In fact, the southwestern U.S. in general is expected to experience greater population growth than the rest of the U.S. (Theobald et al. 2013). New Mexico alone is expected to see an increase in population by another third by 2030 according to the Census Bureau's population predictions (Theobald et al. 2013). The population of the El Paso, Texas-Juarez, Mexico area is expected to increase significantly, driven by commerce stimulated by the North American Free Trade Agreement (NAFTA) (Anderson and Gerber 2008, Borderplex Alliance 2016, TWDB 2016). Population growth around the El Paso/Juarez area has resulted in declining air quality from the increasing numbers of vehicles and commercial sources of pollution (Anderson and Gerber 2008).

Several military installations in the region support nearby cities. Alamogordo and Las Cruces New Mexico are located near White Sands Missile Range and Holloman Air Force Base, and Fort Bliss is outside El Paso Texas. Although these installations contain within them large areas of relatively undisturbed land, they significantly affect adjacent development.

Except for Alamogordo, New Mexico, and Fort Stockton, Texas, the urban areas of the U.S. portion of the ecoregion all straddle or lie alongside rivers: Roswell, New Mexico, alongside the Rio Hondo;

Carlsbad and Artesia, New Mexico, alongside the Pecos River, and Socorro, Truth or Consequences, and Las Cruces, New Mexico, and El Paso, Texas, alongside the Rio Grande. Ciudad Juarez, Mexico, lies immediately across the Rio Grande from El Paso. These locations result in urban development of floodplains and implementation of measures to prevent flooding of developed lands. These trends of floodplain development around urban areas are expected to continue into the foreseeable future (Ruhlman et al. 2012, Theobald et al. 2013, Borderplex Alliance 2016).

2.6.2 Land Ownership

About 28 percent of the U.S. portion of the ecoregion is owned and managed by the federal government (Ruhlman et al. 2012), with the remainder almost entirely privately owned (NMDGF 2006). Most of the public lands are in New Mexico, the majority of which are managed by the Bureau of Land Management and the Department of Defense. The National Park Service manages two units in the New Mexico portion of the ecoregion: White Sands National Monument and Carlsbad Caverns National Park. The U.S. Fish and Wildlife Service operates three wildlife refuges in the New Mexico portion of the ecoregion, the Bosque del Apache, San Andres, and Bitter Lake refuges. New Mexico maintains 12 State Parks in the ecoregion, eight of which cluster along the Pecos and Rio Grande Rivers.

The Texas portion of the ecoregion is mostly private land, with several national and state parks. The National Park Service manages five units here (NPS CDIMN 2010), covering 385,000 ha. The largest are Big Bend and Guadalupe Mountains National Parks. Texas state parks in the area include the 120,000-hectare Big Bend Ranch State Park, the largest state park in Texas.

2.6.3 Land Use

Changes in land use and land cover have been noted across the U.S. portion of the ecoregion for over a century. Shrub invasion of grasslands over the past 50-150 years has been linked to altered fire regimes, inappropriate grazing, and climate change (Ruhlman et al. 2012) (see Chapter 5). Researchers warn that soil erosion on shrub-invaded areas might make it difficult to restore more desirable vegetation to these lands (Herrick et al. 2006).

Ruhlman et al. (2012) found that the effects of development on land cover in the U.S. portion of the ecoregion did not change much from 1973 to 2000. About 0.5% of land changed from one land cover type to another during this period (Ruhlman et al. 2012), with only four types of land cover changing by more than 100 km². These changes most commonly involved the conversion of grasslands and shrublands to mining and/or oil and gas production (217 km²) or to developed properties (187 km²), while some agricultural land reverted to grasslands or shrublands (158 km²; Ruhlman et al. 2012).

Most of the mapped impacts of oil and gas extraction on land cover occur in the eastern portion of the Chihuahuan desert in the U.S. (see below). Most of the mapped increase in residential and commercial development occurred near cities and Holloman Air Force Base. The conversion of grasslands and shrublands to mining and residential/commercial land is projected to continue into the future (Ruhlman et al., 2012).

Ranching, farming, and mining have deep roots in the U.S. portion of the ecoregion (NMDGF 2006, Texas

Land Trends 2015). These are still important to the region's economy, although the shape of these industries has changed with time. Livestock ranching spread north from New Spain and reached the U.S. portion of the ecoregion by A.D. 1600 (Havstad et al. 2006). Sheep vastly outnumbered cattle in the region until 1821 when Mexico became independent and Spanish settlements declined (Havstad et al. 2006). Cattle ranching exploded in the U.S. southwest after the U.S. Civil War. By the late 1800s, rangelands in the region were widely reported to be degraded by improper grazing (Havstad and Schlesinger 2006).

Oil and gas development has been a way of life across the Permian Basin since the late 1920s (see Geology, this Chapter), with a sharp increase during and following World War II. Production in the Permian Basin increased sharply again between 2009 and 2015, but stabilized in 2016 (USEIA 2016). The Basin, which straddles the New Mexico-Texas border, is the most prolific oil producing area in the U.S. (USEIA 2014). The increased and enhanced use of horizontal drilling and hydraulic fracturing (*aka* "fracking") in the most recent two decades has resulted in the extraction of oil and gas from so-called "tight" formations where it was previously unavailable (USEIA 2014). This most recent development is concentrated in a band from Fort Stockton to Pecos, Texas, and north into southeastern New Mexico (Texas General Land Office 2015, USEIA 2015). Impacts from oil and gas development include not only the development of drill sites and the road networks among them, but the construction and operation of waste disposal, pipelines, and pumping facilities.

Irrigation agriculture contributes significantly to the economy and patterns of land development in the U.S. portion of the ecoregion. Important crops in the area include nuts, onions, grains, cotton, vegetables, and fruits. However, farmers in the New Mexico portion of the ecoregion lately have shifted emphasis away from traditional crops to higher value crops such as pecans, pistachios, and chili peppers (SENMEDD/COG 2010). Irrigated farming is concentrated on former floodplains and largely depends on water diverted from surface streams and rivers. Additional areas of irrigation agriculture rely on groundwater (see Water Use, below). Large dairies are increasing in this area, especially in Chavez County, where the average herd has over 2100 cows (NASS 2013). Farming of floodplains requires the construction and maintenance of drainage systems to carry return flows and natural soil drainage back to surface water bodies, and especially along the Rio Grande also has required the construction of levees to protect the farms from rare but potentially harmful floods.

Timber harvesting is not a significant activity in the U.S. portion of the ecoregion. Timber is harvested in mountain ranges surrounding the assessment area, such as in the Sacramento and Mogollon Mountains. The numerous national and state parks in the ecoregion draw significant numbers of recreational visitors, as do some springs (see Water Use, below).

2.6.4 Water Use

Water use from both surface waters and aquifers in the U.S. portion of the ecoregion is highly regulated under irrigation district and state law, interstate compacts, and bi-national agreements (Hogan 2013). Two major dams (Elephant Butte and Caballo) and six smaller diversion dams (San Acacia, Leasburg, Mesilla, American, International, and Riverside) on the Rio Grande within the ecoregion store and divert water for municipal use, irrigated agriculture, hydroelectric power generation, and to meet treaty

obligations for the delivery of water to Mexico (U.S. Bureau of Reclamation 2011, Ruhlman et al. 2012, Hogan 2013). Beyond the ecoregional boundaries, ten other dams lie upstream on the Rio Grande, and the international Amistad and Falcon dams lie downstream. Nearly 600 miles of canals and laterals, and over 450 miles of drains support extensive agriculture along the Rio Grande in south-central New Mexico and west Texas (U.S. Bureau of Reclamation 2011, Hogan 2013, NMOSE 2016a).

Three moderate-size dams on the Pecos River – Brantley, Avalon, and Red Bluff – store and divert water for irrigated agriculture and, in the case of Red Bluff, generate hydroelectric power. Beyond the ecoregional boundaries, two other dams lie upstream on the Pecos River and ten lie upstream on the Rio Grande in New Mexico and Colorado. The international Amistad Reservoir, behind Amistad Dam on the Rio Grande, inundates the confluence of the Pecos River with the Rio Grande. Seven dams regulate the flow of the Río Conchos and three of its major tributaries (Kelly 2001).

The Gila River presently has no dams along its mainstem, and diversions along the mainstem within the U.S. portion of the ecoregion deliver water only to local users. However, efforts are ongoing to permit construction of a large diversion facility somewhere along the mainstem immediately upstream from the present analysis extent, under the terms of the New Mexico-Arizona Water Settlement Act of 2004 (New Mexico Interstate Stream Commission 2017).

Most of the waters of the Pecos River, Rio Grande, and lower reaches of their perennial tributaries are diverted for use by municipalities and irrigation districts in the U.S. and in Mexico (Hoyt 2002, Hogan 2013). The impoundments also inundate large sections of floodplain, while also trapping almost all of the sediment that the rivers formerly carried past their locations – sediment historically crucial to habitat dynamics within the rivers and across their floodplains. Further, the combination of diversions, consumption of the diverted water, and operation of the impoundments significantly alter river hydrology and connectivity. The Rio Grande sometimes runs dry for some distance below Elephant Butte Dam and again below El Paso as a result of upstream water consumption and impoundment (Hogan 2013). Return flows from agricultural and municipal water uses carry heavy loads of dissolved salts (Hogan 2013, IBWC 2013). River regulation, dams, diversions, and return flows with degraded water quality have contributed to changes in native fish populations and floodplain forests and wetlands (see Chapters 8-9).

Farmers, municipalities, and some industries also heavily use groundwater in the ecoregion. Some aquifers lie under multiple jurisdictions, receive recharge from the Rio Grande, or discharge (or formerly discharged) to the Rio Grande or Pecos River, resulting in jurisdictional conflicts (Hogan 2013). For example, Texas filed a suit in 2013 against New Mexico concerning pumping along the Mesilla (Rio Grande) Valley between Radium Springs, New Mexico, and El Paso, Texas, that appears likely go to the U.S. Supreme Court. The suit has the potential to dramatically limit groundwater pumping for New Mexico agriculture in the valley (Bryan 2016). Groundwater pumping for irrigation is also common along the Pecos River valley between Roswell and Artesia, New Mexico; around Deming and Nutt; around the margins of the Salt Basin near Dell, Texas; between Balmorhea and Pecos, Texas; and around Stockton, Texas. Texas and New Mexico have resorted to the courts to resolve disputes about this pumping as well (NMOSE 2016). Groundwater extraction from basin fill and alluvial aquifers along both the Rio Grande

and Pecos River has reduced the flow of water from springs and lowered floodplain water tables, which can negatively affect floodplain and emergent wetlands, endemic fish, and invertebrate species. Some of the groundwater in the U.S. portion of the ecoregion also is brackish (e.g., Huff and Chace 2006, George et al. 2011, Meyer et al. 2012), and its use results in salt deposition at the ground surface and/or releases of brackish wastewater into the surface water system.

The oil and gas development in the Permian Basin between 2009 and 2015, noted above, involved a massive expansion in the use of hydraulic fracturing to force open geologic formations to permit the escape of the oil or gas, as noted above. This practice requires large volumes of water, only some of which can be recycled following use. It also poses risks of water pollution from flowback, well leakage, and waste spills, although these risks are subject to significant regulation (NMOGA 2012, NMEMNRD 2016). Hydraulic fracturing has been used in oil and gas extraction in the ecoregion for many decades (NMOGA 2012), but its use is expanding as a result of the more recent coupling of hydraulic fracturing with horizontal drilling technologies (USEIA 2015, NMEMNRD 2016). The USGS estimates that water use for oil and gas extraction accounted for the largest increase in water use in New Mexico between 2005 and 2010 (Maupin et al. 2014). However, changes in technology have reduced the amount of water needed, including both fresh and brackish water (NMEMNRD 2016, New Mexico Energy Forum 2016).

Finally, many springs in the ecoregion, such as the Balmorhea Springs complex in Texas, have been developed for recreational use. While not resulting in water consumption, such recreational development typically eliminates wetland habitats and significantly alters aquatic habitat conditions.

3 Conservation Elements, Change Agents, and Management Questions

The Pre-Assessment Phase (*aka* Phase I) for the Chihuahuan Desert Rapid Ecoregional Assessment (REA) focuses on (1) identifying the Conservation Elements, Change Agents, and Management Questions on which to focus the assessment, and (2) developing conceptual ecological models for the Conservation Elements. The conceptual models show how the Change Agents may affect each Conservation Element, and provide a means for translating Management Questions into terms specific to each individual Conservation Element and/or Change Agent. This chapter describes the process followed in the Chihuahuan Desert REA to identify the Conservation Elements, Change Agents, and Management Questions on which to focus the assessment.

3.1 Conservation Elements

REAs do not attempt to assess all ecological values in an ecoregion—an impossible task, as noted in Chapter 2. Instead, REAs focus on a limited set of key resources, termed *Conservation Elements* (CEs), consisting of regionally-significant terrestrial and aquatic species and ecological systems of management concern. The Assessment Management Team (AMT) for the Chihuahuan Desert REA identified the CEs for this REA in cooperation with a Technical Team through discussions that also considered the most pressing Change Agents (see below) for the U.S. portion of the ecoregion and the ecological resources they affect.

The Chihuahuan Desert REA focuses on fourteen (14) CEs, listed in Table 3-1. These consist of three terrestrial or “dry” ecological system types, five “wet” ecological system types, and four individual species and two assemblages of species of management concern associated with terrestrial ecological systems. One of the wet system types, “Playas and Playa Lakes,” has both dry and wet (inundated) phases, and thus shares features with both wet and dry system types.

Table 3-1. Chihuahuan Desert REA Conservation Elements

Conservation Element Group	Conservation Element Name
Dry System Types	<ul style="list-style-type: none">• Chihuahuan Desert Grasslands• Chihuahuan Desert Scrub• Pinyon-Juniper Woodlands
Wet System Types	<ul style="list-style-type: none">• Montane-Headwater Perennial Streams• Lowland-Headwater Perennial Streams• Large River-Floodplain Systems• Springs-Emergent Wetlands• Playas and Playa Lakes

Species and Assemblages	<ul style="list-style-type: none"> • Pronghorn • Mule Deer • Banner-tailed Kangaroo Rat • Black-tailed Prairie Dog • Grassland Bird Assemblage • Grassland Small Mammal Assemblage
-------------------------	------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------

The term, “ecological system” here refers to “... recurring groups of biological communities that are found in similar physical environments and are influenced by similar dynamic ecological processes, such as fire or flooding” (Comer et al. 2003). The following paragraphs provide brief descriptions of the fourteen CEs for the Chihuahuan Desert REA. Chapters 5-17 provide detailed descriptions with citations (Chapter 8 addresses the two perennial stream system types together).

3.1.1 Dry System Types

The three “dry system types” selected as CEs for the Chihuahuan Desert REA consist of three groups of similar, related terrestrial ecological system types that occur across the U.S. portion of the Chihuahuan Desert ecoregion. Together, these three CEs cover 84% of the lands in the U.S. portion of the ecoregion (Chihuahuan Grasslands – 22%, Chihuahuan Scrub – 59%, and Pinyon-Juniper Woodlands – 3%). These three groups are drawn from the master list of terrestrial ecological system of the ecoregion, presented in Table 2-1 in Chapter 2. Chapters 5-7 discuss the individual ecological system types included within each of the three dry system CEs for the Chihuahuan Desert REA. Other terrestrial ecological system types occurring within the U.S. portion of the ecoregion are not included in the CEs for this REA. For example, several woodland ecological system types occur at higher elevations in the ecoregion, but do not occur on lands managed by the BLM within the ecoregion.

The “Chihuahuan Desert Grasslands” CE (Chapter 5) occurs on four major landforms in the U.S. portion of the ecoregion: piedmont on coalesced alluvial fans, foothills on colluvium, lowlands on basins and playas, and sandy plains on sand sheets. These desert grasslands typically occur between 1,100 and 1,700 m in elevation. Although dominated by grasses, occurrences of this CE may also include shrubs or sub-shrubs as natural parts of the plant community. Chihuahuan desert grasslands can grow on a range of soil types ranging from clayey to rocky. Fire, grazing, and drought are common natural disturbances shaping plant community composition in these grasslands. The ecological stressors of extreme drought, inappropriate livestock grazing, human use, changed fire regime, and climate change allow desert scrub plants and some exotic plant species to invade and dominate areas historically dominated by grasslands.

The “Chihuahuan Desert Scrub” CE (Chapter 6) occurs across the lower elevations of the U.S. portion of the ecoregion on multiple landforms from basin floors to piedmont alluvial fans and foothills. Much of the desert scrub occurs over limestone parent material. Fire and drought are common sources of natural disturbance although fire has less influence in some settings than others. The most common dominant plant species of the Chihuahuan desert scrub is creosotebush (*Larrea tridentate*), which often occurs with tarbush (*Flourensia cernua*). Other potentially dominant shrubs include whitethorn acacia *Acacia constricta*, viscid acacia (*Acacia neovernicosa*), Rio Grande saddlebush (*Mortonia scabrella*), and ocotillo (*Fouquieria splendens*).

The “Pinyon-Juniper Woodlands” CE (Chapter 7) occurs on a variety of landforms across the U.S. portion of the ecoregion, including basins, hills, and slopes on a variety of soils at moderate elevations between 1,400 and 2,200 m in elevation. They are often bordered by grasslands at the lower elevations. Juniper is often more common than pinyon at lower elevations. However, in southern New Mexico, juniper is more common than pinyon at higher elevations, as well likely because of greater summer precipitation in the these areaa. Common tree species in this system type include Mexican pinyon (*Pinus cembroides*), border pinyon (*Pinus discolor*), two-needle pinyon (*Pinus edulis*), alligator juniper (*Juniperus deppeana*), one-seed juniper (*Juniperus monosperma*), redberry juniper (*Juniperus coahuilensis*), and Pinchot's juniper (*Juniperus pinchotii*). The most influential natural disturbances that modify juniper-pinyon community structure are climate variation, fire, and insect infestations.

3.1.2 Wet System Types

The five “wet system types” identified as CEs for the Chihuahuan Desert REA consist of groups of aquatic, wetland and other related ecological system types that occur across the lands managed by the BLM in the U.S. portion of the Chihuahuan Desert ecoregion. The aquatic, wetland and other related ecological system types assigned to each of these five groups consist of types or sub-divisions of types drawn from the master list presented in Table 2-1 in Chapter 2. Chapters 8-11 discuss the individual ecological system types and sub-divisions of these system types included within each wet system CE. Other seasonal and perennial wetland ecological system types occur within the U.S. portion of the ecoregion but are not included in this REA. For example, several seasonal and perennial wetland ecological system types occur at higher elevations in the ecoregion, but not on lands managed by the BLM. Seasonal depressional wetland types other than playas that occur within the dry system CEs are addressed as components of these systems rather than as part of any wet system type.

The Chihuahuan Desert REA distinguishes two types of perennial streams based on the sources of their waters and the geological conditions that characterize these sources (Chapter 8). “Montane-Headwater Perennial Streams” originate at higher-elevation, montane settings. The elevation of these settings results in higher rates of precipitation than occur at lower elevations across the surrounding valley floors, with some of the precipitation occurring as snowfall. Streams that originate in these settings receive their water as runoff from both rainfall and snowmelt, as groundwater drainage from shallow montane soils and montane bedrock aquifers, and at discrete tributary springs. Cooler air temperatures, cold-air drainage along stream valleys, and montane riparian vegetation canopies help maintain relatively cool water temperatures. However, water temperatures vary with the time of day, season, and hydrologic conditions. The montane topographic settings result in steeper stream gradients and higher flow velocities on average, than found in streams with comparable discharge in lowland settings. Because of their steeper gradients, montane-headwater perennial stream waters spend relatively little time in contact with the vegetative litter, soils, and bedrock of the montane setting, and so contain relatively low concentrations of dissolved matter. The montane topographic settings also result in relatively narrow riparian vegetation corridors. Natural disturbances include riparian fire, which may originate in the surrounding uplands, and floods and droughts. Many montane-headwater perennial streams in the ecoregion flow out onto valley floors, where they may develop wider floodplains and where groundwater discharge, evapotranspiration, and infiltration further alter their flow, temperature,

and chemistry regimes. The combined effects of evapotranspiration and infiltration at lower elevations may cause flow to become seasonal or otherwise intermittent rather than perennial. However, flooding along large river-floodplain systems downstream can force floodwaters upstream along the lower reaches of tributaries.

“Lowland-Headwater Perennial Streams,” in contrast, originate around the bases of mountains or in surrounding valleys. Streams that originate in these settings receive their water primarily from discharges of groundwater—sometimes at discrete springs—from lower-montane bedrock, basin-fill, and other larger-scale aquifers. The groundwater discharged into these streams originates as recharge at higher elevations, but may spend years, decades, or longer moving through the groundwater system before re-emerging. As a result, the water in each stream emerges with a distinct but relatively constant temperature year-round, controlled by the temperatures in the aquifers through which the water has passed, some of which may be affected by geothermal activity. The water in each resulting stream also emerges with a distinct pattern of concentration of dissolved matter, controlled by the chemistry of the groundwater pathways along which the water has traveled. Finally, because of their geological and topographic settings, lowland-headwater streams have relatively low gradients with relatively constant rates of baseflow year-round. However, the combined effects of evapotranspiration and infiltration downstream from the stream sources may cause flow to become seasonal or otherwise intermittent rather than perennial. Natural disturbances to these streams include riparian fire, which may originate in the surrounding uplands. Short-term hydrologic disturbance is rare because of the low contribution of runoff to the hydrologic regime. However, again, flooding along large river-floodplain systems downstream can force floodwaters upstream along the lower reaches of tributaries.

The “Large River-Floodplain Systems” CE consists of the three largest rivers in the U.S. portion of the ecoregion, the Gila River, Rio Grande, and Pecos River (Chapter 9). This aquatic-wetland CE type contrasts with both types of perennial stream types in several ways. Most influentially, these three rivers receive their greatest headwater inputs almost entirely outside this ecoregion. The Gila River originates in the Mogollon Mountains to the north and northwest of the ecoregional boundary, and the Pecos River originates in the southern Rocky Mountains to the north. The Rio Grande originates both in the southern Rocky Mountains to the north and in the mountains of the Rio Conchos basin to the southwest, in Mexico, the latter of which joins the Rio Grande just upstream from the Big Bend of the river. The external, mountainous sources of these three large rivers produce greater annual discharges of water and transported matter and different seasonal patterns of discharge than would occur if these rivers originated entirely within the U.S. portion of the ecoregion. Natural short-term disturbances included riparian fire, which may originate in the surrounding uplands, droughts, and inundation and sediment erosion and deposition by floods.

The Rio Grande and Pecos River today are fragmented, strongly regulated by dams, and greatly diminished by diversions, with many of these alterations taking place both inside and outside the U.S. portion of the ecoregion. Prior to their regulation, however, the flows of water and sediment along these two rivers maintained more complex channels, much larger and more geologically active floodplains with extensive wetlands within the U.S. portion of the ecoregion, and much larger alluvial aquifers than associated with any montane- or lowland-headwater stream in the ecoregion. Their longer

flow distances also resulted in higher water temperatures and higher concentrations of dissolved matter, both conditions exacerbated by river regulation. Historically, these two rivers – and the Gila River mostly to the west of the ecoregion – supported fauna and flora adapted to large, warm-water river settings, active river-floodplain exchanges of water and nutrients, flood cycles and disturbances, and extensive riparian wetland and woodland communities. Further, their riparian wetlands, particularly along the Rio Grande and Pecos River, provided – and in some areas still provide – substantial areas of stopover or over-wintering habitat for numerous migratory bird species, some in very large numbers.

“Springs-Emergent Wetlands” in the U.S. portion of the ecoregion (Chapter 10) are known by a variety of names, including ciénega, marsh, spring complex, and cenote, and occur across a wide range of elevations. They depend on discharges of groundwater from alluvial or other shallow aquifers, montane bedrock, basin-fill, or larger-scale or regional aquifers. Their waters therefore may originate as recharge nearby or elsewhere and may spend years, decades, or longer moving through the groundwater system before re-emerging. As a result of this variety in their hydrogeology, each spring in the ecoregion has a distinctive pattern of discharge or water level; a distinctive pattern of temperature controlled by the temperatures in the aquifers through which the water has passed, some of which may be affected by geothermal activity; and a distinctive water chemistry controlled by the chemistry of the groundwater pathways along which its water has traveled. Some springs in the ecoregion discharge into and may even constitute the dominant source(s) of water for a perennial stream, while others may only support a localized wetland unconnected to the regional surface drainage network. Spring-emergent wetland sites connected to the regional surface drainage network of the region may share species across the larger network, even if such connections occur only intermittently over historic or geologic time scales. The connections allow aquatic species to move among spring-emergent wetland sites or between springs and streams: the more isolated a site, the more likely the site will come to harbor unique, endemic species. Short-term hydrologic disturbance at spring-emergent wetland sites is rare because of the low contribution of runoff to the hydrologic regime, although river flooding downstream from a spring outlet can cause water to back up into a spring. Other natural disturbances include fire in the emergent or surrounding vegetation, which may originate in surrounding uplands. Data on potential criteria (e.g., hydrogeology, morphology, discharge rates, chemistry, fauna, and flora) for distinguishing spring-emergent wetland sub-types are available for only a few sites across the ecoregion. Consequently, the present assessment does not distinguish any sub-types.

Thousands of “Playas and Playa Lakes” (Chapter 11) occur throughout the U.S. portion of the ecoregion. These consist of barren and sparsely vegetated depressions in desert basins, tectonic lows, interdunal flats, wadis, and abandoned channels that experience seasonal or episodic wetting. The larger playa and playa lake sites in the U.S. portion of the ecoregion, such as Alkali Flat and Lake Lucero, Lordsburg Playa, Isaacks Lake, and Playas Lake in New Mexico, are remnants of Pleistocene lakes. Wetting today comes from runoff following seasonal or episodic storms, supported by a rise in the local water table following such storms. Annual variation in precipitation strongly affects the inundation regime of playas. Some may fill and dry multiple times per year while others may remain dry for years. Intermittent flooding followed by evaporation often leaves behind an alkaline salt residue, which in turn strongly affects the vegetation. Small saltgrass beds often form in playa depressions, while sparse shrubs and phreatophytes

may occur around the margins. This vegetation has been classified as alkaline scrub and includes many halophytic species. Dune fields often form downwind of large playas in windswept desert basins, which may also have deeper water supplies. The largest and best-studied such dune field in the ecoregion – indeed, one of the best-studied playa basin dune fields in the world – is found at White Sands National Monument (WSNM) in New Mexico. The vegetation surrounding playas and playa lakes in the ecoregion, outside of any associated dune fields, typically consists of grasslands. Playas in the ecoregion vary in size, soil texture/geologic origin, and the presence of vegetation. Many playa lake sites in the U.S. portion of the ecoregion are important migratory stopover points for a diversity of shorebirds and other waterbirds, and many also provide habitat for unique assemblages of freshwater invertebrates and vertebrates including various branchiopods (clam shrimp, fairy shrimp), beetles and a number of anuran species adapted to the unreliable habitat conditions.

The AMT initially proposed a sixth wet system type as a CE, “Gypsum Systems,” consisting of playas, playa lakes, dunes, and inter-dunal wetted areas in localities affected by gypsiferous geology, geochemistry, and soils. As discussed in Chapter 2, these localities have unique ecological characteristics and dynamics. However, the AMT proposed including this sixth wet system type as a CE out of recognition that gypsiferous geology, geochemistry, and soils affect both wetland *and terrestrial* ecological dynamics across large areas of the ecoregion, because of the unique geology of the region (see Chapter 2). Unfortunately, designating “Gypsum Systems” as a sixth wet system CE did not address all management concerns stemming from this recognition. These concerns focus on the unique biota of gypsiferous terrestrial, wetland, and aquatic settings in the ecoregion, and their unique potential sensitivities to Change Agents. Additionally, a review of the ecoregional data on land cover types revealed that the analyses expected to show the distribution of gypsiferous ecological conditions do not do so. That is, they do not consistently capture all areas where gypsiferous geology, geochemistry, and soils affect terrestrial or wetland or aquatic ecological conditions. The relevant land cover types include the Chihuahuan Gypsophilous Grassland and Steppe and adjacent areas of North American Warm Desert Active and Stabilized Dune, Western Great Plains Saline Depression Wetland, and North American Warm Desert Playa cover types (see Table 2-1, Chapter 2).

The literature on the ecoregion includes numerous studies and reports on the gypsiferous bedrock geology, soils, playas, dunes, aquifers, karst regions, and biota of the ecoregion, for comparison (see the sections on Chihuahuan Desert Biophysical Setting and Chihuahuan Desert Biodiversity in Chapter 2). These considerations led the Technical Team to propose – and the AMT to approve – addressing management concerns related to the gypsiferous lands, wetlands, and waters of the ecoregion instead through a Management Question applied to all CEs (see Management Question #13, below). This Management Question takes advantage of the consistent availability of data on the distribution of potentially gypsiferous bedrock formations across the ecoregion. These data make it possible to assess (a) the distribution of potentially gypsiferous geologic, geochemical, and soil conditions across the ecoregion, and (b) the distributions of CEs and Change Agents relative to the distribution of these conditions.

3.1.3 Species and Species Assemblages

The four individual species and two assemblages of species selected as CEs for the Chihuahuan Desert REA either depend on or significantly affect the landscape-scale ecological integrity of grasslands across the U.S. portion of the ecoregion.

Pronghorn (*Antilocapra Americana*) (Chapter 12) is a wide-ranging ungulate herbivore. Its overall geographic range extends well beyond the boundaries of the Chihuahuan desert ecoregion, but it uses and moves among several natural communities within the U.S. portion of the ecoregion. As a result, the distribution and status of pronghorn in the ecoregion provides an indicator of the overall ecological condition of the landscape. Pronghorn travel in herds and are highly visible because they occupy open habitat consisting primarily of flat prairies, shrub steppes, and semiarid grasslands. They avoid mountainous terrain. They feed preferentially on low vegetation, primarily on forbs and small shrubs rather than grass, but require taller vegetation as cover for fawns that are nearly immobile shortly after birth. Competition between pronghorn and other native ungulates appears to be minimal, although there is dietary overlap with mule deer (*Odocoileus hemionus*). Pronghorn is an important prey species for several of the native large predators, and is an important game species, providing economic benefits to landowners and area commerce. Some taxonomists distinguish five subspecies of pronghorn, including (*Antilocapra americana Mexicana*) in the Chihuahuan Desert. However, other taxonomists disagree about the validity of these subspecies, which genetic evidence suggests alternatively may be clines rather than subspecies. The present assessment does not distinguish subspecies.

Mule deer (*Odocoileus hemionus*) (Chapter 13) also is a wide-ranging ungulate herbivore. Its overall geographic range also extends well beyond the boundaries of the Chihuahuan desert ecoregion, but it also uses and moves among several natural communities within the ecoregion. As a result, the distribution and status of mule deer in the ecoregion also provides an indicator of the overall ecological condition of the landscape. Mule deer occupy a variety of habitats across their overall range, including agricultural lands, forests, grasslands, savannas and shrublands. In much of their range, they migrate from high elevations in the summer to lower elevations in winter. They require adequate and available foraging opportunities, access to water, including water from forage, good visibility and terrain allowing for movement for foraging, safe habitat selection and to avoid predation. In the U.S. Southwest, mule deer occur in desert shrublands, semi-desert shrubland-grasslands, chaparral, mountain shrublands and woodlands and forests at higher elevations. Additionally, washes are important for water, food, escape, and resting, and provide corridors for travel. In the Chihuahuan desert they browse primarily on shrubs and forbs and consume very little grass. As with pronghorn, mule deer is an important prey species for several of the native large predators; and is an important game species. Deer hunting provides economic benefits to landowners and area commerce. Some taxonomists classify the mule deer occupying the southwestern deserts, including the Chihuahuan Desert, as (*Odocoileus hemionus eremicus*).

The banner-tailed kangaroo rat (*Dipodomys spectabilis*) (Chapter 14) is a nocturnal, granivorous heteromyid rodent found throughout the grasslands of the Chihuahuan desert. It can be locally common, but is threatened by widespread degradation of its desert grassland habitat throughout much of its range. It is a mound-building rodent, and this ecological engineering can dramatically affect the

community structure of both grassland plants and associated animals within the footprint of past and active mounds. Consequently, the banner-tailed kangaroo rat is considered a keystone species for the grasslands of the ecoregion, one with very specific habitat requirements. Its distribution and abundance provide important indicators of the overall ecological condition of the grasslands of the ecoregion.

The black-tailed prairie dog (*Cynomys ludovicianus*) (Chapter 15) is a colonial, burrowing rodent that inhabits several types of open grassland habitats from the Great Plains to the deserts of northern Mexico. Their burrowing alters the structure and composition of the grasslands they occupy, creating and maintaining suitable habitat for many other species, making the black-tailed prairie dog another keystone species in the grasslands of the Chihuahuan desert ecoregion. Many of the ecoregion's predators feed on these rodents, including the federally endangered black-footed ferret (*Mustela nigripes*), an obligate predator of prairie dogs. (The black-footed ferret formerly occupied but no longer occurs anywhere in the U.S. portion of the Chihuahuan desert ecoregion; <http://ecos.fws.gov/ecp0/profile/speciesProfile?slid=6953>). Estimates suggest that the abundance of the black-tailed prairie dog has fallen by more than 90 % from historic levels, particularly as a result of efforts eliminate its burrowing from grazing lands. Despite the dramatic decline in black-tailed prairie dog in the 20th century, public eradication programs have continued and conservation initiatives have struggled to strike a balance with agricultural interests. Its distribution and abundance also provide important indicators of the overall ecological condition of the grasslands of the ecoregion.

Grassland birds (Chapter 16) as a group have experienced the steepest population decline of any group of North American avifauna. This trend is evident among many bird species that are endemic, or near-endemic, to the grasslands of the Chihuahuan desert. The "Grassland Bird Assemblage" consists of the Arizona grasshopper sparrow (*Ammodramus savannarum ammoregus*), Baird's sparrow (*Ammodramus bairdii*), chestnut-collared longspur (*Calcarius ornatus*), Cassin's sparrow (*Peucaea cassinii*), and scaled quail (*Callipepla squamata*). These five grassland-obligate bird species have similar ecological requirements, and together provide another important indicator of the overall ecological condition of the grasslands of the ecoregion.

The Chihuahuan desert supports a large number of small rodent species. These animals are important components of the desert ecosystem because of their effects on ecosystem process and properties. They are important consumers of plants, seeds, and invertebrates and are often considered to be ecological engineers; and provide food for many predators. Healthy populations of these small mammals are essential for a healthy desert ecosystem. The "Grassland Small Mammal Assemblage" (Chapter 17) consists of the deer mouse (*Peromyscus maniculatus*), southern plains woodrat (*Neotoma micropus*), hispid cottonrat (*Sigmodon hispidus*), tawny bellied cottonrat (*Sigmodon fulviventor*), and yellow-nosed cottonrat (*Sigmodon ochrognathus*). These five species live in a variety of habitats and feed on a variety of organisms, but all can be found in the Chihuahuan desert grasslands and all share grasslands, or a component of grasslands, as critical habitat. Together, they provide another important indicator of the overall ecological condition of the grasslands of the ecoregion.

The Technical Team debated whether to treat the grassland bird and grassland small mammal assemblages as separate CEs, or alternatively to treat their distributions and abundances simply as

indicators of the condition of the Chihuahuan Desert Grasslands CE. The Technical Team recommended, and the AMT agreed, to treat the assemblages as separate CEs, because they are important in the ecoregion as management concerns by themselves, not merely as indicators of grassland condition.

The AMT initially also proposed two assemblages of fish species as additional CEs: a “Gila River Fish Assemblage,” a “Pecos River Fish Assemblage,” and a “Chihuahuan Desert Amphibian Assemblage.” The two fish assemblages consist of endemic species with specialized adaptations to the unique hydrology, water chemistry, and fluvial geomorphology of their respective hydrologic systems. Specifically, the Gila River fish assemblage includes the Gila trout (*Oncorhynchus gilae*), loach minnow (*Tiaroga (Rhinichthys) cobitis*), and spikedace (*Meda fulgida*), and several other species. These species occur in the mainstem and headwaters of the Gila River within the ecoregion. Some members of the assemblage also occur in the Mimbres River, an endorheic river with headwaters adjacent to several southeastern tributary headwaters of the Gila River. The Pecos River fish assemblage occurs in both the mainstem and headwaters of the Pecos River within the ecoregion, including perennially and intermittently tributary springs. It includes the gray redhorse (*Moxostoma congestum*), Bluntnose shiner (*Notropis simus*), and Pecos pupfish (*Cyprinodon pecosensis*), as well as several other species. Some members of the Pecos River fish assemblage also occur in the Rio Grande, into which the Pecos River flows. The AMT subsequently agreed to a recommendation from the Technical Team to treat the two fish assemblages instead as indicators of the condition of the wet system types with which they are associated, to avoid redundancy. Chapter 8 provides a detailed review of the fishes of the ecoregion and their distribution among the wet system types defined for the REA.

The AMT initially also considered including a “Chihuahuan Desert Amphibian Assemblage” as an additional CE. The Technical Team reviewed the literature on the amphibians of the ecoregion, to provide guidance. Table 3-2 Table 3-2. Amphibians of the U.S. portions of the Chihuahuan desert ecoregion. lists the 25 amphibians recognized as occurring, or potentially occurring, within the U.S. portion of the ecoregion (El-Hage and Moulton 1998, Bell et al. 1999, Dinerstein et al. 2001, Pronatura Noroeste 2004, NMDGF 2006, WWF-SIA 2007, Misztal et al. 2013, NatureServe 2015). Table 3-2 also provides notes on the habitat associations of these species, their Global and State Heritage ranks (NatureServe 2015) and their classification as a State Comprehensive Wildlife Conservation Strategy, Species of Greatest Conservation Need (SGCN) (NMDGF 2016, TPWD 2016).¹

As with the two fish assemblages, the AMT approved the recommendation of the Technical Team to treat a Chihuahuan desert amphibian assemblage not as a separate CE. Specifically, based on the findings presented in Table 3-2, the AMT approved the recommendation of the Technical Team to recognize a Chihuahuan desert amphibian assemblage consisting of ten (10) species that occur in the U.S. portions of the ecoregion, use river-, stream-side, and spring wetlands, and are ranked as Globally or State rare (G1-G3 or S1-S3) or identified as SGCNs in the New Mexico or Texas State Comprehensive

¹ Texas and New Mexico State Comprehensive Wildlife Conservation Strategy (CWCS), Species of Greatest Conservation Need (SGCN) listings per Texas Parks and Wildlife Department and New Mexico Department of Game and Fish (respectively http://tpwd.texas.gov/huntwild/wild/wildlife_diversity/nongame/tcap/sgcn.phtml and <http://www.wildlife.state.nm.us/conservation/comprehensive-wildlife-conservation-strategy/>).

Wildlife Conservation Strategies as an indicator of the condition of the wet system types with which its species are associated. This assessment focuses only on four of these species, for which detailed distribution data are available: Arizona toad (*Anaxyrus microscaphus*), Chiricahua leopard frog (*Lithobates* (aka *Rana*) *chiricahuaensis*), northern leopard frog (*L. pipiens*), and Yavapai leopard frog (*L. yavapaiensis*). The assessment does not include the Eastern tiger salamander (*Ambystoma tigrinum*), which is ubiquitous across the eastern U.S. including across the U.S. portion of the ecoregion. Its distribution therefore is not sensitive to variation in conditions within the ecoregion.

The Arizona toad requires habitat near and within functioning riparian corridors, and therefore is highly sensitive to losses of riparian habitat (NatureServe 2015). The three leopard frogs preferentially use shallow-water and/or floodplain wetland habitat along perennial streams and rivers for all stages of their life cycles; but may also use non-riparian wetlands, including those around springs and seeps; and, at times, temporary pools along streams. As a result, their abundance and distribution are sensitive to changes in surface water availability and water quality alongside springs, streams, and rivers. No data were located indicating that perennial streams and large rivers in the ecoregion differ in the amphibian species they can support under natural conditions. The Yavapai leopard frog may be extirpated within the U.S. portion of the ecoregion.

Table 3-2. Amphibians of the U.S. portions of the Chihuahuan desert ecoregion.

Species	Common Name	Near-Stream*	Other Wetlands*	Temporary Pools*	Upland	State SGCN*	Global Rank*	State Rank, NM*	State Rank, TX*
<i>Ambystoma tigrinum</i>	Eastern tiger salamander	X	X	X	X	X	5	5	5
<i>Anaxyrus cognatus</i>	Great Plains toad			X	X		5	5	5
<i>Anaxyrus debilis</i>	Eastern green toad			X	X		5	4	4
<i>Anaxyrus microscaphus</i>	Arizona toad	X			X	X	3	2	
<i>Anaxyrus punctatus</i>	Red spotted toad		X	X	X		5	5	5
<i>Anaxyrus speciosus</i>	Texas toad			X	X		5	3	5
<i>Anaxyrus woodhousii</i>	Woodhouse's toad	X	X	X		X	5	5	U
<i>Aneides hardii</i> (1)	Sacramento Mountain salamander				X	X	3	3	
<i>Craugastor augusti</i>	Barking frog				X	X	5	2	4
<i>Eleutherodactylus cystignathoides</i>	Rio Grande chirping frog				X		4		3
<i>Eleutherodactylus guttilatus</i>	Spotted chirping frog				X		4		3
<i>Eleutherodactylus marnockii</i>	Cliff chirping frog				X		5		5
<i>Gastrophryne olivacea</i>	Great Plains narrowmouth toad			X	X	X	5	1	5
<i>Hyla arenicolor</i>	Canyon treefrog	X		X	X		5	4	4
<i>Hyla wrightorum</i>	Mountain or Arizona treefrog	X	X	X	X		4	3	
<i>Lithobates berlandieri</i>	Rio Grande leopard frog	X		X		X	5	3	5
<i>Lithobates blairi</i>	Plains leopard frog	X	X	X		X	5	4	5
<i>Lithobates catesbeianus</i> (2)	American bullfrog	X	X				5		
<i>Lithobates chiricahuaensis</i>	Chiricahua leopard frog	X	X	X		X	2/3	1	
<i>Lithobates pipiens</i>	Northern leopard frog	X	X			X	5	1	1
<i>Lithobates yavapaiensis</i> (3)	Yavapai leopard frog	X		X		X	4	1	
<i>Scaphiopus couchii</i>	Couch's spadefoot toad			X	X		5	5	5
<i>Smilisca baudinii</i>	Mexican treefrog			X	X		5		3
<i>Spea bombifrons</i>	Plains spadefoot			X	X		5	5	5
<i>Spea multiplicata</i>	New Mexico spadefoot			X	X		5	5	5
<p>* Column Headings and Color Highlighting:</p> <ul style="list-style-type: none"> • Near-Stream: Associated with shallow-water and floodplain wetland habitat along perennial streams • Other Wetlands: Associated with permanent non-riparian wetlands, including around springs and seeps • Temporary Pool: Associated with intermittent or ephemeral pools, including along stream channels • Upland: Associated with habitats away from/independent of open water or wetlands • State SGCN: Listed as a State Wildlife Action Plan "Species of Greatest Conservation Need" in NM or TX • Global Rank: Global conservation status, with G1-G3 (critically imperiled, imperiled, or vulnerable) species highlighted • State Rank: State conservation status, with S1-S3 (critically imperiled, imperiled, or vulnerable) species highlighted 									
<p>Notes: (1) Range of the species may not extend into the analysis extent for the REA; (2) the species is exotic/invasive in the Chihuahuan desert; (3), the species is believed extirpated within US portion of Chihuahuan desert.</p>									

3.2 Change Agents

Rapid ecoregional assessments do not attempt to assess all threats to CEs in an ecoregion – another impossible task, as noted in Chapter 2. Instead, REAs focus on a limited set of key stressors, termed

Change Agents (CAs). All REAs address a core set of four overarching CAs: climate change, wildfire, invasive species, and development. The latter category includes crop production and industrial (e.g., energy) development as well as urban and suburban growth. The AMT recognizes that wildfire *per se* is a type of natural disturbance that can affect most – if not all – of the fourteen CEs selected for the Chihuahuan Desert REA. However, alterations to the natural fire regime that result in *unusual* fire patterns do constitute a Change Agent. This CA therefore alternatively may be termed “uncharacteristic wildfire.” The AMT, in cooperation with the Technical Team, also selected two additional CAs for this REA, concerning domestic grazing, and landscape restoration. The AMT recognizes that domestic grazing need not change CEs in ecologically significant ways, but can do so if not managed to prevent such drastic change. This CA therefore has been termed “excessive domestic grazing.” Landscape restoration, in turn, is not a stressor but an intentional counter-measure against some stressors that can bring about significant changes in this ecoregion of interest to the BLM.

These CAs do not encompass all stressors affecting the CEs of the ecoregion. For example, the taking of plants is especially problematic for some endemic cactus species that may only occur within small areas, where they are highly vulnerable to extinction (Hoyt 2002). REAs cannot adequately assess such highly localized stressors, which require the detailed knowledge of local management districts and experts.

The term “Change Agent” points to a concern with change and possible future conditions. As discussed below – see Management Questions, this Chapter – the present REA examines the present distribution and impacts of all CAs, but evaluates the possible future impacts of only two CAs, climate change and development, for which forecasts are available.

3.2.1 Climate Change

The climate of the southwestern U.S. has changed over the past century and particularly over the past few decades. Seasonal average temperatures have recently increased by 0.16-0.21 °F per decade, particularly during spring and summer (Kunkel 2013a, 2013b). The magnitude and frequency of extreme heat periods have increased while the magnitude and frequency of extreme cold periods have decreased. Unlike temperature regimes, long-term precipitation patterns have not shown significant trends, although there appears to be a slight increase in fall precipitation (Kunkel 2013b).

Multiple alternative climate models consistently predict several changes in the climate of the Chihuahuan Desert in the U.S. over the next century. Temperatures are predicted to increase, with extreme weather events such as droughts becoming more severe (Kunkel et al. 2013a, 2013b; Melillo et al. 2014). The models also predict increasing spatial variability in temperatures, with some areas warming more than others. Additionally, the models predict a decrease in the amount of average annual precipitation and an increase in the number of days with little to no precipitation (Kunkel 2013b). The Phase II (Assessment Phase) report for the Chihuahuan Desert REA will include a detailed discussion of climate change forecasts for the ecoregion.

The conceptual models for the Chihuahuan Desert REA Conservation Elements indicate that climate change potentially will significantly affect all fourteen CEs. The effects include direct impacts from changes in air temperatures and precipitation on the metabolisms of species, which may be individuals

CEs, members of species assemblage CEs, or species critical to the dynamics of ecological systems. The effects also include indirect impacts, such as effects on wildfire dynamics, groundwater recharge-discharge, or on the viability of invasive species with differing tolerances for the altered climate. Climate change will also result in changes in human activities on the landscape, such as rates of water consumption, which will themselves have additional impacts on CEs.

3.2.2 Uncharacteristic Wildfire

Fire has historically played a different, but significant, role in each of the dry system types within the ecoregion. Fire was common in the desert grassland systems, with fire return intervals typically 10 years or less. These frequent fires limited encroachment by shrubs by killing recruits before they get established. Similarly, the pinyon-juniper woodlands experienced frequent low-intensity fires that consumed the fine fuels in the herbaceous layer while leaving the trees unscathed.

These historic fire regimes have changed following the introduction of livestock by European settlers. Foraging livestock reduced both the cover and abundance of grasses and forbs, which changed the amount and continuity of the fine fuels resulting in less frequent fire. In the grasslands this allowed for encroaching shrubs to establish further changing the fire regime. In the woodlands it allowed for an accumulation of woody fuels resulting in larger, and more severe, fires when they did burn. Such stand replacing fires can result in significant erosion on slopes with erodible soils.

The desert scrub system burned infrequently in the past because of the lack of fine fuels and the discontinuity of the native shrubs. This has not changed significantly, and the current fire regime is likely very similar to historic.

3.2.3 Invasive Species

Non-native species introduced into a landscape can have a range of effects, from no measurable impacts to facilitating system transition. Within the Chihuahuan there are examples of all types. For example, both buffelgrass, (*Pennisetum ciliare*), and Lehmann lovegrass, (*Eragrostis lehmanniana*) can displace native grasses and forbs and change the fire regime of native communities. Cheatgrass, (*Bromus tectorum*), which is widely distributed in the Chihuahuan has converted thousands of square kilometers of Great Basin sagebrush steppe into monospecific grasslands. Tamarisk, (*Tamarix* spp.), has displaced native riparian communities throughout the southwest.

Native species can also be invasive. For example, honey mesquite, (*Prosopis glandulosa*), readily encroaches into desert grasslands facilitated by cattle. Once established it will displace the native grasses transitioning the land into a mesquite duneland (Peters and Gibbens 2006).

3.2.4 Development

Land development for crop production, industry, recreation, and urban/suburban growth affects most CEs in the U.S. portion of the ecoregion, as discussed in Chapter 2. Water use associated with these forms of development in turn affects all wet system CEs:

Alluvial soils along the Pecos River and Rio Grande and along smaller rivers such as the Mimbres River

and Rio Hondo are intensively farmed, irrigated from surface water and groundwater sources. The spatial extent of this intensive, irrigated farming is affected by the availability of water, crop demand, and efficiencies in farming and irrigation practices.

Human population density and urban development have increased over the last 150 years in the southwestern U.S. in general as well as in the Chihuahuan Desert ecoregion in particular (Ruhlman et al. 2012, Theobald et al. 2013). Much of the human population increase has occurred in urban areas that continue to spread, with surrounding zones of expanding suburban and exurban development as well (Ruhlman et al. 2012, Theobald et al. 2013). The five largest urban areas in the U.S. portion of the ecoregion (populations > 20,000) are El Paso, Texas, and Las Cruces, Roswell, Alamogordo, and Carlsbad, New Mexico. With the exception of Alamogordo, these urban areas all straddle rivers: Roswell, New Mexico, straddles the Rio Hondo; Carlsbad, New Mexico, the Pecos River; and Las Cruces, New Mexico, and El Paso, Texas, the Rio Grande. These juxtapositions result in urban development of floodplains and implementation of measures to prevent flooding of developed lands. These trends of population growth and urban expansion are expected to continue into the foreseeable future (Ruhlman et al. 2012, Theobald et al. 2013). In fact, the southwestern U.S. is expected to experience greater population growth than the rest of the U.S. (Travis 2007, Theobald et al. 2013). New Mexico alone is expected to see an increase in population by another third by 2030 according to the Census Bureau's population predictions (Theobald et al. 2013).

Impoundments on the Pecos River and Rio Grande control flooding and supply water to irrigation districts and to municipalities including Las Cruces and Roswell, New Mexico, and El Paso, Texas (Ruhlman et al. 2012). Most of the water from the Rio Grande, much of the water from the Pecos River, and most of the water from the lower reaches of their perennial stream tributaries is diverted for use by municipalities and agriculture (Hoyt 2002). Together, the diversions and the impoundments and their operations inundate large areas of former floodplain and alter river hydrology and connectivity. The return flows from agricultural and municipal water uses carry heavy loads of dissolved salts as consequences of these uses. River regulation, dams, diversions, and return flows with degraded water quality have contributed to changes in native fish populations and floodplain forests and wetlands. Groundwater extraction from both basin fill and alluvial aquifers has reduced the flow of water from springs and lowered water tables, which can negatively affect floodplain and emergent wetlands, endemic fish, and invertebrate species.

Many springs in the ecoregion, such as the Balmorhea Springs complex in Texas, have been developed for recreational use. While not resulting in water consumption, such recreational development typically eliminates wetland habitats and significantly alters aquatic habitat conditions.

Land development for solar and wind energy production, and for oil and gas production and transport also have affected and have the potential to further affect CEs in the ecoregion (Ruhlman et al. 2012, USEIA 2015, NMEMNRD 2016). As discussed in Chapter 2, the Pecos River basin includes the western third of the Permian Basin, an area of extensive conventional oil and gas extraction and the most productive conventional oil and gas basins in the entire U.S. (USEIA 2015). This same landscape also contains several tight oil and gas plays – i.e., geologic formations that produce oil and gas efficiently

using the non-conventional methods of horizontal drilling and hydraulic fracturing (see Chapter 2) (USEIA 2016). As a result, the Pecos River basin has a high density of oil and gas wells and associated processing and transport infrastructure, from which radiate additional pipelines. Land development for oil and gas production and transport in the U.S. portion of the ecoregion mostly has affected grassland land cover. Ruhlman et al. (2012) estimated that oil and gas extraction resulted in the conversion of 217 ±101 km² of grasslands in the ecoregion between 1973 and 2000, although their definition of the ecoregion extends further east than does the definition used in the present REA. Both conventional and tight oil and gas extraction are expected to continue expanding in the ecoregion (USEIA 2015, NMEMNRD 2016). Hydraulic fracturing requires large volumes of water, only some of which can be recycled following use. It also poses risks of water pollution from well leakage and waste spills, although these risks are subject to significant regulation (NMOGA 2012, NMEMNRD 2016). Hydraulic fracturing has been used in oil and gas extraction in the ecoregion for many decades (NMOGA 2012), but its use is expanding as a result of the more recent coupling of hydraulic fracturing with horizontal drilling technologies (USEIA 2015, NMEMNRD 2016). The USGS estimates that water use for oil and gas extraction accounted for the largest increase in water use in New Mexico between 2005 and 2010 (Maupin et al. 2014). However, changes in technology have reduced the amount of water needed, including both fresh and brackish water (NMEMNRD 2016, New Mexico Energy Forum 2016).

3.2.5 Excessive Domestic Grazing

The Chihuahuan desert was not heavily grazed by bison or other ungulates for at least the last 10,000 years prior to European-American colonization (Mack and Thompson 1982, Bock and Bock 1993, Havstad and Schlesinger 2006). The lack of continuous, intensive grazing pressure from large ungulates allowed plant species with low tolerance to defoliation and grazing to populate the Chihuahuan desert along with less palatable plants. The arrival of domestic cattle to the region by the Spanish in the 1500s, introduced cattle grazing as a new disturbance to the ecoregion (Havstad et al. 2006). The intensity of cattle grazing increased significantly in the U.S. portions of the ecoregion following the acquisition of the lands of the southwestern states by the U.S. At the peak of grazing intensity between 1890 and 1920, ranchers grazed more than a million cattle in the southwestern U.S. (Frederickson et al. 1998), altering vegetation, soil structure and erosion, and runoff dynamics, as discussed in Chapter 2 and also in Chapters 5-7.

Excessive domestic grazing is considered to be one of the major degraders of rangeland health, as discussed in detail in Chapters 5-7. Briefly, excessive grazing can alter plant community composition because cattle select those more palatable herbaceous plants that have fewer defenses against herbivory. This reduces the amount of resources taken up by palatable plants and leaves more resources available for less palatable plants to increase in size and density. Honey mesquite (*Prosopis glandulosa*), a woody species that can invade grasslands following excessive grazing, illustrates the ways in which a plant species can benefit from excessive grazing of rangelands (Havstad et al. 2006). This plant has physical and chemical characteristics that deter grazing but produces seeds that are readily consumed and dispersed by livestock (Havstad et al. 2006). The resulting expansion of honey mesquite into former grasslands alters ecological processes such as net primary productivity, nutrient cycling, energy flow, fire regimes, and food web dynamics (Sims and Singh 1978, Detling 1988, Archer and Smeins 1991, Hobbs et

al. 1991, Havstad et al. 2006). Reductions in herbaceous species and increases in shrub species can also negatively affect wildlife such as grassland obligates while simultaneously benefiting shrubland wildlife. Trampling of wetland habitat and stream banks, and inputs of cattle wastes into water bodies, also can alter CEs. The effects of excessive grazing thus are diverse, as discussed further in the conceptual models for individual CEs.

3.2.6 Landscape Restoration

Upland restoration in the Chihuahuan desert has been largely focused on recovering degraded grasslands. Desert grasslands occur between the desert scrub at lower elevations and the pinyon-juniper woodlands at higher elevations. As noted above, the introduction of livestock into the Chihuahuan desert had dramatic effects on the fire regimes within the region resulting in the expansion of woody shrubs and trees into areas that were historically grasslands.

Grassland restoration efforts have focused on shrub removal, either through chemical, mechanical (e.g., chaining) or prescribed fire treatments. These efforts are rarely successful with a single treatment as mesquite and other invasive shrubs are deep rooted and resilient to most disturbance. Repeated treatments are typically necessary to transition these mixed shrub/grass communities back to grasslands. Once the woody shrubs become dominant they largely eliminate the native grasses and forbs reducing the stability of the soil to wind and water erosion. This is a threshold point where the community is likely to transition to a mesquite-duneland. Restoring the site from a duneland back to a grassland system is likely impossible.

3.3 Management Questions

Rapid ecoregional assessments also prioritize the questions they seek to answer, concerning the CEs and CAs. Specifically, REAs focus on a limited set of Management Questions (MQs) that can be addressed using geospatial data, as discussed in Chapter 2. Four basic MQs for all REAs concern the geographic distribution of each CE, how the condition of each CE varies across its geographic distribution, the geographic distribution of each CA, and the forecasted future geographic distributions of impacts of those CAs for which forecasts are available. Additional, more specific MQs address management concerns that cannot be resolved by individual offices alone and have regional importance. In the case of the present REA, most of these more specific questions focus on either (1) interactions between specific CAs and specific CEs, or (2) the distribution and condition of specific attributes of individual CEs, such as particular habitat types or particular groups of species within an ecosystem.

The AMT for the present REA initially identified 73 possible MQs, which the AMT and Technical Team then pared down to 53 possible MQs. Table 3-3 lists the 17 highest-priority MQs, for which sufficient resources were expected to be available for at least some form of assessment, and identifies the CEs and CAs to which each MQ pertains. In preparing Table 3-3, the Technical Team condensed several individual MQs that addressed variants of the four basic MQs, labeling the results MQs #A-#D, and clarified the wording of other MQs. Appendix I presents the complete final list of 53 MQs developed by the AMT.

The AMT also posed potential MQs about the individual CEs that needed to be addressed in the conceptual model for each CE rather than through analyses of geospatial data. These questions concern the ways in which each CA potentially could affect each CE—the causal processes and outcomes involved. The conceptual model for each CE thus needed to answer questions such as, through what causal processes might climate change affect the condition of montane-headwater perennial streams, or through what causal processes might a change in the wildfire regime affect the condition of habitat for pronghorn? In this way, the conceptual model for each CE sets the stage for answering several of the geospatial MQs listed in Table 3-3, and for identifying management implications of the findings for MQs #C and #D in relation to the findings for MQs #A and #B.

Table 3-3. Chihuahuan Desert REA Management Questions.

MQ #	Question	CE(s)	CA(s)
A	What is the geographic distribution of each CE?	All	n/a
B	What is the current condition of each CE across its geographic distribution?	All	n/a
C	What is the current geographic distribution of the impacts of each CA, both in general and in relation to each CE?	All	All except Climate Change, for which “current distribution” is the baseline MQ #D.
D	What are the forecasted geographic distributions of development and climate change impacts in relation to each CE?	All	Climate Change, Development
1	Where have restoration treatments been applied to dry system CEs, and what is the status (e.g., success rate) of those treatments?	All Dry-System CEs	Landscape Restoration
2	What is the geographic distribution of the Chihuahuan desert amphibian assemblage?	All Dry- and Wet-System CEs	n/a
3	Where would uncharacteristic wildfire likely increase sedimentation and loss of habitat among the wet systems?	All Wet Systems	Uncharacteristic Wildfire
4	What areas of potential black-tailed prairie dog habitat would support restoration?	Black-tailed Prairie Dog	Landscape Restoration
5	Where are the areas of greatest faunal species biodiversity among the species and species-assemblage CEs taken together?	All Species and Species Assemblage CEs	n/a
6	Where will urban and industrial growth impact intact grasslands or impede their recovery?	Chihuahuan Desert Grasslands CE	Development, Landscape Restoration
7	How do the current and historic geographic distributions of the dry system CEs differ?	All Dry-System CEs	n/a
8	How will urban and industrial growth alter the geographic distribution of the grassland bird assemblage?	Grassland Bird Assemblage CE	Development
MQ#	Question	CE(s)	CA(s)

9	What and where are the aquifers and their recharge zones that support the wet systems?	All Wet-System CEs	Development
10	How do the current and historic geographic distributions of the Pecos River and Gila River fish assemblages differ?	All Wet-System CEs except Playas	n/a
11	Where are the breeding, winter, and year-around habitats for pronghorn and mule deer?	Pronghorn; Mule Deer	n/a
12	Are there areas where invasive plants are being killed on a broad scale (e.g., by the tamarisk leaf-eating beetle) where managers need to focus on restoration or controlling succession?	All Wet-System CEs	Invasive Species; Landscape Restoration
13	What is the current geographic distribution of the impacts of gypsum in the soil and water, in general and in relation to each CE and CA?	All	All except Climate Change

4 Conceptual Models and Modeling Methods

As noted at the start of Chapter 3, the Pre-Assessment Phase (*aka* Phase I) of a REA focuses on (1) identifying the Conservation Elements (CEs), Change Agents (CAs), and Management Questions (MQs) on which to focus the assessment, and (2) developing conceptual ecological models for the conservation elements. The conceptual ecological models show how the change agents may affect each conservation element, and provide a means for translating management questions into terms specific to each individual conservation element and/or change agent. The present chapter describes the methods applied to develop the conceptual models for the Chihuahuan Desert REA CEs, and presents the overarching terrestrial (“dry”) and aquatic-wetland (“wet”) system conceptual models for the ecoregion. Chapter 2 (see above) presents a general, narrative conceptual ecological model for the Chihuahuan desert ecoregion as a whole.

4.1 Conceptual Modeling Framework

Conceptual models in ecological resource management synthesize existing knowledge about how some ecological resource “works.” They are widely recognized as critical to the success of ecological resource management actions (Fischenich 2008, Conroy and Peterson 2012, DiGennaro et al. 2012). The resource of concern may be one or several populations of some particular species, a type of habitat or set of related habitats, or an ecological system. Conceptual models for ecological resources explicitly identify:

- The *variables or attributes* that best characterize resource condition.
- The factors, commonly termed *drivers*, that most strongly shape or control these variables, under both natural and altered (including managed) conditions. Change Agents as defined in REAs are always drivers.
- The *causal relationships* through which the drivers do their shaping or controlling, and characteristics of these relationships including their strength, predictability, feedbacks, and threshold effects.
- How the characteristics of the resource vary as these causal relationships play out over time and space.

By integrating and explicitly organizing existing knowledge in this way, the conceptual model for an ecological resource summarizes scientific understanding about (1) how and why the condition of the resource varies in response to natural variation in driver conditions, and (2) how and why it would be expected to change in response to changes in driver conditions beyond natural ranges of variation. At the same time, the conceptual model for an ecological resource necessarily also identifies the sources of information available concerning the resource and the drivers of its condition, and the certainty of this information. In effect, all statements in the conceptual model for an ecological resource constitute hypotheses about how characteristics of the resource are likely to vary or change as a result of changes in its drivers, including changes due to management actions. These hypotheses can then guide management action, including actions to test hypotheses to improve the model.

A well-constructed conceptual model for an ecological resource also identifies key attributes of the resource that managers can use to monitor resource condition and test hypotheses about the possible

effects of change agents and management actions. The Chihuahuan Desert REA refers to such key attributes as the “key ecological attributes” for each CE, on which to focus the assessment. Key ecological attributes include defining physical, biological, and ecological characteristics of a CE, along with its abundance and/or spatial distribution. The defining physical, biological, and ecological characteristics of a CE may include characteristic biological and ecological processes. When one or more key ecological attributes of a CE become stressed in a specific setting, i.e., are altered so that they depart significantly from long-term historic conditions, the entire CE in that setting is degraded or, in extreme circumstances, will disappear. A well-constructed conceptual model for a CE necessarily identifies a limited set of key ecological attributes to represent the overall condition of the CE. Ecosystem complexity, the limits of scientific knowledge, and the constraints of budgets prevent evaluation of all possible characteristics and processes of any single resource.

REAs also typically focus on subsets of the key ecological indicators for each CE. Specifically, REAs focus on those key ecological attributes for each CE with readily measurable, geospatial indicators. REAs focus on the status and patterns of variation in such geospatial indicators over time and space, using only geospatial datasets that can be analyzed without additional research and are available for all or most of the ecoregion of interest. REAs do not conduct research or collect new data. Additionally, the conventional indicators for some key ecological attributes may consist of spatially intensive rather than extensive measurements collected only directly at ground level, in water bodies, or from individual organisms. As a result, investigative teams may not always find geospatial indicators for some key ecological attributes for an individual CE, for inclusion in an REA. Phase II of an REA includes a concerted effort to locate suitable indicator data for the key ecological indicators identified for each CE in Phase I.

4.2 Chihuahuan Desert Conceptual Model Hierarchy

The ecological dynamics of the Chihuahuan desert play out at multiple ecological scales, each best characterized by a conceptual model specific to that scale. Following the hierarchical approach advocated by Miller et al. (2010) and the National Park Service, Chihuahuan Desert Inventory and Monitoring Network (2010), the Chihuahuan Desert REA developed separate conceptual models for the ecoregion as a whole, for all dry ecological system types taken together, for all wet ecological system types taken together, and for each individual CE, as follows:

- The **Chihuahuan Desert Ecoregional Model** is the broadest in scope and identifies the major abiotic and biotic components of the ecoregion and how they are related. This model outlines the overall biophysical setting and constraints on the dry and wet ecological systems and the individual CEs within these broad sets of systems. In the terminology of Miller et al. (2010), the Chihuahuan Desert Ecoregional Model is a “control” model. It identifies the major abiotic and biotic features of the ecoregion, the major types of drivers that shape these features, and identifies which drivers shape which features. The Chihuahuan Desert Ecoregional Model is a narrative model, presented in Chapter 2.
- The **Chihuahuan Desert Dry System Model** and **Chihuahuan Desert Wet System Model** are control models that graphically and narratively identify how the landscape is divided into specific dry and wet system types by major abiotic controlling factors such as elevation,

precipitation, watershed catchment size, groundwater discharge, and so forth. In doing so, these models summarize the physical and biological settings within which individual CEs function.

- **Conservation Element Models** graphically and narratively describe the key ecological attributes of each CE and identify the drivers that shape the condition of these attributes, including the REA change agents. The conceptual models for the individual CEs provide significantly greater detail than the Dry System and Wet System control models, in order to highlight important causal relationships. In particular, these more detailed conceptual models show how the change agents may affect each conservation element, and provide a means for translating management questions into terms specific to each individual conservation element and/or change agent. The conceptual modeling method applied to the present CEs varies by type of CE, following conventions in the different fields of ecology that apply to these different types of CEs, as follows:
 - The conceptual models for the three dry system CEs have two parts. The first part consists of a control model, similar to the overarching Chihuahuan Desert Dry System Model but with details specific to each individual CE. The second part consists of a “state-and-transition” model (STM). STMs categorize the conditions of vegetation and soil into “states,” transitions between which take place through internal natural processes and through the effects of external drivers. External drivers can include stressors, defined as novel drivers and natural drivers that vary beyond their natural ranges of variation that alter processes within and patterns of transition among states. STMs thus are a type of “stressor” model in the terminology of Miller et al. (2010).
 - The conceptual models for the wet system, species, and species assemblage CEs for the Chihuahuan Desert REA also have two parts. The first part again consists of a control model, in this case similar to the overarching Chihuahuan Desert Wet System Model but with details specific to each individual CE. The second part consists of a “stressor” model as defined by Miller et al. (2010), but not a state-and-transition model. Aquatic and wetland ecological systems, and species in general, typically are not readily amenable to state-transition modeling. The stressor model methodology implemented for the Chihuahuan Desert REA for the wet system, species, and species assemblage CEs instead is a “driver-linkage-outcomes” (DLO) methodology adapted from that of the Sacramento-San Joaquin Delta (California) Ecosystem Restoration Program (ERP):
https://www.dfg.ca.gov/ERP/conceptual_models.asp (e.g., DiGennaro et al. 2012).

4.3 Chihuahuan Desert Dry System Model

Figure 4-1 presents the control model (*sensu* Miller et al. 2010) for the Chihuahuan desert dry system overall. Each arrow in the diagram represents a relationship in which one model component affects or influences another. The model shows how the distribution of the major terrestrial ecological system types within the ecoregion is controlled by two closely related vertical gradients – in elevation and in effective moisture – and by close interactions between vegetative cover and soils. The structure, composition, and distribution of the major terrestrial ecological system types within the ecoregion also affect and are affected by the abundances and distributions of both characteristic and endemic wildlife. These dynamics ultimately are driven by climate and geology, which shape watershed topography and hydrography, weather patterns (e.g., precipitation and temperature), and soil parent materials. These factors in turn shape spatial and temporal variation in snowmelt, evapotranspiration, and soil infiltration, which directly affect the vertical gradients and soil-ground cover interactions; and also shape

wildfire dynamics, which directly affect soil-ground cover interactions and the internal dynamics of all the major terrestrial ecological system types in the ecoregion.

Figure 4-1. Chihuahuan desert dry system conceptual ecological model.

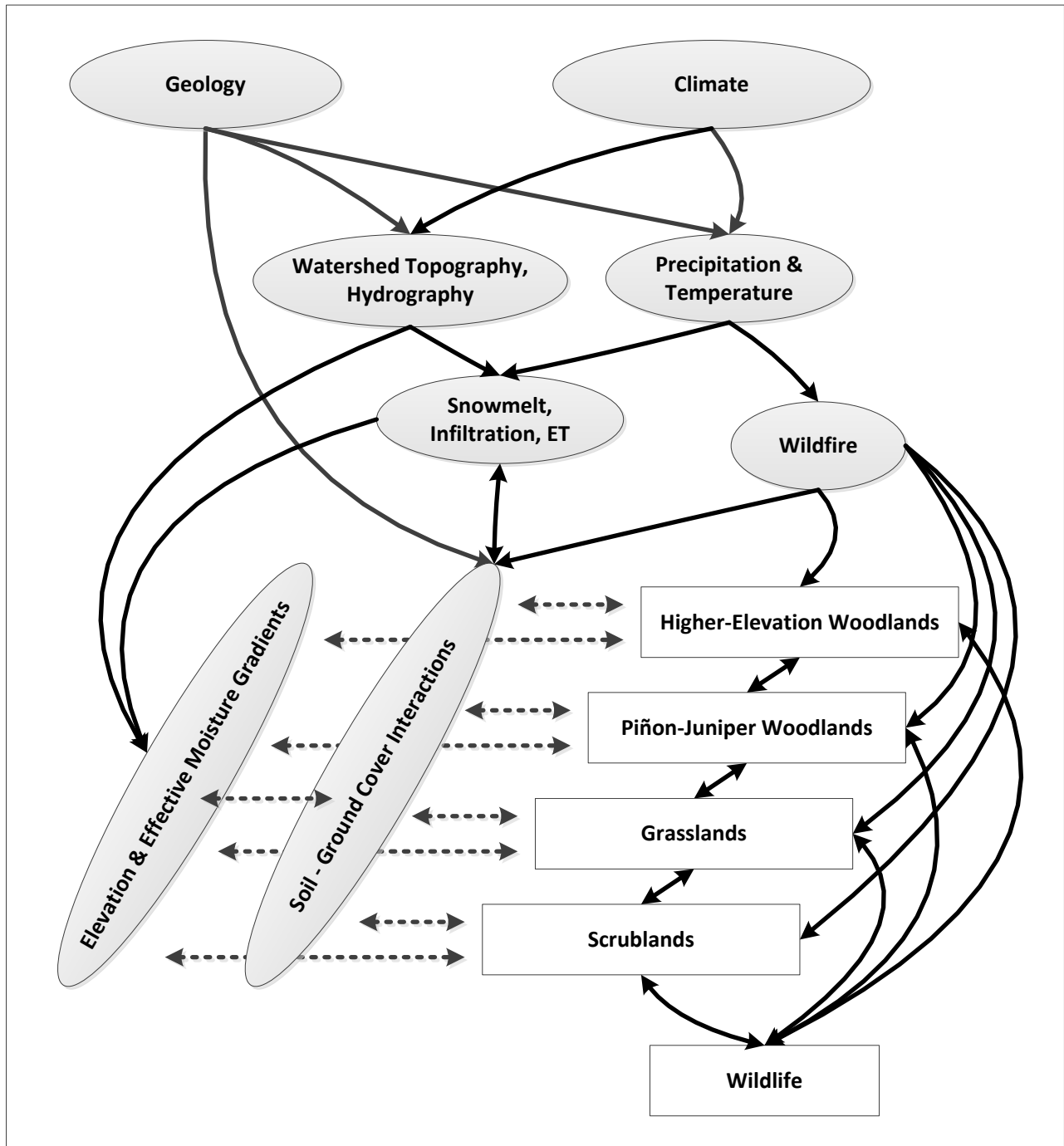
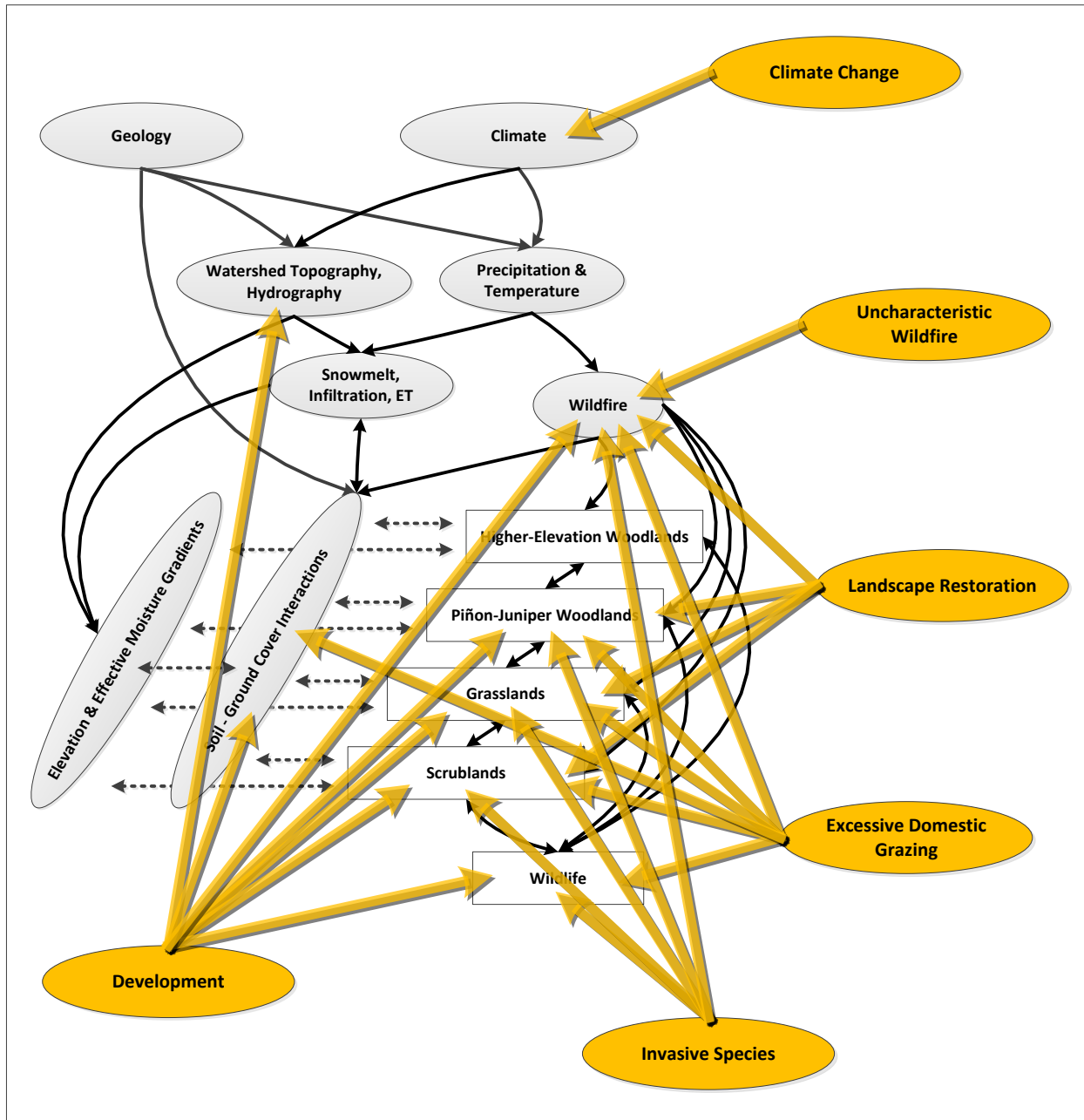


Figure 4-2 shows which change agents affect which components of the Chihuahuan desert dry system identified in Figure 4-1, with emphasis on the CEs selected for the present REA. A quick look at Figure 4-2 might suggest that climate change and uncharacteristic wildfire have only limited effects on the overarching dry system and its CEs. However, the effects of these two change agents in fact cascade through the entire system overall. Their effects consequently are just as far reaching as those of any of the other change agents.

Figure 4-2. Chihuahuan desert dry system conceptual ecological model with change agents.



4.4 Chihuahuan Desert Wet System Model

Figure 4-3 presents the control model (*sensu* Miller et al. 2010) for the Chihuahuan desert wet system overall. Each arrow in the diagram again represents a relationship in which one model component affects or influences another. The model shows how the distribution and biotic composition of the major aquatic-wetland ecological system types within the ecoregion is controlled by the interplay of discharge from both groundwater (aquifer) systems and watershed surfaces. Watershed processes affect surface water movement, chemistry, and temperature; the erosion, surface transport, and deposition of sediment and organic matter; and the chemistry and temperature of recharge. The biotic composition of the aquatic-wetland ecological system types within the ecoregion also is controlled by surface drainage connectivity, which affects which species can occur in what locations; and by connectivity among aquifer systems, which may harbor endemic species. These dynamics also ultimately are driven by climate and geology, which shape watershed topography and hydrography, weather patterns (e.g., precipitation and temperature), and soil parent materials. These factors in turn shape spatial and temporal variation in snowmelt, evapotranspiration, and soil infiltration, which directly shape each other as well as runoff and aquifer recharge; and also shape wildfire dynamics, which directly affect soil-ground cover dynamics.

The Chihuahuan desert wet system control model identifies four broad types of riparian-stream systems in the ecoregion: (1) intermittent and ephemeral stream courses (washes) at all elevations; (2) perennial streams with montane headwaters fed by shallow montane groundwater systems and runoff from rainfall and snowmelt; (3) perennial streams with lowland headwaters fed predominantly by valley- and regional-scale groundwater systems; and (4) larger streams and rivers fed by the combined discharge of montane- and lowland-headwater streams. The definitions of these broad types of riparian-stream systems include both their riparian and aquatic subsystems. The three largest rivers in the ecoregion, the Gila River, Rio Grande, and Pecos River, receive large fractions of their water, sediment, and nutrients from sources outside the CHD ecoregion. In addition to the subsidies of water, sediment, and nutrients that these three rivers bring to the CHD ecoregion, they also connect the aquatic ecosystems of the ecoregion to each other and to other ecoregions. With the exception of intermittent and ephemeral stream courses, these broad types of riparian-stream systems in the ecoregion all are CEs for the Chihuahuan Desert REA.

Similarly, the Chihuahuan desert wet system control model identifies three broad types of wetland systems in the ecoregion: (1) upland wetlands, e.g., in montane zones; (2) seeps and springs, and their associated emergent wetlands at all elevations; and (3) terminal wetlands, playas, and playa lakes, found at the lowest elevations in closed basins. All three are strongly shaped by groundwater dynamics, but also by local catchment or basin runoff dynamics. The definitions of these broad types of wetland systems include both their aquatic and wetland subsystems. With the exception of upland wetlands, these broad types of wetland systems in the ecoregion also are CEs for the Chihuahuan Desert REA.

Figure 4-4 shows which change agents affect which components of the overarching Chihuahuan desert wet system identified in Figure 4-3, with emphasis on the CEs selected for the present REA. A quick look at Figure 4-4 again might suggest that climate change and uncharacteristic wildfire have only limited

effects on the overarching wet system and its CEs. However, the effects of these two change agents in fact cascade through the entire system overall. Their effects again consequently are just as far reaching as those of any of the other change agents. Climate change in particular has the potential to have far-reaching and long-lasting effects on every aquatic and wetland resource in the ecoregion, because of the ways in which climate change every aspect of the ecoregion's hydrologic system. Figure 4-4 indicates that development has more far-reaching effects on aquatic and wetland resources in the ecoregion than on terrestrial resources. Development often involves the diversion and/or consumption of water, and also often takes place preferentially around and alongside springs, streams, and rivers.

Figure 4-3. Chihuahuan desert wet system conceptual ecological model.

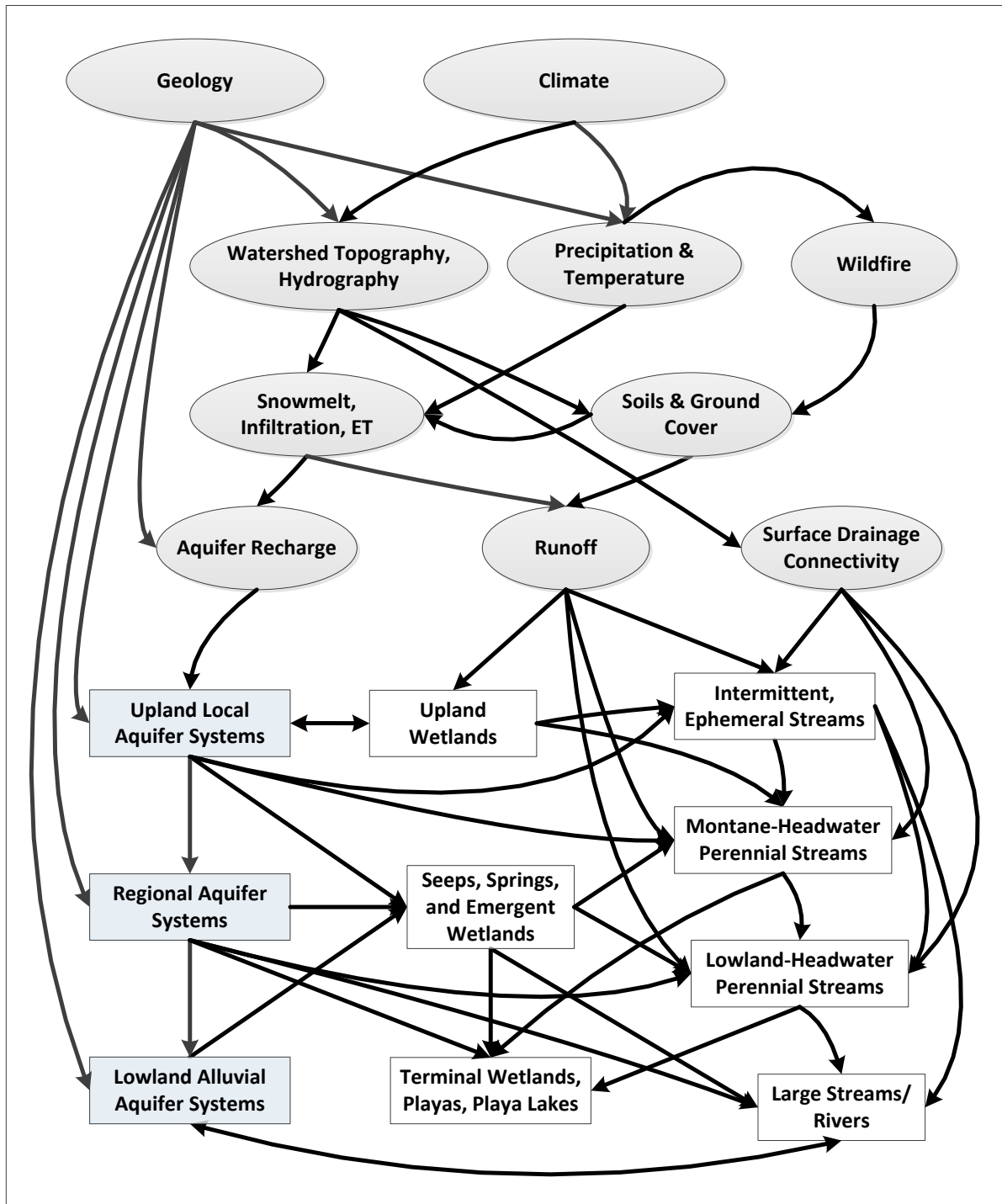
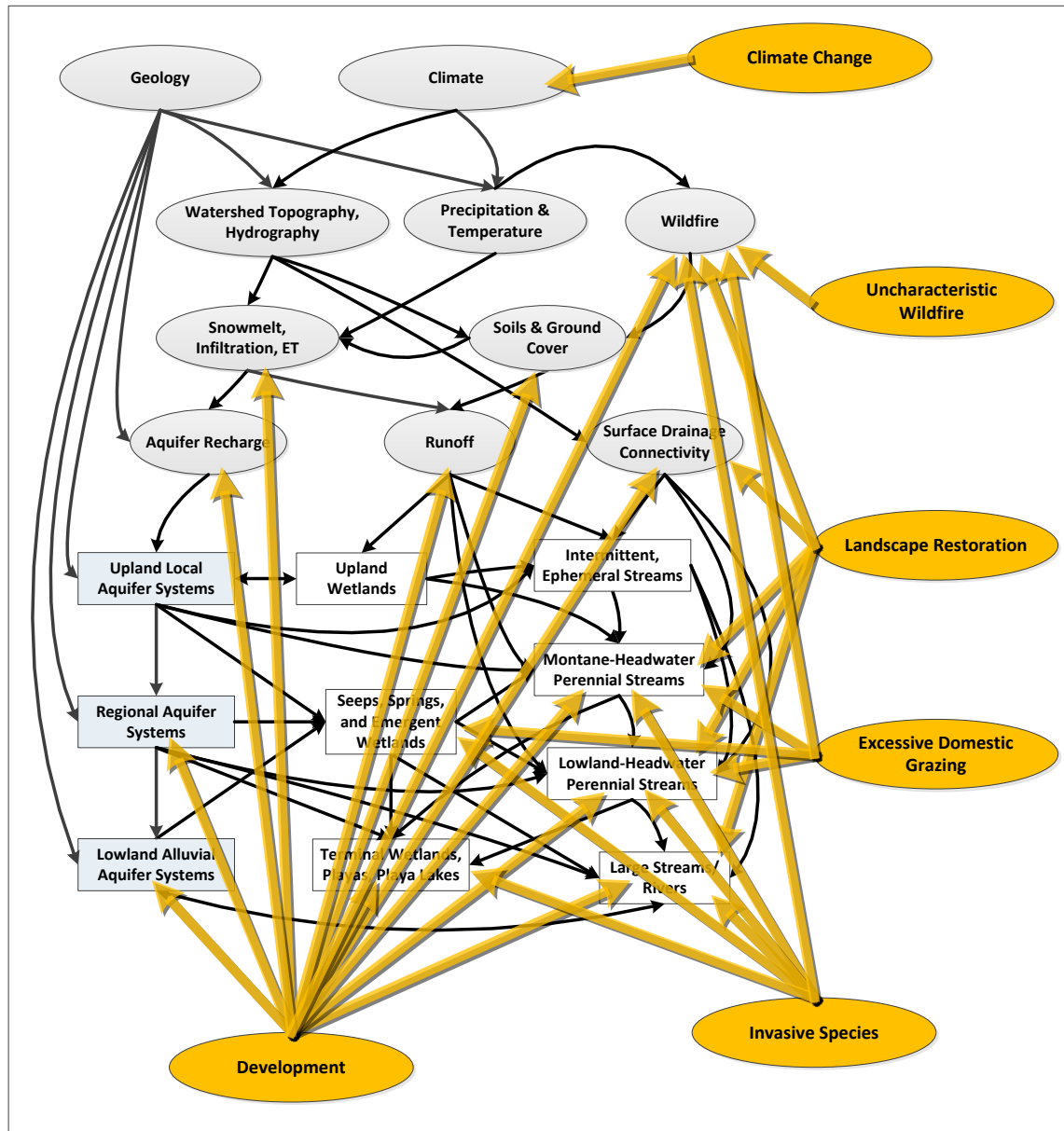


Figure 4-4. Chihuahuan desert wet system conceptual ecological model with change agents.



4.5 Dry-System Conservation Element Conceptual Model Methodology

The conceptual models for the three dry system CEs for the Chihuahuan Desert REA all derive from work by the Integrated Landscape Assessment Project (Gaines et al. 2013). Natural Heritage New Mexico (NHNM), a division of the Museum of Southwestern Biology at the University of New Mexico, developed the original conceptual models for ILAP using the Vegetation Dynamics Development Tool (VDDT, ESSA Technologies Ltd.). The resulting STMs are not spatially highly explicit. They identify large areas of land with the same potential natural vegetation type (PNVT). PNVT is largely determined by biogeochemical and disturbance processes. Land managers use data on vegetation state, soils, and disturbance dynamics to monitor system conditions and trends. The key ecological attributes identified in VDDT

models consist of key drivers, such as fire frequency and severity, precipitation and the frequency and severity of precipitation extremes, and temperature and the frequency and severity of temperature extremes; key soil characteristics; and key characteristics of plant community composition and structure.

Chapters 5-7, below, present the detailed conceptual models for the Chihuahuan Desert Grasslands, Chihuahuan Desert Scrub, and Pinyon-Juniper Woodlands CEs, respectively. The presentations of these three models have a common structure, consisting of the following sections:

- Sources of information
- Control model
- State-and-transition model
- Key ecological attributes
- Literature cited

4.6 Wet System, Species, and Assemblage Conservation Element Conceptual Model Methodology

The conceptual models for the wet system CEs, species, and assemblages were created using a DLO methodology adapted from the methodology of the Sacramento-San Joaquin Delta (California) Ecosystem Restoration Program (ERP; https://www.dfg.ca.gov/ERP/conceptual_models.asp) as noted above. DLO models identify drivers and constraints that act on an ecological resource, and the effects of those actions on characteristics of the resource such as its taxonomic and functional composition, abundance, spatial and temporal distributions, or other qualities. These effects are termed “outcomes.” The “linkages” in the model are the cause-effect relationships between drivers (or constraints) and outcomes (DiGennaro et al. 2012).²

The DLO methodology implemented here for constructing the conceptual models for the wet system, species, and assemblage CEs expands on the ERP methodology. Specifically, it incorporates recommendations that emerged from the ERP (Kondolf et al. 2008, Burke et al. 2009) for a more hierarchical approach. This expanded approach identifies causal linkages and outcomes in greater detail, by identifying four types of model components among which to evaluate causal linkages, instead of only two in the original ERP methodology.

The four types of model components included in the resulting conceptual models for the wet system, species, and assemblage CEs are:

² The ERP DLO methodology also characterizes each causal linkage along four dimensions: (1) the character and direction of the effect, (2) the magnitude of the effect, (3) the predictability (consistency) of the effect, and (4) the certainty of present scientific understanding of the effect (DiGennaro et al. 2012). The purpose of rating causal linkages along these several dimensions is twofold: (1) to build a decision support tool with which to estimate the possible consequences of changes to a driver through management actions; and (2) to track data on crucial gaps in knowledge. Rapid ecoregional assessments do not require all of this additional level of detail, and the present assessment only characterizes the character and direction of each causal linkage. Information on the other three dimensions can be added later if/as needed.

- **Ecological outcomes.** These consist of critical properties of the species, assemblage, or ecological system of interest that unambiguously characterize the status of the resource. For species, such properties typically include reproductive success, population size and distribution, health, and so forth. For species assemblages and ecological subsystems, such properties may include demographic, taxonomic, or functional (e.g., feeding guild) composition; abundance and/or spatial and temporal distributions, and other qualities (e.g., stability/resilience). Taken together, the ecological outcomes describe the overall condition of the CE.
- **Critical ecological processes.** These consist of disturbance regimes and processes of interaction among environmental and biological conditions that strongly shape the ecological outcomes of the subsystem. They include processes that may inhibit, sustain, or promote particular ecological outcomes. Critical ecological processes vary in magnitude, frequency, duration, timing, and other or similar “rate” variables.
- **Critical environmental elements.** These consist of specific features of the physical environment that strongly inhibit, allow, or promote particular critical ecological processes in the subsystem. Other non-human species that interact with a species or assemblage of concern, including predators, parasites, or competitors, may function as critical environmental elements. Human activities (here treated as external drivers – see below) may introduce critical environmental elements not present in the natural system. Critical environmental elements may vary in their abundance, spatial or temporal distributions, and other qualities that affect the ways in which they inhibit, allow, or promote particular critical ecological processes.
- **Drivers.** These consist of natural environmental constraints, dynamics, and human activities that strongly shape the abundance, spatial and temporal distributions, and other important properties of critical environmental elements or critical ecological processes for a subsystem. All change agents for the REA constitute drivers. A hierarchy of drivers always exists, affecting the system at different scales of time and space, with very-long-term dynamics of climate and geology at the top of the hierarchy (Burke et al. 2009). The conceptual models for the wet-systems, species, and assemblage CEs focus on drivers within the scope of potential human manipulation, including management actions directed toward the CEs of interest.

The drivers, critical environmental elements, critical ecological processes, and ecological outcomes in a DLO model are linked by causal relationships, as noted above. A causal relationship exists when a change in one characteristic of a resource results in a change in another. Change in the first characteristic is said to cause change in the second. Causal relationships may be either “direct” or “indirect.” A direct causal relationship exists when one characteristic affects another without any intervening steps in the chain of causation. An indirect causal relationship exists when the chain of causation involves two or more steps. For example, a change in rural residential development in a watershed could directly affect the rate of human consumption of surface and/or groundwater in that watershed. In turn, such increased human consumption of groundwater could affect the elevation of the water table, which could affect the magnitude or even the existence of baseflow in surface streams within the watershed, which could affect the availability of habitat for fishes and other aquatic species in these streams, which could affect the abundance and taxonomic composition of the aquatic faunal communities along these streams. Such latter consequences would be classified as *indirect effects* of the change in rural residential development.

The key ecological attributes identified in the DLO models for the wet systems, species, and assemblage CEs for the Chihuahuan Desert REA consist of (1) all ecological outcomes and critical ecological

processes; and (2) critical environmental elements that directly and strongly affect ecological outcomes.

Chapters 8-11, below, present the detailed conceptual models for the Montane-Headwater and Lowland-Headwater Perennial Streams, Large River-Floodplain Systems, Springs-Emergent Wetlands, and Playas and Playa Lakes CEs, respectively. The presentations of these four models have a common structure, consisting of the following sections:

- Sources of information
- Control model
- Stressor model
- Key ecological attributes
- Stressor sub-models by change agent or other driver
- Literature cited

Chapters 12-17 present the detailed conceptual models for the Pronghorn, Mule Deer, Banner-tailed Kangaroo Rat, Black-tailed Prairie Dog, Grassland Bird Assemblage, and Grassland Small Mammal Assemblage CEs, respectively. The presentations of these six models have a common structure, consisting of the following sections:

- Sources of information
- Species or assemblage overview
- Stressor model
- Key ecological attributes
- Stressor sub-models by change agent or other driver
- Literature cited

5 Chihuahuan Desert Grasslands Conceptual Model

This chapter presents the conceptual ecological model for the Chihuahuan Desert Grasslands CE. The presentation follows the structure described in Chapter 4.

5.1 Sources of Information

The Chihuahuan Desert Grasslands conceptual model is adapted from the Integrated Landscape Assessment Project (Gaines et al. 2013). Natural Heritage New Mexico (NHNM), a division of the Museum of Southwestern Biology at the University of New Mexico, developed the original conceptual models for ILAP using the vegetation dynamics development tool (VDDT, ESSA Technologies Ltd.). The adaptation included converting the VDDT model diagrams into state-and-transition conceptual model (STM) diagrams similar in style to the hierarchical approach of Miller et al. (2010) using Microsoft Visio software. The Chihuahuan Desert Grasslands conceptual model specifically incorporates information from the Chihuahuan Semi-Desert Grassland-Sand Plains, -Piedmont, and -Foothills and Lowlands STMs (ILAP STM 2012). These STMs are not location specific but rather refer to large areas of land with the same general potential natural vegetation type (PNVT). PNVT is largely determined by biogeochemical and disturbance regime constraints.

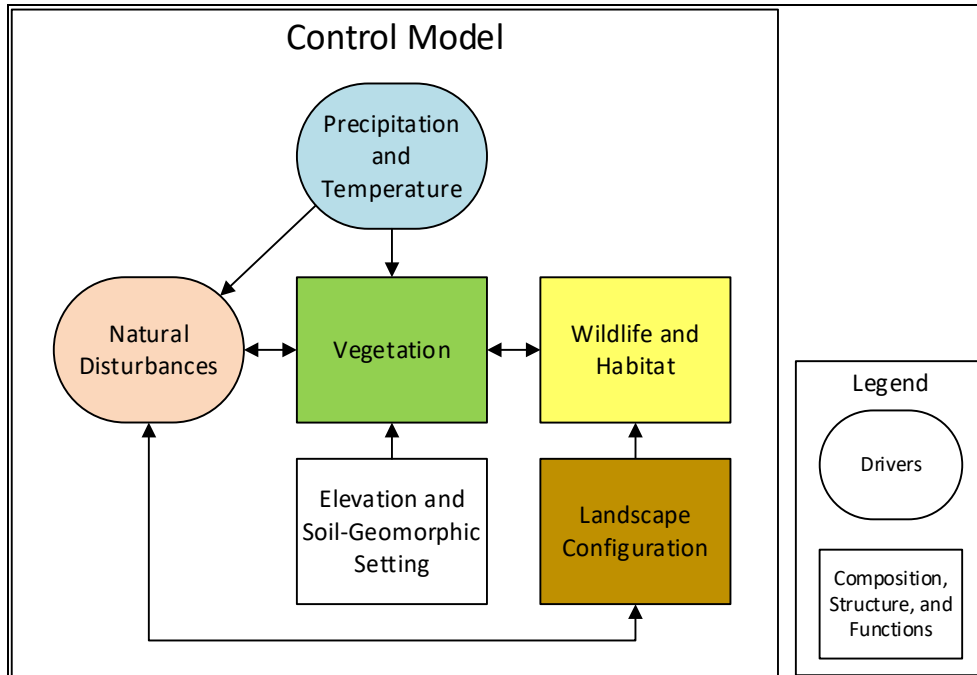
The Chihuahuan Desert Grasslands ecological system is ecologically important and has been classified by other organizations. As noted in Chapter 3, the term, “ecological system” here refers to “... recurring groups of biological communities that are found in similar physical environments and are influenced by similar dynamic ecological processes, such as fire or flooding” (Comer et al. 2003). NatureServe refers to parts of this CE as Chihuahuan Loamy Plains Desert Grassland ecological system (International Ecological Classification Code CES302.061) with Chihuahuan-Sonoran Desert Bottomland and Swale Grassland ecological systems (CES302.746) in the depressional or swale areas (NatureServe 2014). Chihuahuan Loamy Plains Desert Grassland transitions into the Apacherian-Chihuahuan Semi-Desert Grassland and Steppe ecological system (CES302.735) on foothills and piedmont slopes (NatureServe 2014). The Chihuahuan Desert Grasslands also include part of the Chihuahuan Sandy Plains Semi-Desert Grassland ecological system (CES302.736) that can degrade into the Chihuahuan Stabilized Coppice Dune and Sand Flat Scrub ecological system (CES302.737) (NatureServe 2014). The LANDFIRE classification approach, related to the NatureServe approach, uses the same classification names but assigns them as biophysical settings 2515030, 2515040, 2511210, 2511330, and 2510760, respectively (LANDFIRE 2014). The loamy ecological site description (R042XB014NM) developed by the U.S. Department of Agriculture, Natural Resources Conservation Service (USDA NRCS 2006) also applies to the Chihuahuan Desert Grasslands ecological system. The Chihuahuan Desert Grasslands conceptual model incorporates information from these publications as well.

5.2 Chihuahuan Desert Grassland Control Model

The Chihuahuan Desert Grasslands control model (Figure 5-1) illustrates current understanding of how the main ecological components and controlling processes for this ecological system. The ecological

components of the control model include landscape configuration, elevation and soil-geomorphic setting, precipitation and temperature, vegetation, natural disturbance regimes, and wildlife and habitat. The following paragraphs explain these individual model components.

Figure 5-1. Chihuahuan Desert Grasslands control model.



5.2.1 Landscape Configuration, and Elevation and Soil-Geomorphic Setting

Chihuahuan Desert Grasslands in the U.S. portions of the ecoregion occur on four major landforms including piedmont on coalesced alluvial fans, foothills on colluvium, lowlands on basins and playas, and sandy plains on sand sheets (ILAP STM 2012). These grasslands mainly occur between 1,100-1,700 m in elevation (Brown 1994).

5.2.2 Precipitation and Temperature

Chihuahuan Desert Grasslands receive approximately 250-450 mm of annual average precipitation with more than half occurring during April-September (Brown 1994) from thunderstorms with high precipitation variability among years (Brown 1994). During the summer, days with temperatures exceeding 100° F are common (Brown 1994). Winters are mild with usually less than 100 days per year with freezing temperatures (Brown 1994). Rain during the winter can deeply infiltrate the ground to the benefit of C₃ shrubs and subshrubs with deep roots (Burgess 1995). A lot of winter precipitation may also result in a flush of annuals that uptake soil nutrients and reduce nutrient availability for warm season grasses that start growing later in the year (Burgess 1995, Neilson 1986). The infrequent severe frosts in the otherwise mild winter climate of the Chihuahuan desert region (Brown 1982, McClaran 1995) limits the distribution of some of the subtropical woody species with extensive deep roots (e.g., velvet mesquite; Glinski and Brown 1982) and water storing succulent plants (Burgess 1995).

Additionally, the duration of soil moisture resulting from a precipitation event determines the length of plant growth periods, which greatly influence where the different plant growth forms can grow and persist over the long term (Burgess 1995). Predicted changes in temperature and precipitation may alter these constraints, causing some species distributions to expand, retract, or shift within the ecoregion.

The dominance of grasslands by C₄ plants across southern New Mexico is thought to be largely a result of the warm growing seasons with low and varying amounts of rain mixed with periodic drought (Peters and Gibbens 2006, Schmutz et al. 1992). Dry weather during winter and spring with subsequent wet late-summer or early-fall weather favor black grama grass (*Bouteloua eriopoda*) seedling establishment. These weather patterns were more common in the late 1800s than during the 1900s (Burgess 1995). The shift of some summer precipitation to winter during the 1900s could explain part of the shift from black grama grass to mesquite dominance (Burgess 1995). This vegetational shift has been recorded on both grazed and ungrazed desert grasslands (Burgess 1995, Hennessy et al 1983). Regularly occurring droughts of long duration can also kill most perennial grasses, However, frequent, brief droughts may help grasses by inhibiting other, competing species from developing extensive, deep root systems (Burgess 1995). Increased winter precipitation can also hasten the transition to shrub dominance because it favors C₃ shrubs over C₄ grasses (ILAP STM 2012). Creosotebush (*Larrea tridentate*), for example, is a C₃ evergreen species adapted to living with drought that can survive for as many as 400 years (Miller and Huenneke 2002, Peters and Gibbens 2006).

5.2.3 Natural Disturbances

Fire and drought are common sources of natural disturbance that influence plant community composition in the Chihuahuan desert grasslands, as also suggested in the discussion of precipitation and temperature. Wildfire can alter ecological condition and plant community composition (Drewa et al 2006, Drewa and Havstad 2001, Parmenter 2008). Further, drought may promote fire, and fire may cause greater mortality to vegetation during droughts. These interactions are discussed more fully below, in the presentation of the state-and-transition model for this CE.

5.2.4 Vegetation

Chihuahuan Desert Grasslands occur at elevations intermediate between Chihuahuan Desert Scrub (at lower elevations) and southwestern Great Plains Grasslands and contains some plants and animals from these adjacent systems (Brown 1994). As noted above, the Chihuahuan Desert Grasslands (Figure 5-1) occur on four landforms that include piedmont, foothill, lowland, and sandy plains (ILAP STM 2012).

The desert grasslands on piedmont landforms of coalesced alluvial fans were historically dominated by black grama grass, (*Bouteloua eriopoda*), bush muhly (*Muhlenbergia porteri*), and fluffgrass (*Dasyochloa puchella*; ILAP STM 2012). In grasslands dominated by black grama grass, black grama grass cover can range from 44% in dry years to 75% in wet years (Paulsen and Ares 1962, Peters and Gibbens 2006). Black grama grass is a C₄ long-lived perennial lasting approximately 35-40 years that grows in open stands (Canfield 1939, Peters and Gibbens 2006, Wright and Van Dyne 1967) and spreads through stoloniferous growth (Nelson 1934, Peters and Gibbens 2006). Black grama grass has minimal establishment from seed because it only produces a few viable seeds, has a limited seedbank, and has

limited microenvironmental soil conditions conducive to its establishment from seed (Minnick and Coffin 1999, Peters 2000 and 2002, Peters and Gibbens 2006) Although, some research suggests that improved varieties may provide sufficient establishment and production (NMSU 1975). A variety of other grasses can also dominate including tobosagrass (*Pleuraphis mutica*), sideoats grama (*Bouteloua curtipendula*), hairy grama (*Bouteloua hirsuta*), blue grama (*Bouteloua gracilis*; ILAP STM 2012), *Bouteloua warnockii*, *Scleropogon brevifolius*, and *Enneapogon desvauxii*. Blue grama grasslands in southern New Mexico are likely remnants of plains-mesa grasslands (Dick-Peddie 1993). Other grasses that can occur on these piedmont grasslands include plains lovegrass (*Eragrostis intermedia*), purple three-awn (*Aristida purpurea*), red grama (*Bouteloua trifida*), and Arizona cottontop (*Digitaria californica*; ILAP STM 2012). Lehmann's lovegrass (*Eragrostis lehmanniana*), an exotic perennial introduced to improve rangeland production, has displaced native grasses on some parts of these piedmont landforms with displacement most noticeable where soils have been disturbed (ILAP STM 2012).

While grasses are the dominant vegetation type on piedmont grasslands, several shrubs and sub-shrubs occur and can form a shrub-steppe composition. Common shrubs in this area include longleaf jointfir (*Ephedra trifurca*) and Torrey's yucca (*Yucca treculeana*). Common sub-shrubs include pricklyleaf dogweed (*Thymophylla acerosa*) and woody crinklemat (*Tiquilia canescens*). The ecological stressors of drought, livestock grazing, and the reduced role of fire enable shrubs of the desert scrub to invade although there is some question about the historic role of fire in these areas. Invading shrubs can include whitethorn acacia (*Acacia constricta*), viscid acacia (*Acacia neovernicosa*), honey mesquite (*Prosopis glandulosa*), creosotebush (*Larrea tridentata*), catclaw mimosa (*Mimosa aculeaticarpa*), and tarbush (*Flourensia cernua*). Following disturbance, broom snakeweed (*Gutierrezia sarothrae*) and burroweed (*Isocoma tenuisecta*) can increase on piedmont grasslands (ILAP STM 2012). *Gutierrezia microcephala* also may increase, especially on gravel soils.

The foothills grasslands occur on colluvial foothill landforms with soils that are usually rocky. These grasslands are historically dominated by sideoats grama (*Bouteloua curtipendula*), curlyleaf muhly (*Muhlenbergia setifolia*), New Mexico feathergrass (*Hesperostipa neomexicana*), and bullgrass (*Muhlenbergia emersleyi*). Other grasses that may also be abundant include purple grama (*Bouteloua radicata*), tanglehead (*Heteropogon contortus*), blue grama (*Bouteloua gracilis*), hairy grama (*Bouteloua hirsuta*), southwestern needlegrass (*Achnatherum eminens*), and slim tridents (*Tridens muticus*). While grass is the dominant form of vegetation on foothill grasslands, shrub and sub-shrubs can form shrub steppe. The common shrubs include green sotol (*Dasyllirion leiophyllum*), common sotol (*Dasyllirion wheeleri*), sacahuista (*Nolina microcarpa*), Texas sacahuista (*Nolina texana*), banana yucca (*Yucca baccata*), Torrey's yucca (*Yucca torreyi*), ocotillo (*Fouquieria splendens*), and resinbush (*Viguiera stenoloba*). Common sub-shrubs in foothill grasslands include mariola (*Parthenium incanum*), featherplume (*Dalea formosa*), lechuguilla (*Agave lechuguilla*), Harvard's century plant (*Agave harvardiana*), and plumed crinklemat (*Tiquilia greggii*). In foothill grasslands that have experienced drought, livestock grazing, and reduced fire occurrence, Chihuahuan Desert Scrub shrubs can invade and those species include viscid acacia (*Acacia neovernicosa*), tarbush (*Flourensia cernua*), catclaw mimosa (*Mimosa aculeaticarpa*), and turpentine bush (*Ericameria laricifolia*; ILAP STM 2012).

Lowland grasslands occur in basins and playas where during wet years surface waters collect. Dominant grasses in these areas include tobosagrass (*Pleuraphis mutica*), burrograss (*Scleropogon brevifolius*), alkali sacaton (*Sporobolus airoides*), big sacaton (*Sporobolus wrightii*), or vine mesquite (*Panicum obtusum*) with less abundant grasses including blue grama (*Bouteloua gracilis*) and bush muhly (*Muhlenbergia porteri*). Some desert lowlands have been invaded by desert scrub shrubs. The scrub species that have the potential to invade include honey mesquite (*Prosopis glandulosa*), creosotebush (*Larrea tridentata*), fourwing saltbush (*Atriplex canescens*), and tarbush (*Flourensia cernua*; ILAP STM 2012).

Sandy plains grasslands occur on sand sheet landforms. The dominant herbaceous species include black grama grass (*Bouteloua eriopoda*), sand dropseed (*Sporobolus cryptandrus*), mesa dropseed (*Sporobolus flexuosus*), spike dropseed (*Sporobolus contractus*), ear muhly (*Muhlenbergia arenacea*), and sand muhly (*Muhlenbergia arenicola*). Other less abundant grasses include blue grama (*Bouteloua gracilis*), purple three-awn (*Aristida purpurea*), low woollygrass (*Dasyochloa pulchella*), bush muhly (*Muhlenbergia porteri*), sandhill muhly (*Muhlenbergia pugens*), and giant dropseed (*Sporobolus giganteus*). Where the parent material produces gypsum soils, dominant vegetation includes gyp dropseed (*Sporobolus nealleyi*), gyp grama (*Bouteloua breviseta*), and New Mexico bluestem (*Schizachyrium neomexicanum*). Even though grasses are dominant, shrubs and sub-shrubs are always present and sometimes form shrub steppe. Indicator shrub species on sandy plains include Torrey's jointfir (*Ephedra torreyana*) and soap tree yucca (*Yucca elata*). Sandy plains grasslands that have suffered drought, livestock grazing, and lack of fire can be invaded by shrubs from the desert scrub. The desert scrub shrubs and sub-shrubs with the potential to invade include sand sagebrush (*Artemisia filifolia*), fourwing saltbush (*Atriplex canescens*), creosotebush (*Larrea tridentata*), honey mesquite (*Prosopis glandulosa*), frosted mint (*Poliomintha incana*), broom dalea (*Psoralea scoparioides*), littleleaf sumac (*Rhus microphylla*), and broom snakeweed (*Gutierrezia sarothrae*; ILAP STM 2012).

5.2.5 Wildlife and Habitat

Common mammals in the Chihuahuan Desert Grasslands include black-tailed jack rabbit (*Lepus californicus*), spotted ground squirrel (*Spermophilus spilosoma*), hispid pocket mouse (*Perognathus hispidus*), Ord's kangaroo rat (*Dipodomys ordii*), banner-tailed (*Dipodomys spectabilis*), Merriam (*Dipodomys merriami*), white-footed mouse (*Peromyscus leucopus*), cotton rat (*Sigmodon hispidus*, *Sigmodon fluviventer*), southern grasshopper mouse (*Onychomys torridus*), wood rat (*Neotoma micropus*, *Neotoma albigula*), badger (*Taxidea taxus*), and coyote (*Canis latrans*; Brown 1994).

Many bird species occur in the Chihuahuan Desert Grassland, common nesting birds include Swainson's hawk (*Buteo swainsoni*), prairie falcon, kestrel, mourning dove (*Zenaidura macroura*), scaled quail (*Callipepla squamata*), road-runner (*Geococcyx californianus*), burrowing owl, poor-will (*Phalaenoptilus nuttallii*), ladder-backer woodpecker (*Picoides scalaris*), western kingbird (*Tyrannus verticalis*), ash-throated flycatcher (*Myiarchus cinerascens*), Say's phoebe (*Sayornis saya*), horned lark, barn swallow (*Hirundo rustica*), white-necked raven (*Corvus cryptoleucus*), verdin (*Auriparus flaviceps*), cactus wren (*Campylorhynchus brunneicapillus*), mockingbird (*Mimus polyglottos*), curve-billed thrasher (*Toxostoma curvirostre*), black-tailed gnatcatcher (*Polioptila melanura*), loggerhead shrike (*Lanius ludovicianus*),

meadow lark (*Sturnella magna*, *Sturnella neglecta*), brown-headed cowbird (*Molothrus ater*), Scott's oriole (*Icterus parisorum*), house finch (*Carpodacus mexicanus*), lark sparrow (*Chondestes grammacus*), and Cassin's sparrow (*Aimophila cassinii*; Brown 1994).

Some animals in the Chihuahuan Desert Grasslands are visitors while others are endemic. Several animals are from nearby scrublands or other desertlands. These visiting species include Gambel's quail (*Lophortyx gambelii*), mule deer (*Odocoileus hemionus crooki*), and black-throated sparrow (*Amphispiza bilineata*). Javelina (*Dicotyles tajacu*) and white-tailed deer sometimes occur in ocotillo or thornscrub areas (Brown 1994). Species endemic to the Chihuahuan Desert Grasslands include scaled quail, western yellow box turtle (*Terrapene ornata luteola*), desert-grassland hognose snake (*Heterodon nasicus kennerlyi*), western hooknose snake (*Ficimia cana*), desert grassland whiptail (*Cnemidophorus uniparens*), southwestern earless lizard (*Holbrookia texana scitula*), and western green toad (*Bufo debilis insidiosus*; Brown 1994). In general, scrubland animals have been more successful than grassland animals this past century due to livestock reducing fire frequency by reducing grassland fuels, which helped scrub type vegetation increase (Brown 1994, Humphrey 1958). Antelope have largely disappeared from the desert grassland but javelina and mule deer have increased in distribution along with the increased distribution of scrublands (Brown 1994).

5.3 Chihuahuan Desert Grasslands State-and-Transition Model

State-and-transition models (STMs) conceptualize current and predicted ecological conditions ("states") of vegetation and soil based on our understanding of past interactions among ecological elements, processes, and stressors. Stressors are ecological drivers that are outside their natural range of variability and uncharacteristically push ecological conditions to a degraded state. These ecological stressors and states can serve as key ecological attributes for land management agencies to monitor and gauge subsystem conditions and trends.

The Chihuahuan Desert Grasslands STM has three parts (Figure 5-2, Figure 5-3, Figure 5-4), representing the ecological system and its dynamics on three landforms, Sand Plains, Piedmont, and Foothills and Lowlands (ILAP STM 2012). The ILAP identifies the ecological system as the "Chihuahuan Desert Semi-Grassland." The three STM diagrams follow a single set of diagram conventions. Rectangles represent ecological states of plant community composition. Arrows represent stressors that are ecological drivers outside their normal range of variation that transition ecological condition from one state to another. Columns represent life form and rows represent plant community composition. Conceptual models are not location specific and refer to large areas of land. Ecological states transition from grass and forb dominance on the left to shrub dominance on the right and from reference states with native vegetation at the top to modified states with dominance by exotic species at the bottom.

The Chihuahuan Desert Grasslands STM for all three landforms contains several grassland and shrubland states. Grassland states vary in grass species composition, specifically in the relative abundance of native versus nonnative grasses and in the relative abundance of perennial versus ruderal and annual grasses. Grassland states usually have at least some shrubs or shrub resprouts where shrub reduction treatments have been applied but were not severe enough to kill shrub roots (ILAP STM 2012). Shrubland states

have a herbaceous understory ranging from little to moderate amounts of biomass in relation to the level of shrub dominance (ILAP STM 2012).. Herbaceous states are on the left of each diagram, with shrub states on the right. Reference states with native vegetation are located at the top of each diagram and transition to states dominated by exotics species at the bottom (ILAP STM 2012).

Multiple stressors and processes influence transitions among Chihuahuan Desert Grasslands ecological states. These stressors include the Change Agents identified for assessment in the Chihuahuan Desert REA. The area covered by grasslands in the Chihuahuan desert has declined over the past several decades. Chihuahuan desert grasslands commonly have shrubs or sub-shrubs as a natural part of the plant community. However, drought, excessive livestock grazing, land development, the reduced role of fire, and climate change have allowed – and will continue to allow – desert scrub plants and some exotic plant species to invade and dominate the historic grasslands. Woody species expansion beyond historic levels has reduced grassland distribution in southern New Mexico in particular (Betancourt et al. 1993). Creosotebush has invaded grasslands from the lower elevations and juniper has invaded grasslands from the higher elevations (Grover and Musick 1990). The takeover of grasslands by shrubs changes ecological processes. As fine herbaceous plant materials are replaced by recalcitrant woody plant material, the processes of decomposition and nutrient cycling slow down. This further reduces fine fuels and the historic fire cycle (natural disturbance regime) where fire occurred prior to European settlement. The soil resources needed for plant growth become more unevenly distributed under shrubs, with fewer resources available in the interspaces between shrubs. Many shrubs have deeper roots systems than grasses, which affects soil water availability and rates evapotranspiration. The timing of active photosynthesis also differs between shrubs and grasses, with the result that the amount and timing of energy capture for an area changes with any shift from grasses to shrubs.

The following paragraphs discuss how each Change Agent affects Chihuahuan Desert Grassland states and transitions, as illustrated in Figure 5-2, Figure 5-3, Figure 5-4.

5.3.1 Climate Change

As discussed earlier in this chapter, the native and invading plant species of the Chihuahuan desert differ widely in their sensitivity to the frequency and duration of droughts and to the seasonal timing and magnitude of precipitation. Changes in these climate variables therefore will favor some plant species over others and influence plant community composition (for example, see Drought and Increased Winter Precipitation processes depicted in Figure 5-2). For example, an increase in night-time winter temperatures and a shift in precipitation seasonality towards more winter precipitation would be expected to favor C₃ shrubs over C₄ grasses (see earlier in this chapter), and therefore promote shrubland expansion (Brown et al. 1997, D’Odorico et al. 2010, ILAP STM 2012, Munson et al. 2013, Pennington and Collins, 2007).

Changes in wildfire and excessive grazing will also affect the way different plant species respond to climate change in the ecoregion. The climate of the Southwest over the next 100 years is expected to experience greater temperatures and more frequent extreme events such as drought (Kunkel et al. 2013a, 2013b; Melillo et al. 2014). The drier and hotter periods are expected to experience more wildfires (Melillo et al. 2014, Westerling et al. 2006) and shifts in the zones of suitable climate for

Chihuahuan Desert vegetation communities. Fire is especially detrimental to black grama grass during drought (Cable 1965, Drewa and Havstad 2001 Peters and Gibbens 2006), although brief, frequent droughts may favor perennial grasses over shrubs, as discussed earlier in this chapter (Burgess 1995). Drought also reduces the ability of shrubs to resprout after fire (McPherson, 1995). Plant species can take 50 years or longer to recover after fire, when they are also stressed by drought and excessive grazing (Canfield 1939, McPherson 1995, Nelson 1934, Reynolds and Bohning 1956).

5.3.2 Uncharacteristic Wildfire

The effects of altered fire frequency and/or intensity on Chihuahuan desert grasslands varies, depending on the initial state of the vegetation that burns, whether by a wildfire or a controlled (aka prescribed) burn applied during landscape restoration (see below). The STM diagrams below illustrate this complexity. For example, vegetation stands with grass understories that have not been lost to shrub dominance have enough fuels to carry fire but usually not sufficient fuels for a severe fire that would kill all shrub roots and convert a shrub dominated state into a grassland dominated state (ILAP STM 2012). If a severe wildfire did occur that killed the mature shrub roots, it would be considered a Treatment Shrub Lethal transition. A less severe wildfire that did not kill mature shrub roots would be a Fire Stand Replacing and Treatment Shrub Non-lethal transition. Invasion by shrub seedlings can initiate a transition towards a shrub-dominated state, but shrub seedlings do not resprout following fire. Fire in stands invaded by shrub seedlings therefore can return the plant community back to its initial grassland composition (see Fire Stand Replacing in Figure 5-2 and Figure 5-4). Fire Stand Replacing also applies when fire is used as a deliberate treatment, to delay the return of shrub dominance when some other, previous shrub control treatment did not kill mature shrub roots, as discussed below under Landscape Restoration (ILAP 2012).

The frequency and extent of fires in desert grasslands have declined since the 1880s (Bahre 1991, McPherson 1995). Prior to the late 1800s, desert grassland fire was common, although less common than in mixed or tall grass prairies (McPherson 1995, Wright and Bailey 1980). However, wildfire may not have significantly shaped Chihuahuan desert grasslands prior to European settlement (Buffington and Herbel 1965, Cable 1967; 1973, Cornelius 1988. The patchy distribution of vegetation simply may not have been highly conducive to the spread of large wildfires (Dick-Peddie 1993). Additionally, black grama grass, a common species in Chihuahuan desert grasslands, tolerates fire poorly, especially during drought (Cable 1965, Drewa and Havstad 2001, Peters and Gibbens 2006), and does not provide sufficient fine fuels to carry fire during dry years. Other factors than fire alone therefore must control where plant species grow in these kinds of desert grassland systems. Burgess (1995) suggests that drought is the dominant form of natural disturbance in the more arid portions of the desert grasslands, with fire and grazing the dominant forms of disturbance in the more mesic parts of the desert.

Grassland plant communities in different parts of the ecoregion therefore differ in their adaptations to fire and resilience to this type of disturbance, and these differences in adaptation affect how these communities respond to changes in the frequency and/or intensity of fire. The most arid portions of the Chihuahuan desert, including areas historically dominated by non-fire tolerant plant species, are less resilient to severe wildfire. Non-fire tolerant plants typically have reproductive structures exposed to

fire (e.g., near or above the soil surface) or have slow reproductive characteristics. In contrast, fire tolerant plant species in areas with historically sufficient precipitation to regularly produce sufficient fuels for frequent wildfires are more resilient to the historic fire regime. Fire tolerant plant species typically have reproductive structures protected from fire (e.g., below the soil surface) or have fast reproductive characteristics.

Plant species in fact differ widely in their strategies for surviving fire. Fire can kill up to 70% of succulents in the Chihuahuan Desert grassland (Bunting et al. 1980, Wright and Bailey 1982) including death related to increased vulnerability to insects (e.g., cactus bug; *Chelinidea vittiger*; McPherson 1995, Sickerman and Wangberg 1983). However, many succulents can sprout from rhizomes and roots after fire (Benson and Darrow 1981, Freeman 1973, Gentry 1972) or start growing again from live scattered fragments (Cable 1973, McPherson 1995, Thomas 1991. Many of the mature desert woody plants respond to fire by resprouting from buds that were protected from fire damage by soil, leaves, or bark (Gill 1977, McPherson 1995). The ability of many mature desert shrubs to survive fire allows them to grow quickly due to access to resources made available by the fire (McPherson 1995).

Growth form greatly affects the response of a grassland plant species to fire. The stoloniferous grasses of black grama and buffalo grass (*Buchloe dactyloides*) with growing points aboveground are more susceptible to fire injury especially if already stressed by drought and grazing (Cable 1975, Reynolds and Bohning 1956) than rhizomatous plants with growing points belowground more protected from fire injury (Humphrey, 1949; McPherson, 1995; Wright and Bailey, 1982). Grasses with growing points close to the surface of the soil (e.g., threeawns, *Aristida* spp.) or filled with lots of litter also are more likely to be injured by fire than plants with deeper growing points and less aboveground biomass to burn (McPherson, 1995; Wright, 1971). Spatial distribution of plant species also influences their response to fire. Grass species that grow in shrub mounds are more likely to suffer fire injury due to more heat being released during the burning of the greater fuel loads (McPherson, 1995) but without fire, shrubs may help protect sub-canopy grasses (e.g., bush muhly, *Muhlenbergia porteri*) from grazing.

A review of fire records back to 1528 suggests that wildfire helps maintain Chihuahuan desert grasslands by limiting shrub dominance (Humphrey, 1958; McPherson, 1995). The historic mean fire return interval in desert grasslands has been suggested as 9-10 years (Cable, 1967; McPherson, 1995; Peters and Gibbens, 2006) in areas that had fire as part of the historic disturbance regime. The most likely fire season would have been during June when lightning frequency is highest (Gosz et al., 1995) and when the vegetation is dry during periods of high winds, low humidity, and high temperatures.

The frequency and size of fires in Chihuahuan desert grasslands have declined since the 1880s (Bahre 1991, McPherson 1995). Prior to the late 1800s, desert grassland fire was common although less common than in mixed or tall grass prairies (McPherson 1995, Wright and Bailey 1980). Some fires burned more than a hundred square miles before 1882 (Bahre 1991, Humphrey 1949, McPherson 1995).

Shrubs did not dominate desert grasslands prior to the 1880s because most of the shrubs in desert grasslands can be killed by fire during their seedling phase (Bock and Bock 1992a, Cable 1967, Cox et al. 1993, Glendening and Paulsen 1955, Humphrey 1949, McPherson 1995, Reynolds and Bohning 1956,

Wright et al. 1976) without resprouting (McPherson 1995). Many of these shrubs do not produce seed until 10+ years of age (Chew and Chew 1965, Humphrey 1958, Martin 1975, McPherson 1995). More specifically, fire can limit mesquite encroachment into grasslands because fire can kill mesquite seeds and seedlings (Brown and Archer 1999, Cox et al. 1993, Peters and Gibbens 2006). However, mature mesquite is not usually killed during fire and resprouts prolifically following fire or cutting (Peters and Gibbens 2006). A mean fire return interval of 7-10 year would maintain an open grassland state without many shrubs (Griffiths 1910, Leopold 1924, McPherson 1995, Schmutz et al. 1985, Wright and Bailey 1982).

The effects of fire in association with other ecological drivers can have long lasting results on the structure of desert grasslands (McPherson 1995). While fire typically reduces plant biomass production and cover for between 1 and 3 years given sufficient post-burn moisture (Bock et al. 1976, Bock and Bock 1992a, Cable 1967, Humphrey 1949, McPherson 1995, Reynolds and Bohning 1956), drought or grazing following fire greatly lengthens vegetation recovery time (McPherson 1995). Fire is especially detrimental to black grama grass during drought (Cable 1965, Drewa and Havstad 2001, Peters and Gibbens 2006). Plant species can take 50 year or longer to recover after fire due to drought and grazing stress (Canfield 1939, McPherson 1995, Nelson 1934, Reynolds and Bohning 1956). The effects of fire on black grama grass are also influenced by soil moisture during fire, post-burn precipitation, grazing intensity (Gosz and Gosz 1996), and many other factors including fuel loads, fire residence time, soil heating, wind, and temperature (Peters and Gibbens 2006).

Season influences the ability of plants to survive fire. The season when vegetation burns influences desert grassland composition because actively growing plants are more easily injured by fire than dormant plants (Cable 1965, 1967, 1973; McPherson 1995). Early summer fires, for example, are more damaging to warm season plants than fall-spring fires (Cable 1965, 1967, 1973; McPherson 1995). As an example, fire in June can be more damaging to mesquite than fire in November (McPherson 1995). However, there may be some exceptions. Summer fire has resulted in big sacaton (*Sporobolus wrightii*) and buffelgrass (*Pennisetum ciliare*) producing more biomass than fire in fall and winter (Cox et al. 1990, McPherson 1995). Changes in the seasonal distribution of fire therefore has the potential to significantly alter plant community composition.

The environmental conditions following fire often influence roots more than do the direct effects of the fire itself (McPherson 1995). Following fire, soil temperatures and plant available nitrate can be elevated for weeks to months, resulting in increased growth of surviving plants (Raison 1979), although burned areas often have drier soils that can drought-stress shallow-rooted plant species (McPherson 1995, Wright and Bailey 1982). Along with potentially warmer soils and increased nitrate, the burned area can experience greater sunlight and a wider temperature range (Cable 1967, McPherson 1995). These site characteristics can potentially improve the establishment of new plants that have seeds that easily disperse from undisturbed areas into burned areas (Cable 1967, McPherson 1995). Examples of these types of plant species include the annual grasses of six-weeks needle grama (*Bouteloua aristidoides*) and six-weeks threeawn (*Aristida adscensionis*; Cable 1967, McPherson 1995) or the perennial exotic Lehmann lovegrass (*Eragrostis lehmanniana*; McPherson 1995, Ruyle et al. 1988, Sumrall et al. 1991).

Lehmann lovegrass is more fire tolerant than several grasses native to the Chihuahuan desert (McGlone and Huenneke 2004). As discussed further below (see Landscape Restoration), managers may worry that burning in areas where Lehman lovegrass is present may increase this invasive, exotic grass; however, as also noted above some research suggests that fire does not result in the spread of Lehman lovegrass (McDonald and McPherson 2011, McGlone 2013). These findings suggest that fire may be an option as a vegetation management tool in areas with Lehmann lovegrass, again as noted above.

5.3.3 Invasive Species

A major land management issue in the southwestern U.S. is shrub or woodland invasion and the conversion of grasslands to shrublands by several species including tarbush (*Flourensia cernua*), whitethorn (*Acacia neovernicosa*), creosotebush (*Larrea tridentata*), mesquite (*Prosopis* species), and juniper (*Juniperus monosperma*) invasion (Brown 1994, ILAP STM 2012). Woody species invasion of grasslands during recorded history has been reported by numerous authors (Brown 1994, Castetter 1956, Hastings and Turner 1965, Humphrey 1958, Leopold 1924, Parker and Martin 1952).

Mesquite is a C₃ deciduous shrub that can live 200 years and develops a large root system (Gibbens and Lenz 2001, Peters and Gibbens 2006). Grazing and fire suppression have allowed mesquite to change grassland states into shrubland states (ILAP STM 2012). Mesquite shrublands typically occur on deep sands with a calcium carbonate soil layer (Peters and Gibbens 2006). Windblown sand collects around mesquite shrubs developing coppice dunes (Langford 2000). Mesquite shrubs usually comprise 30-55% of total cover in mesquite shrublands (Paulsen and Ares 1962) with other commonly occurring plants of saltbush (*Atriplex canescens*), broom snakeweed, dropseed and threeawn grasses, and various forbs (Peters and Gibbens 2006).

Several other scrub plant species currently and historically have been a part of the Chihuahuan Desert grasslands but now occur in greater density than before (Brown 1994). These plant species include lotebush (*Zizyphus obtusifolia*, *Condalia spathulata*), allthorn (*Koeberlinia spinosa*), Mormon tea (*Ephedra trifurca*, *Ephedra antisyphilitica*), mimosa (*Mimosa biuncifera*, *Mimosa dysocarpa*), false mesquite (*Calliandra eriophylla*), Wright's beebush (*Aloysia wrightii*), catclaw acacia (*Acacia greggii*), littleleaf sumac (*Rhus microphylla*), desert hackberry (*Celtis pallida*), javelina-bush (*Condalia ericoides*), barberry (*Berberis trifoliata*), and Ocotillo (*Fouquieria splendens*) (Brown 1994).

Invasive plants have traits that make them more adapted to an area than the previous vegetation. Buffelgrass (*Pennisetum ciliare*) distribution previously had been limited by cold winter temperatures but a new variety of buffelgrass has been developed and planted south of Arizona. This new variety may increase the extent of land (e.g., cooler or higher elevation) that buffelgrass can inhabit and potential invade (Archer and Predick 2008). The ability of Lehmann lovegrass (*Eragrostis lehmanniana*) to compete with native grasses is likely due to its ability to make more use of winter precipitation and better tolerate drought than some of the species (Archer and Predick 2008). Lehmann lovegrass has other impressive characteristics that help facilitate its ability to maintain its current extent and spread into new territories. Some of these traits include prolific production of seeds that are easily spread by water or wind, the ability to produce multiple seed crops in a year, and the production of seeds that have different dormancy periods. This spreads germination out over time increasing the chances of at

least some seeds establishing when the environment is conducive to plant establishment (Humphrey 1994).

Invasion by exotic plant species into the Chihuahuan Desert varies by location (see Exotic Invasion in Figure 5-2 to Figure 5-4; ILAP STM 2012). Scrublands are the least susceptible to exotic plant invasion whereas, grasslands on piedmont or foothill landforms are more easily invaded (ILAP STM 2012). Uninvaded areas near existing exotic plants are more likely to be invaded than areas far away from exotic plants due to the distance propagules would need to travel (ILAP STM 2012). The exotic plant species of *Eragrostis lehmanniana*, *Eragrostis cilianensis*, *Erodium cicutarium*, *Fumaria parviflora*, *Salsola tragus*, and *Sisymbrium irio* are found in the southwestern US and northern Mexico region (Van Devender et al. 2013). The exotic African buffelgrass (*Pennisetum ciliare*) also can be an invasive plant that strongly competes for water and nutrients and helps carry fire in non-fire prone areas (Van Devender et al. 2013). Buffelgrass occurs in both the southern New Mexico and west Texas portions of the Chihuahuan Desert (USDA, NRCS 2014). It occurs mainly below 1060 m in elevation and is a serious problem in the Big Bend area of Texas within the Chihuahuan Desert (Van Devender et al. 2013). A state-wide list of noxious weeds is available at <http://www.nmda.nmsu.edu/apr/noxious-weed-information/> for New Mexico and http://texasinvasives.org/plant_database/ for Texas.

5.3.4 Development

Chapters 2 and 3 summarize information on the types and extent of development in the U.S. portions of the Chihuahuan desert. The spatial distributions of the different types of development in the ecoregion reflect the interaction of several geographic and economic factors. Grassland habitat has been replaced in several areas of the ecoregion by expansions of commercial development along the U.S.-Mexico border, most notably in the El Paso/Juarez area; expansions of military and security activities and infrastructure along the border and around military facilities; and expansions of residential and commercial development around cities and towns. (Expansions of oil and gas production and distribution systems in and around the Permian Basin across large portions of the east side of the ecoregion have affected scrubland more than grassland—see Chapter 6). Transportation infrastructure developments associated with these expansions into former grasslands also fragment these grasslands. All these developments increase air pollution, risks of accidental fires, and pressures on wildlife; and associated demands on water supplies can lower water tables, potentially eliminating some natural watering holes for wildlife that may also have supported patches of distinctive vegetation. Chapters 12-17 provide additional information on the impacts of development on Chihuahuan desert grasslands wildlife – i.e., impacts to pronghorn, mule deer, banner-tailed kangaroo rat, lack-tailed prairie dog, grassland birds, and grassland small mammals and their habitat.

5.3.5 Excessive Domestic Grazing

Excessive livestock grazing alters ecological states in Chihuahuan desert grasslands by reducing the biomass of highly palatable plant species or in extreme instances eliminating these species entirely (see Livestock Grazing in Figure 5-2 to Figure 5-4; ILAP STM 2012). The remaining plants have less photosynthetic tissue to support growth and less energy to extend roots to collect soil resources. The unused resources that formerly were used by these grasses then can be used by the less palatable plant

species including woody species, which can then increase in size and the area they cover. In this way, excessive livestock grazing reduces the health and ability of historic plant communities to resist invasion by aggressive plant species and can result in reduced grassland distribution. Grazing pressure may lessen after the more palatable plants are greatly reduced or eliminated. However, excessive livestock grazing can change community composition sufficiently to reduce the production of propagules (e.g., seeds, stolons, rhizomatous roots) by the native species of the community, and thereby reduce the ability of the historic plant community to recover after the grazing pressure lessens. Excessive grazing has reduced the more palatable grasses in many places in the Chihuahuan desert grasslands, leaving the less palatable grasses that include hairy tridens (*Tridens pilosus*), fluffgrass (*Dasyochloa pulchella*), red three-awn (*Aristida longiseta*), and burrograss (*Scleropogon brevifolius*; Brown 1994).

Excessive livestock grazing in combination with drought and/or increased winter precipitation (see Induced Shrub Invasion in all three diagrams) together can cause Chihuahuan desert grasslands to transition to shrubland states. Excessive livestock grazing alone does not cause this transition (ILAP STM 2012), although some authors have suggested that it alone can cause black grama grasslands to transition to mesquite dunelands (Campbell 1929, Dick-Peddie 1993). An example of excessive livestock grazing influencing the transition between states is the transition from perennial grassland states to ruderal (early successional) grassland states (Belsky 1992, Berg et al. 1997, Drewa and Havstad 2001), which can happen in all Chihuahuan Desert Grasslands and Scrub subsystems (ILAP STM 2012). Livestock can also spread mesquite seeds from invaded areas to nearby areas and thereby encourage transitions from grassland states to mesquite-dominated shrubland states (see Grazing Mesquite Invasion in Figs 2-3) or from shrubland states to mesquite-dominated shrubland states in piedmont and sandy plains subsystems (see Grazing Mesquite Invasion Shrub in Figure 5-2 and Figure 5-3) (ILAP STM 2012). These latter transitions often result in further degradation, because mesquite shrubs do not provide sufficient protection of the soil and thereby allow the surface soil to erode. Where the top-soil horizon erodes away, the lower soil horizons often cannot support the historic biodiversity and plant community composition because of less suitable soil characteristics including less organic matter, lower nutrient availability, and sometimes less water holding capacity. These changes result in degraded nutrient and water cycles.

5.3.6 Landscape Restoration

Reducing shrub encroachment into grasslands is an important management concern in the U.S. portions of the Chihuahuan desert. Although many grasslands in the ecoregion had some historic shrub component, shrubs did not dominate Chihuahuan desert grasslands prior to the 1880s. Most of the shrubs in Chihuahuan desert grasslands can be killed by fire during their seedling phase (Bock and Bock 1992a, Cable 1967, Cox et al. 1993, Glendening and Paulsen 1995, Humphrey 1949, McPherson 1995, Reynolds and Bohning 1956, Wright et al. 1976) without resprouting (McPherson 1995). Many of these shrubs do not produce seed until 10+ years of age (Chew and Chew 1965, Humphrey 1958, Martin 1975, McPherson 1995). Additionally, exotic plants historically were not a problem until foreign explorers and transoceanic trade arrived in the Americas. Since European settlement, shrubs and invasive plants have threatened historic plant communities. A lack of fire in these desert grasslands where it was a historic natural disturbance (noting that some areas did not have sufficient fuel for fire) can allow shrubs to

dominate and thereby reduce herbaceous biomass and alter grassland vegetation structure and function (Burgess 1995, McAuliffe 1994, McPherson 1995). Shrub invasion reduces the availability of fine surface fuels for wildfire, making it difficult to reintroduce fire as a shrub management tool (McPherson 1995). Under these conditions, fire is not sufficient to kill mature shrubs and land managers would need to use chemical, mechanical, or other tools to return shrubland states back to grassland states (McPherson 1995).

Creosotebush and mesquite are two widespread shrub species. Fire can limit mesquite encroachment into grasslands because fire can kill mesquite seeds and seedlings (Brown and Archer 1999, Cox et al. 1993, Peters and Gibbens 2006). However, mature mesquite is not usually killed during fire and resprouts prolifically following fire or cutting (Peters and Gibbens 2006). A mean fire return interval of 7-10 years is needed to maintain an open grassland state without many shrubs (Griffiths 1910, Leopold 1924, McPherson 1995, Schmutz et al. 1985, Wright and Bailey 1982).

A common invasive grass, Lehmann lovegrass, degrades plant communities and may warrant restoration efforts depending on management goals. This grass is more fire tolerant than several native grasses in the Chihuahuan Desert (McGlone and Huenneke 2004). Some managers worry that burning in areas where Lehman lovegrass is present may increase this invasive, exotic grass; however, some research suggests that fire does not result in the spread of Lehman lovegrass (e.g., (McDonald and McPherson 2011, McGlone 2013). These findings suggest that fire may be an option as a vegetation management tool in areas with Lehmann lovegrass.

Restoring land invaded by Lehmann lovegrass is a long process with multiple steps including developing a long-term restoration plan with an integrated management approach combined with post-treatment monitoring (USFS 2014). Key steps to restoring land invaded by Lehman lovegrass include the following (USFS 2014):

- Maintain diverse and vigorous native plant communities including using the best management practices necessary to achieve healthy and resilient plant communities.
- Map and track weed infestations over time.
- Plan an eradication strategy focusing on the most critical areas.
- Apply appropriate combination of chemical, mechanical, cultural, and biological weed control methods.
- Continue long-term monitoring and re-apply control treatments as appropriate.

Additional directions on planning and implementing restoration efforts aimed at reducing Lehmann lovegrass are presented in (USFS 2014).

A variety of landscape restoration treatments have been applied to millions of acres in New Mexico (BLM 2015), including thousands of acres in the Chihuahuan desert. Attempts to improve plant community composition have not always been successful and the application of treatments sometimes disturbs the soil and aids weedy plant species that can help transition a plant community to an exotic state (see Treatment Soil Disturbance in Figure 5-2 to Figure 5-4; ILAP STM 2012). Treatments, including applications of fire, chaining, and herbicides, are commonly applied to return shrub-invaded states back

to grassland states. Treatments applied at intense levels that are severe enough to kill shrub roots can cause a shrub-dominated state to transition back to a grassland state (see Treatment Shrub Lethal in Figure 5-2 to Figure 5-4; ILAP STM 2012). Less severe treatments do not kill shrub roots and only set the plant community back to a grassland state, leaving live shrub roots that can resprout (see above and also Treatment Shrub Non-lethal in Figure 5-2 to Figure 5-4). This leads to a faster return of shrub cover and need for more frequent treatments (see Shrub Invasion Post-treatment in Figure 5-2 to Figure 5-4; ILAP STM 2012). A similar transition also can occur when livestock grazing, drought, and increased winter precipitation aid this return of shrub dominance from surviving roots (see Induced Shrub Invasion-Post-treatment in Figure 5-2 to Figure 5-4). However, treatments are not always needed to improve ecological condition. Without treatment, perennial grasses can invade ruderal grassland states when the disturbance regimes (e.g., fire frequency) are not too severe and help convert a ruderal grassland state to a perennial grassland state (see Perennial Invasion in Figure 5-2 to Figure 5-4).

Several types of land reclamation approaches have been studied over the decades at or near the Jornada Experimental Range in the Chihuahuan desert (Rango et al. 2002). Many of these vegetation treatments were implemented around the 1930s and often included help from the Civilian Conservation Corps prior to World War II (Rango et al. 2002). Many of the treatment details related to these studies were not sufficiently recorded or are missing. However, retrospective analyses of aerial or satellite imagery reveal the long-term effects, or lack thereof, for several types of treatments. In general, contour terraces, brush water spreaders, strips grubbed free of shrubs, and mechanical root plowing and seeding did not have long-term desired treatment outcomes (Rango et al. 2002).

The analyses of the data on land reclamation at and near the Jornada Experimental Range suggest several reasons why so many treatment methods did not have the desired long-term benefits. Experimental treatments often were not applied at sufficiently large extents resulting in landscape scale hydrological and ecological processes overwhelming small treatment areas (Rango et al. 2002). Treatments were not maintained over time and this resulted in the loss of initial treatment effectiveness. Treatments sometimes were located in the wrong area to find successful outcomes from experimental treatments (Rango et al. 2002).

These findings near the Jornada Experimental Range have implications for land managers. Historical aerial and satellite imagery should be evaluated prior to applying vegetation treatments to ensure that treatments are of sufficient size to not be dominated by hydrological or ecological processes from nearby lands and that treatments are appropriately located on the desired overall landscape position away from decades earlier or current nearby factors that would lessen treatment effectiveness. These recommendations are based on the findings that some factors are more readily apparent from an aerial perspective than from the ground. These findings also suggest that long-term plans for monitoring, accurate record keeping, and repeat treatments are essential to long-term treatment success. Even though several types of treatments were unsuccessful, some experimental treatments did have long-term vegetation benefits. The treatments of water retention dikes, some fence lines, and some boundaries between disparate management practices displayed long-term treatment effectiveness (Rango et al. 2002).

Climate change can limit restoration success when the current, modified temperature and precipitation patterns no longer favor the historic plant community. Climate change is more fully addressed in the climate related sections of the REA reports. In addition, soil erosion following severe disturbances such as land development, severe wildfire, or conversion to mesquite dunelands can alter grassland distribution and landscape restoration potential. Where the top soil horizon erodes away, the lower soil horizons often cannot support the historic biodiversity and plant community composition because of less suitable soil characteristics including less organic matter, nutrient availability, and sometimes less water holding capacity. The ability to re-establish a desired plant community following degradation has been reviewed in (Cox et al. 1982).

5.4 Chihuahuan Desert Grassland Key Ecological Attributes

The variables that distinguish the ecological states from each other in the Chihuahuan Desert Grasslands STMs all constitute key ecological attributes for the system. Additional key ecological attributes include critical aspects of soil condition, wildfire, and soil moisture. The list below identifies eight (8) key ecological attributes for Chihuahuan Desert Grasslands CE based on these criteria. *Characterizing the present condition of a system requires data on indicators for its key ecological attributes.* Indicator sources for these key ecological indicators include: Ecological Site Descriptions developed by the U.S. Department of Agriculture, Natural Resources Conservation Service (USDA NRCE 2006) for ESD R042XB014NM Loamy, R042XB035NM Gravelly Loam, R070DY154NM Swale; LANDFIRE (2014) Biophysical Setting 2515030 Chihuahuan Loamy Plains Desert Grassland; and expert knowledge.

- **System State**
 - Perennial-Ruderal-Invasive Grass Species Composition and Age Structure
 - Grass versus Shrub Composition and Age Structure
 - Woody Species Composition, Abundance, and Age Structure
 - Vegetation Patch Size and Fragmentation
 - Soil Exposure and Stability
- **Critical Dynamics**
 - Wildfire Dynamics
 - Soil Moisture Dynamics
 - Soil Erosion

Figure 5-2. Chihuahuan Desert Grassland-Sandy Plains Landform state-and-transition model.

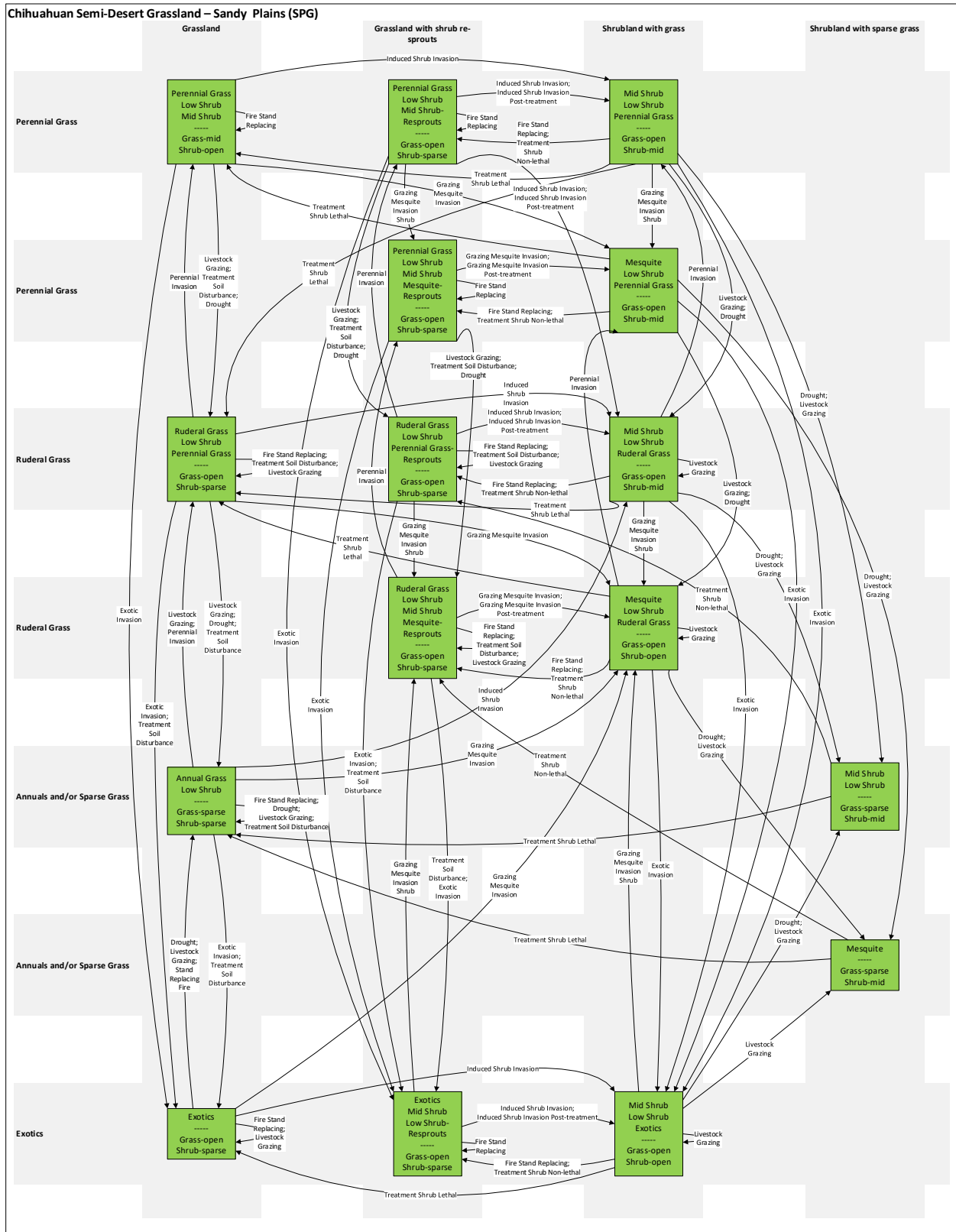


Figure 5-3. Chihuahuan Desert Grassland-Piedmont Landform state-and-transition model.

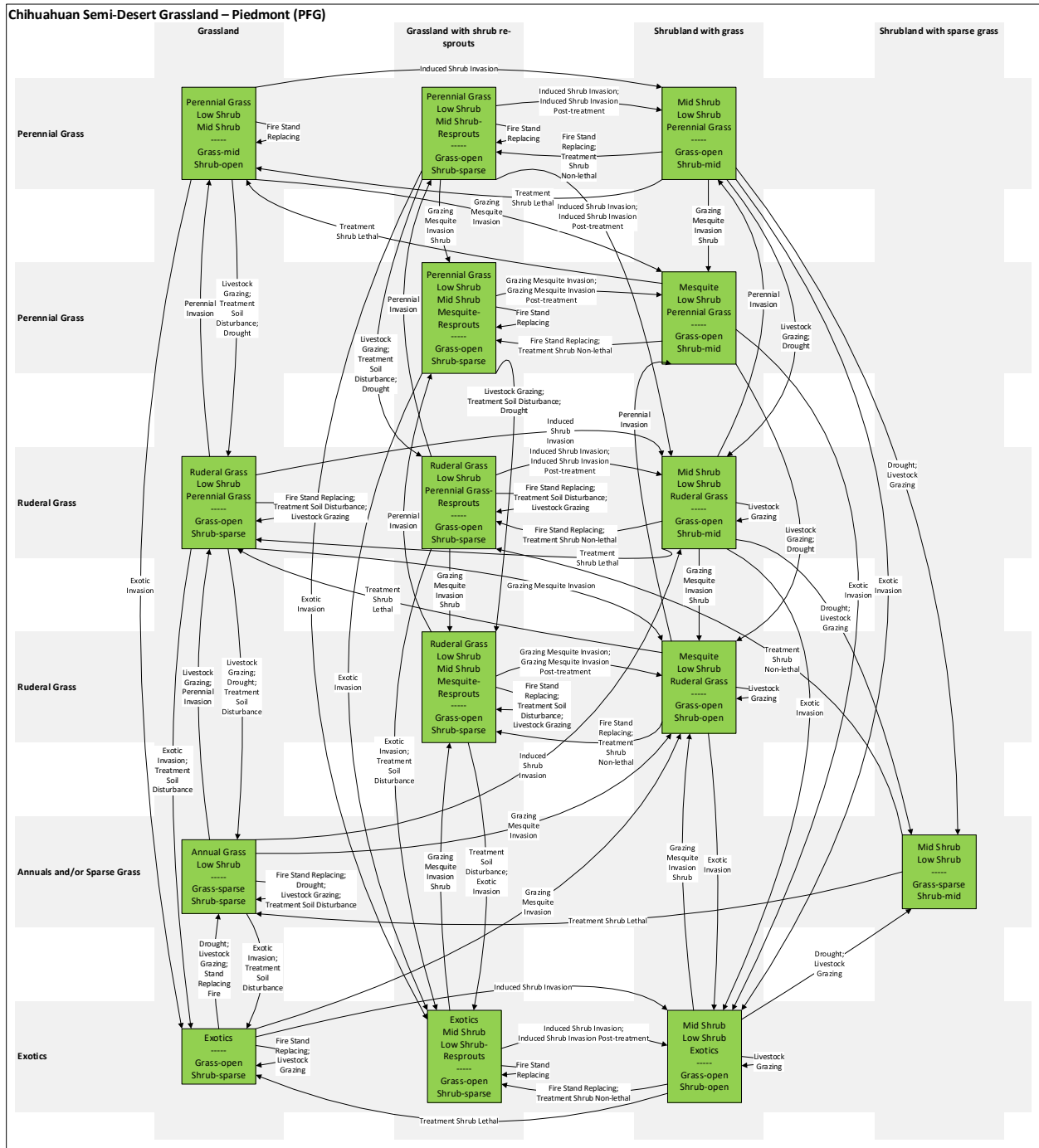
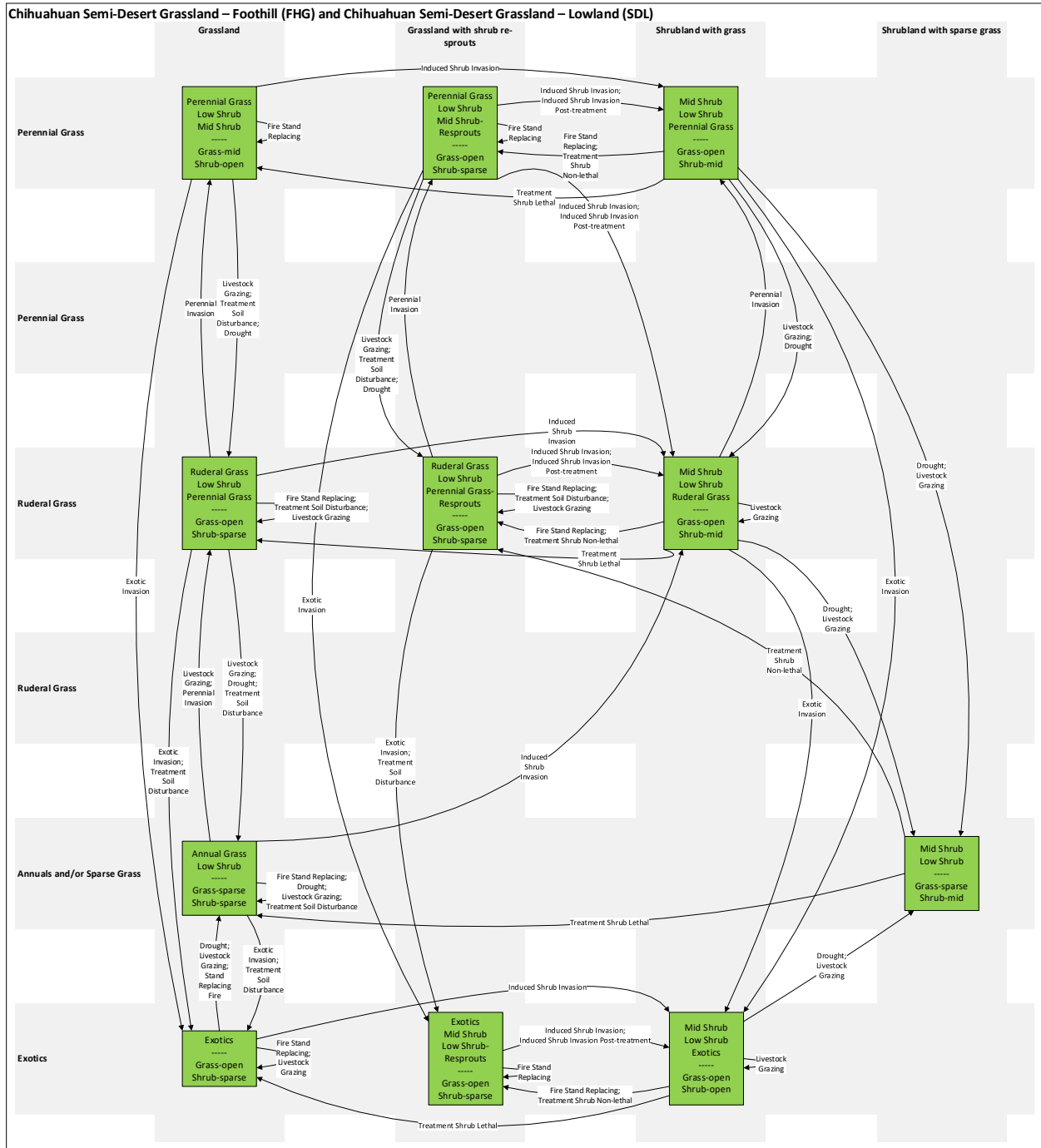


Figure 5-4. Chihuahuan Desert Grasslands Subsystem-Foothill and Lowland Landforms state-and-transition model.



6 Chihuahuan Desert Scrub Conceptual Model

This chapter presents the conceptual ecological model for the Chihuahuan Desert Shrub CE. The presentation follows the structure described in Chapter 4.

6.1 Sources of Conceptual Models and Vegetation Classification

The Chihuahuan Desert Scrub conceptual model is adapted from the Integrated Landscape Assessment Project (ILAP, Gaines et al. 2013). Natural Heritage New Mexico (NHNM), a division of the Museum of Southwestern Biology at the University of New Mexico, developed the original conceptual models for ILAP using the vegetation dynamics development tool (VDDT, ESSA Technologies Ltd.). The adaptation included converting the VDDT model diagram into a state-and-transition conceptual model (STM) diagram similar in style to the hierarchical approach of Miller et al. (2010) using Microsoft Visio software. The Chihuahuan Desert Scrub conceptual model specifically incorporates information from the Chihuahuan Desert Scrub and Chihuahuan Salt Desert Scrub STMs (ILAP STM 2012). These STMs are not location specific but rather refer to large areas of land with the same general potential natural vegetation type (PNVT). PNVT is largely determined by biogeochemical and disturbance regime constraints.

The Chihuahuan Desert Scrub ecological system is ecologically important and has been classified by other organizations. As noted in Chapter 3, the term, “ecological system” here refers to “... recurring groups of biological communities that are found in similar physical environments and are influenced by similar dynamic ecological processes, such as fire or flooding” (Comer et al. 2003). NatureServe refers to parts of this CE as Chihuahuan Creosotebush Desert Scrub (International Ecological Classification Code CES302.731) and Chihuahuan Mixed Desert and Thornscrub (CES302.734), with Chihuahuan Mixed Salt Desert Scrub (CES302.017) in the more saline areas (NatureServe 2014). The LANDFIRE classification approach, related to the NatureServe approach, uses the same classification names but assigns them as biophysical settings 2510740, 2511001, and 2510750, respectively (LANDFIRE 2014). The gravelly ecological site description (R042XB010NM) developed by the U.S. Department of Agriculture, Natural Resources Conservation Service (USDA NRCS 2006) also applies to the Chihuahuan Desert Scrub ecological system. The present Chihuahuan Desert Scrub conceptual model incorporates information from these publications as well.

6.2 Chihuahuan Desert Scrub Control Model

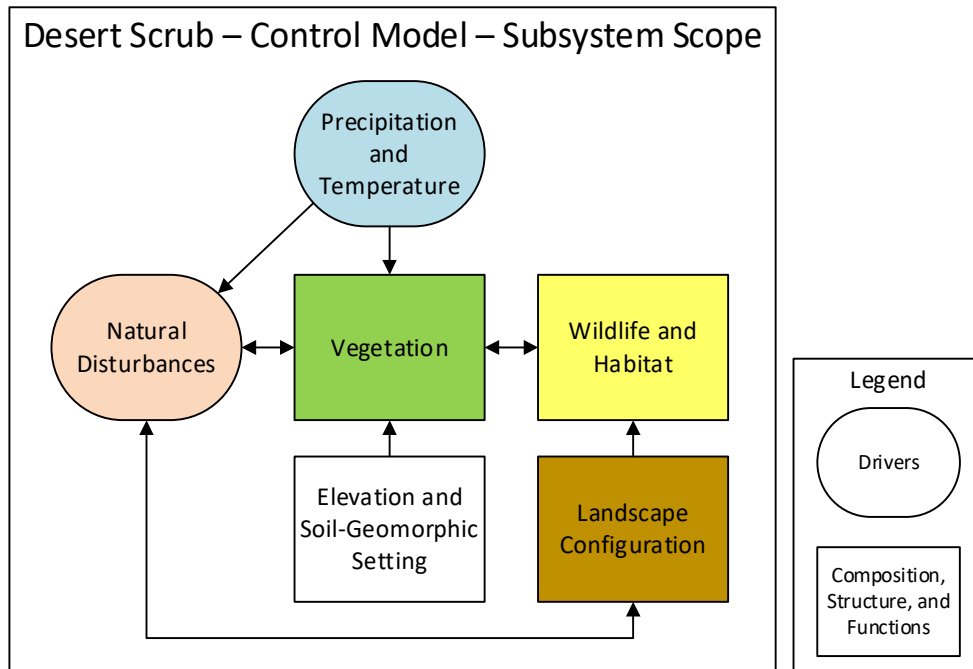
The Chihuahuan Desert Scrub control model (Figure 6-1) illustrates current understanding of how the main ecological components and controlling processes for this ecological system. The ecological components of the control model include landscape configuration, elevation and soil-geomorphic setting, precipitation and temperature, vegetation, natural disturbance regimes, and wildlife and habitat. The following paragraphs explain these individual model components.

6.2.1 Landscape Configuration, and Elevation and Soil-Geomorphic Setting

Chihuahuan Desert Scrub in the U.S. portions of the ecoregion occurs at lower elevations (CDRI 2014) on

multiple landforms from basin floors up into piedmont alluvial fans and foothills (ILAP STM, 2012). Much of the scrub occurs over limestone parent material (Bown 1994). Salt desert scrub, a sub-type of the larger scrub system, occupies basin bottom landforms with clay accumulations and high salt content, often adjacent to playas (ILAP STM 2012).

Figure 6-1. Chihuahuan Desert Scrub control model.



6.2.2 Precipitation and Temperature

Chihuahuan Desert Scrublands receive approximately 200-300 mm annually (Brown 1994). Air temperatures range from below freezing to more than 40 °C with a 200-250 day growing season between frosts (Brown 1994). More than half of the precipitation falls between May and September during summer thunderstorms high inter-annual variability (Brown 1994).

6.2.3 Natural Disturbances

Fire and drought are common sources of natural disturbance across the ecoregion. However, desert scrub communities typically lack sufficient fuels to sustain fire – a significant contrast with grasslands (see Chapter 5), in which grass understories have enough fuels to carry fire but not sufficient fuels for a severe fire that would kill all shrub roots and convert a shrub dominated state into a grassland dominated state (ILAP STM 2012). The dominance of grasslands by C₄ plants across southern New Mexico is largely a result of the warm growing seasons with low and varying amounts of rain mixed with periodic drought (Peters and Gibbens 2006, Schmutz et al. 1992).

6.2.4 Vegetation

The most common dominant plant species of the Chihuahuan Desert Scrub is creosotebush (*Larrea tridentata*) commonly occurring with tarbush (*Flourensia cernua*) (ILAP 2013). Creosotebush typically

accounts for 28-45% of total cover in these areas (Paulsen and Ares 1962, Peters and Gibbens 2006). Other potentially dominant shrubs include whitethorn acacia (*Acacia constricta*), viscid acacia (*Acacia neovernicosa*), Rio Grande saddlebush (*Mortonia scabrella*), and ocotillo (*Fouquieria splendens*). Common sub-shrubs that may also codominate include lechuguilla (*Agave lechuguilla*), cactus apple (*Opuntia engelmannii*), Wright's beebrush (*Aloysia wrightii*), and mariola (*Parthenium incanum*). Other common sub-shrub species include broom snakeweed (*Gutierrezia sarothrae*), picklyleaf dogweed (*Thymophylla acerosa*), plumed crinklemat (*Tiquilia greggii*), and mat rockspirea (*Petrophytum caespitosum*) (ILAP STM 2012). Key herbaceous species include fluffgrass (*Dasyochloa pulchella*) and bush muhly (*Muhlenbergia porter*). Other grasses that can be common in desert scrub include black grama grass (*Bouteloua eriopoda*), tobosagrass (*Pleuraphis mutica*), and burrograss (*Scleropogon brevifolius*) (ILAP STM 2012).

Chihuahuan Salt Desert Scrublands, a sub-type or variant of the larger Scrub system type, occur on soils with high salt content are usually sparsely vegetated (ILAP STM 2012). The characteristic shrub species include fourwing saltbush (*Atriplex canescens*), tarbush (*Flourensia cernua*), and littleleaf sumac (*Rhus microphylla*). Typical herbaceous species include vine mesquite (*Panicum obtusum*), tobosagrass (*Pleuraphis mutica*), alkali sacaton (*Sporobolus airoides*), big sacaton (*Sporobolus wrightii*), burrograss (*Scleropogon brevifolius*), and sometimes bush muhly (*Muhlenbergia porter*) (ILAP STM 2012).

Succulent-scrub plants are more common in scrublands at higher elevations in the Chihuahuan desert, before the scrublands transition to desert grasslands. Succulent-scrub plants in this system include candelilla (*Euphorbia antisyphilitica*), bromeliad (*Hechtia scariosa*), lechuguilla (*Agave lechuguilla*), ocotillo, dogweeds (*Dyssodia pentachaeta*, *D. acerosa*), ratany (*Krameria parvifolia* var. *glandulosa*), snakeweed (*Condalia* spp.), and several others (Brown 1994). Other succulent plants include various yuccas (e.g., *Yucca elata*, *Y. rostrata*, *Y. thompsoniana*, *Y. filifera*, *Y. carnerosana*, *Y. torreyi*), sotol (e.g., *Dasyilirion leiophyllum*, *D. wheeleri*), agaves (*Agave scabra*, *A. falcata*, *A. neomexicana*, *A. parryi*, *A. striata*), and nolina (*Nolina microcarpa*, *N. erumpens*, *N. texana*). Several woody shrubs and cati also coexist in the upper succulent-scrub including ocotillo, coldenia (*Coldenia greggii*), catclaw (*Acacia greggii*), cenizos (*Leucophyllum minus* L. *frutescens*), condalia (*Condalia* spp.), *Zizyphus obtusifolia*, lippia or oreganillo (*Aloysia wrightii*), little leaf sumac (*Rhus microphylla*), and other locally dominant plant species (Brown 1994). The Chihuahuan Desert has more than 1000 endemic plant species (Brown 1994, Johnston 1977).

6.2.5 Wildlife and Habitat

The Chihuahuan Desert Scrub hosts several warm-blooded vertebrates including desert pocket gopher (*Geomys arenarius*), yellow-faced pocket gopher (*Pappogeomys castanops*), Nelson's kangaroo rat (*Dipodomys nelsoni*), Nelson's pocket mouse (*Perognathus nelsoni*), southern grasshopper mouse (*Onychomys torridus*), Goldman's woodrat (*Neotoma goldmani*), Texas antelope squirrel (*Ammospermophilus interpres*), and desert pocket mouse (*Perognathus penicillatus*). Other species that occur in the scrub include the desert shrew (*Notiosorex crawfordi*), desert mule deer (*Odocoileus hemionus crooki*), pronghorn (*Antilocapra americana*), desert bighorn sheep (*Ovis canadensis mexicana*), Merriam's kangaroo rat (*Dipodomys merriami*), and desert cottontail (*Sylvilagus auduboni*; Brown 1994).

Chihuahuan Desert Scrub birds are not exclusive to the scrub but are commonly a mix of grassland birds and birds that occur across multiple desert habitat types. Birds commonly found in the Chihuahuan Desert Scrub include scaled quail (*Callipepla squamata*), white-necked raven (*Corvus cryptoleucus*), mourning dove (*Zenaida macroura*), roadrunner (*Geococcyx californianus*), lesser nighthawk (*Chordeiles acutipennis*), Scott's oriole (*Icterus parisorum*), cactus wren (*Campylorhynchus brunneicapillus*), curve-billed thrasher (*Toxostoma curvirostre*), and black-throated sparrow (*Amphispiza bilineata*) (Brown 1994).

Herpetofauna found in the Chihuahuan Desert include Texas banded gecko (*Coleonyx brevis*), reticulated gecko (*C. reticulatus*), Greater earless lizard (*Cophosaurus texanus*), and roundtail horned lizard (*Phrynosoma modestum*). Other herpetofauna include the several spiny lizards and whiptail lizards (Brown 1994, Morafka 1977, Wright and Lowe 1968). Common snakes found in the desert scrub include the trans-pecos ratsnake (*Elaphe subocularis*), western hooknose snake (*Gyalopion canum*), Texas black-headed snake (*Tantilla atriceps*), whipsnake (*Masticophis taeniatus* and *M. flagellum lineatus*), Mohave rattlesnake (*Crotalus scutulatus*), and western diamondback (*C. atrox*). The boson tortoise (*Gopherus flavomarginatus*) can also be found in the Chihuahuan Desert Scrub (Brown 1994).

6.3 Chihuahuan Desert Scrub State-and-Transition Model

State-and-transition models (STMs) conceptualize current and predicted ecological conditions (“states”) of vegetation and soil based on our understanding of past interactions among ecological elements, processes, and stressors. Stressors are ecological drivers that are outside their natural range of variability and uncharacteristically push ecological conditions to a degraded state. These ecological stressors and states can serve as key ecological attributes for land management agencies to monitor and gauge subsystem conditions and trends.

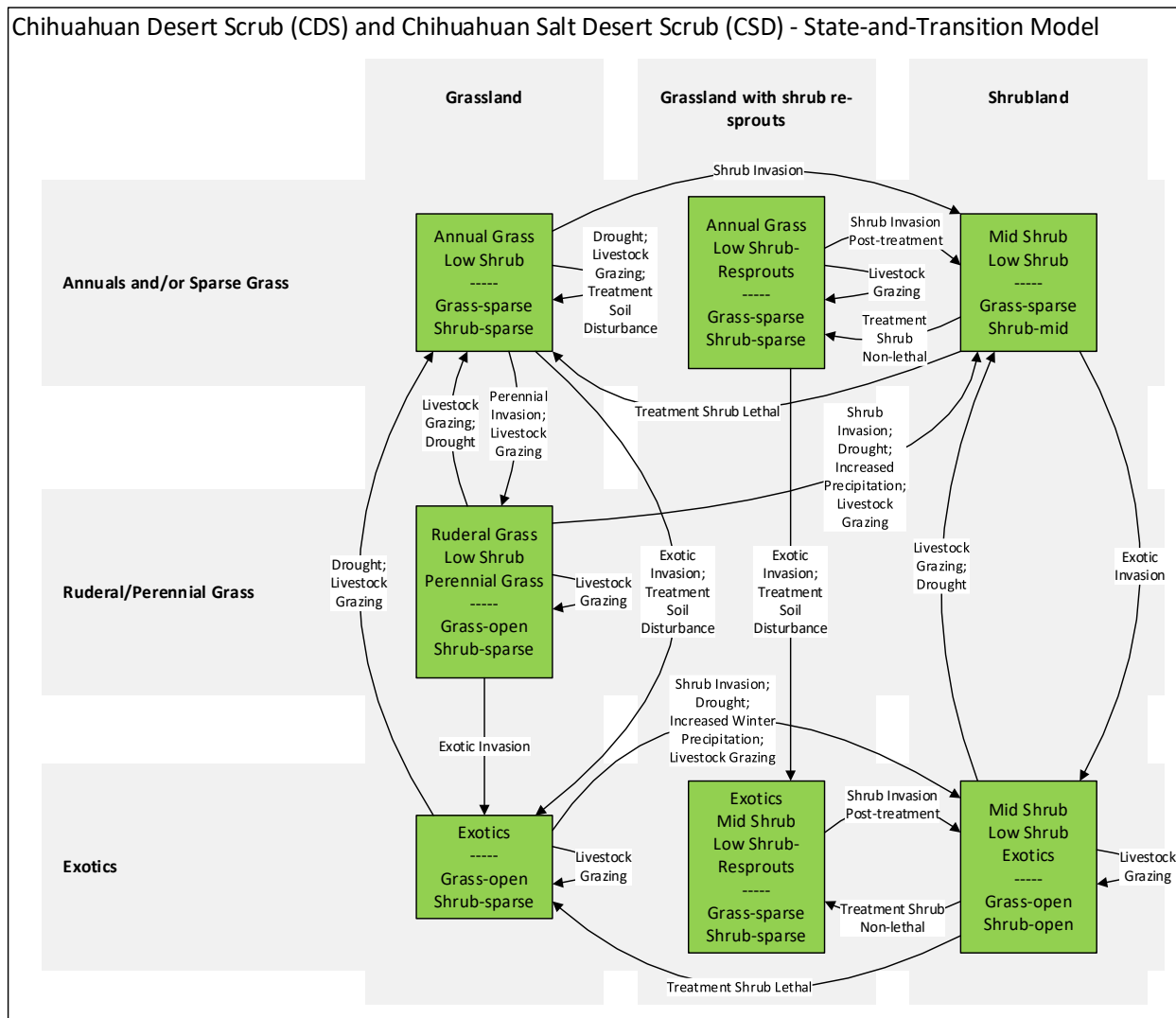
The Chihuahuan Desert Scrub STM for the present REA combines information for the ILAP Chihuahuan Desert Scrub and Salt Desert Scrub STMs (Figure 6-2, after ILAP STM 2012). The STM diagram follows the same conventions as in Chapter 5, above. Rectangles represent ecological states of plant community composition. Arrows represent stressors that are ecological drivers outside their normal range of variation that transition ecological condition from one state to another. Columns represent life form and rows represent plant community composition. Conceptual models are not location specific and refer to large areas of land. Ecological states transition from grass and forb dominance on the left to shrub dominance on the right and from reference states with native vegetation at the top to modified states with dominance by exotic species at the bottom.

The combined Chihuahuan Desert Scrub/Salt Desert Scrub STM (Figure 6-2) contains several grassland and shrubland states. As in the Chihuahuan Desert Grasslands STMs, grassland states in the combined Chihuahuan Desert Scrub/Salt Desert Scrub STM vary in grass species composition, specifically in the relative abundance of native versus nonnative grasses and in the relative abundance of perennial versus ruderal and annual grasses. Grassland states usually have at least some shrubs or shrub resprouts where shrub reduction treatments have been applied but were not severe enough to kill shrub roots (ILAP STM 2012). Shrubland states have herbaceous understory ranging from little to moderate amounts of

biomass in relation to the level of shrub dominance (ILAP STM 2012). Herbaceous states are on the left of each diagram, with shrub states on the right. Reference states with native vegetation are located at the top of each diagram and transition to states dominated by exotic species at the bottom (ILAP STM 2012). Transitions from grassland states to shrubland states occur through the natural succession of shrub invasion (see Shrub Invasion in Figure 6-2), unlike in the Chihuahuan Desert Grasslands ecological system, in which stressors drive this transition (ILAP STM 2012).

Multiple stressors and processes influence transitions among Chihuahuan Desert Scrub/Salt Desert Scrub ecological states. These stressors include the Change Agents identified for assessment in the Chihuahuan Desert REA. The proportion of the U.S. portions of the Chihuahuan Desert ecoregion with woody plant cover is increasing in the (Fuchs 2002). Scrublands have taken over many areas of former grassland (Beck and Gibbens 1999, Lee C. Buffington and Herbel 1965, Schlesinger et al. 1990, York and Dick-Peddie 1969). Large portions of the desert grassland that existed prior to the 1850s are now occupied by desert scrub (Dick-Peddie 1993). Scrubland covers about 70% of the Chihuahuan Desert (Henrickson and Johnson 1986). Creosotebush is the most characteristic shrub of the Chihuahuan Desert and can grow well on multiple soil types (Burke and Dick-Peddie 1973). The spread of scrubland into grasslands is likely the result of many factors including excessive livestock grazing, invasive species, uncharacteristic wildfire, climate change, and development (Figure 6-2).

Figure 6-2. Combined Chihuahuan Desert Scrub and Salt Desert Scrub state-and-transition model.



The following paragraphs discuss how each Change Agent affects Chihuahuan Desert Scrub/Salt Desert Scrub states and transitions, as illustrated in Figure 6-2.

6.3.1 Climate Change

The distribution of native vegetation in the Chihuahuan Desert ecoregion is influenced by several factors including climate (Betancourt et al. 1993). Temperatures in the U.S. portions of the Chihuahuan Desert ecoregion have been warming since the peak of the Little Ice Age in the mid-1880s (Ghil and Vautgard 1991, Miller and Wigand 1994). After about 1880, a decline in precipitation reduced the production of fuels, which in some areas has led to soil erosion and insufficient understory fuel loads to carry fires (Gori and Bate 2007). Interestingly, during the late Holocene (4000-0 years before present), the ranges of some desert scrub indicator species, including creosotebush and ocotillo, have expanded into higher and more northerly locations (Van Devender 1990) and possibly at an accelerated pace during the last century. Creosotebush now appears with pinyon on some southerly, limestone exposures higher than 2000 meters on the Sevilleta National Wildlife Refuge (Betancourt et al. 1993).

As discussed earlier in this chapter and in Chapter 5, the native and invading plant species of the Chihuahuan desert differ widely in their sensitivity to climate variables including the seasonal distribution of precipitation, the frequency and duration of drought, the seasonal distribution of air temperatures, and variability in all of these. Changes in these climate variables therefore will favor some plant species over others and influence plant community composition. An increase in night-time winter temperatures and a shift in precipitation seasonality towards more winter precipitation can favor shrubland expansion because it favors C₃ shrubs over C₄ grasses (Brown et al. 1997, D’Odorico et al. 2010, ILAP STM 2012, Munson et al. 2013, Pennington and Collins 2007). The historic dominance of grasslands by C₄ plants across southern New Mexico is largely a result of the warm growing seasons with low and varying amounts of rain mixed with periodic drought (Peters and Gibbens 2006, Schmutz et al. 1992). Dry weather during winter and spring with subsequent wet late-summer or early-fall weather favor black grama grass growth. These weather patterns were more common in the late 1800s than during the 1900s (Burgess 1995). The shift of some summer precipitation to winter during the 1900s could explain part of the shift from black grama grass to mesquite dominance (Burgess 1995). This vegetational shift has been recorded on both grazed and ungrazed desert grasslands (Burgess 1995, Hannessy et al. 1983).

The climate within the next 100 years in the Southwest is expected to increase in temperature along with the number of extreme events such as droughts (Kunkel et al. 2013a; 2013b, Melillo et al. 2014). Creosotebush, the most common dominant plant species of the Chihuahuan Desert Scrub/Salt Desert Scrub system, is a C₃ evergreen adapted to living with drought and can survive for as long as 400 years (Miller and Huenneke 2000, Peters and Gibbens 2006). In contrast, black grama grass decreased during the 20th century in the northern Chihuahuan desert, giving way to shrubs, largely due to drought during the 1950s (Gibbens and Beck 1988, Peters and Gibbens 2006, Yao et al. 2002). The drought of the 1950s also is suspected to have accelerated shrub expansion into grasslands (Grover and Musick 1990, Herbel et al. 1972, Lohmiller 1963). Regularly occurring droughts of long duration can kill most perennial grasses but if the droughts are of short duration and frequent, they may help grasses stop the establishment of shrub seedlings with extensive, deep root systems (Burgess 1995).

Changes in wildfire and excessive grazing will also affect the way different plant species respond to climate change in the ecoregion. Drier and hotter periods are expected to result in an increased frequency and extent of wildfires (Melillo et al. 2014, Westerling et al. 2006). Droughts also reduce the ability of shrubs to resprout after fire or excessive grazing, with some shrubs requiring 50 years or longer to recover after fire when additionally stressed by drought and grazing (Canfield 1939, McPherson 1995, Nelson 1934, Reynolds and Bohning 1956). The next section of this chapter, on uncharacteristic wildfire, presents additional information on the potential interacting effects of climate change and altered wildfire on the Chihuahuan Desert Scrub/Salt Desert Scrub system.

6.3.2 Uncharacteristic Wildfire

Fire frequencies have declined across the Chihuahuan desert in general since European settlement. This decline in fire frequency plays a significant role in areas that historically had fire, including promoting the expansion of woody species into grasslands. Depending on weather patterns, areas that historically

produced sufficient fuels to carry fire and that have not burned for a long time can accumulate unnaturally high levels of fuel loads. These high fuel loads can result in extreme fire behavior that is more damaging to the plant community compared to historic dynamics. Changes in the historic fire regime result in changes to the structure and composition of plant communities relative to historic conditions. Changes in vegetation composition and structure alter habitat and can reduce populations of obligate wildlife that cannot survive outside of the historic habitat conditions. However, many wildlife species are generalists and do well in multiple vegetation types. Historic fires often created mosaics of plant community types, which often provide habitat for the greatest species diversity. The effects of fire on habitat are discussed in the respective wildlife conceptual model reports. Scrublands with fire-intolerant plant species would not have experienced rapid-fire frequency before European settlement.

As discussed for the grasslands, in Chapter 5, drought historically was likely the dominant form of natural disturbance in the more arid portions of the Chihuahuan desert, with fire and grazing the dominant forms of disturbance in the more mesic parts of the ecoregion. Plant communities in different parts of the ecoregion therefore differ in their adaptations to fire and resilience to this type of disturbance, and these differences in adaptation affect how these communities respond to changes in the frequency and/or intensity of fire. The most arid portions of the Chihuahuan desert, including areas historically dominated by non-fire tolerant plant species, are less resilient to severe wildfire. Non-fire tolerant plants such as grasses and herbaceous species typically have reproductive structures exposed to fire (e.g., near or above the soil surface) or have slow reproductive characteristics. In contrast, fire tolerant plant species in areas with historically sufficient precipitation to regularly produce sufficient fuels for frequent wildfires are more resilient to the historic fire regime. Fire tolerant plant species, including many shrub species common to the Chihuahuan Desert Scrub/Salt Desert Scrub system, typically have reproductive structures protected from fire (e.g., below the soil surface) or have fast reproductive characteristics.

The literature on the effects of wildfire in the Chihuahuan Desert Scrub/Salt Desert Scrub system focuses on the role of fire in limiting scrub encroachment into grasslands, rather than on the ecology of fire in scrub communities *per se* (e.g., Humphrey 1958, McPherson 1995). Historic wildfire regimes prior to the reduction of herbaceous fuels by excessive livestock grazing, may have controlled where Chihuahuan desert shrubs grew (Humphrey 1987). Where fuels are sufficient to carry fire today, prescribed fire can kill shrubs such as creosotebush and burro weed (Cable 1973). In general, high fire frequency damages succulents, subshrubs, and woody plants (Cable 1967, Heirman and Wright 1973). However, several subshrub and woody plant species are fire tolerant (e.g., rabbitbrush, mesquite; Sarmiento and Monasterio 1983, Zedler et al. 1983). Plant communities with a high percent of fire-intolerant plant species (e.g., succulents, black grama grass) are less adapted fire (Burgess 1995) and consequently are not highly resilient to fire as a group. In contrast, plant communities with more fire-tolerant plant species likely had more frequent fire during their evolutionary past and consequently are more resilient to fire. Vegetation resiliency to disturbance such as wildfire also generally increases along a gradient of increasing resource availability and productivity (Chambers et al. 2007, Wisdom and Chambers 2009). The greater resiliency is related to a greater regenerative capacity following disturbance in these areas

and a greater capacity to among the native plants to compete with invasive plants for resources (Brooks and Chambers 2011).

Insufficient amounts of fine fuel to carry fire limit fire frequency in scrub communities. However, when fire does occur, the presence of shrubs and trees that resprout or produce fire-stimulated seeds can reduce the time needed for woody species redominate a site, compared to sites with shrubs that lack such fire adaptations (Barney and Frischknecht 1974, Ward 1977, Wright 1972, Wright et al. 1979). Examples of resprouting shrubs and trees include mesquite, shrub live oak (*Quercus turbinella*), rabbitbrush (*Chrysothamnus viscidiflorus*), snakeweed (*Gutierrezia sarothrae*; Gori and Bate 2007), and alligator juniper (*Juniperus deppeana*; Johnson 2002. Shrubs that do not resprout following severe fire can eventually return over time, if fire or other severe disturbance is not present. Shrubs often become dominant a few decades after stand-replacing fire with shrub dominance potentially lasting more than 100 years (Barney and Frischknecht 1974, Erdman 1970, Gori and Bate 2007, Koniak 1985, Schott 1984, Young and Evans 1978).

Climate change can alter desert wildfire regimes, and this interaction may be exacerbated by the presence of some invasive plant species. A major limiting factor of fire regimes in desert environments is fuel load. Fuel load in turn depends significantly on the amount of precipitation water available to plants and the temperatures in which plants have to grow. Major fire seasons in deserts generally occur following periods of higher than average precipitation, because the precipitation typically promotes increased biomass production, which in turn subsequently results in greater fuel loads (Crimmins and Comrie 2004). In turn, major fire seasons do not require excessively dry weather. The normal Chihuahuan desert dry season, April-June, is usually sufficient to dry fuels enough to allow burning (Crimmins and Comrie 2004). Consequently, a change in precipitation patterns during the normally wetter seasons across the ecoregion would be expected to result in a change in wildfire patterns. For example, El Niño cycles tend to result in greater winter precipitation in the ecoregion (Crimmins and Comrie 2004). Increased winter precipitation favors plants such as the invasive Lehmann lovegrass that use winter precipitation highly efficiently for growth (Cable 1971, Crimmins and Comrie 2004). Increased precipitation during El Niño cycles over much of the last 50 years have allowed plants like Lehmann lovegrass to flourish, resulting in an increase in the fine fuels needed to carry fire in desert environments (Crimmins and Comrie 2004). Other invasive annual plants also can take advantage of winter precipitation more efficiently than can native plants, resulting in further increases in surface fuels that can help carry fire. Phase II of the Chihuahuan Desert REA will further examine the potential impacts of climate change on the vegetation of the ecoregion.

Land development also can alter desert wildfire regimes, and this interaction again may be exacerbated by the presence of some invasive plant species. Humans have converted scrublands to many other uses including croplands, urban and suburban developments, and industrial developments including mines and other sites of geologic resource extraction. These changes remove historic vegetation and fragment remaining scrublands, and have countless other effects on soils, hydrology, and wildlife. Development typically results in changes to courses of natural waterways, and in a decline in water table elevations. The building of roads and structures increases human presence and the presence of ignitions sources that can result in unusually high fire frequencies around developed areas. Fire suppression, which

started during the 1900s and increased significantly after World War II, has reduced fire frequency (Miller et al. 2000) in plant communities previously naturally adapted to wildfire, including scrublands, resulting in increased fuel loads in many areas. The reduced frequency of low- to mid-severity fire has led to rare but more severe fires in areas where fuels have accumulated over the years of fire suppression. These extreme fires can lead to changes in plant community composition, wildlife habitat, soil erosion, and loss of life and property.

6.3.3 Invasive Species

Since European settlement, invasive plants have threatened historic plant communities. Prior to the introduction of exotic, invasive plants, disturbance was less of a problem because the only plant propagules available to re-dominate disturbed areas were native plants. Currently, invasive plants appear almost everywhere and often take up resources more quickly than the desired historic plants. The presence and ability of exotic, invasive plants to dominate an area is a major limiter of restoration potential and puts some treatment areas at risk of failure.

Invasion of the Chihuahuan desert in the U.S. by exotic species varies by location (see Exotic Invasion Figure 6-2; ILAP STM 2012). Among the three Chihuahuan desert dry system CEs, scrublands are the least susceptible to exotic plant invasion. Grasslands on piedmont or foothill landforms are more easily invaded (ILAP STM 2012). Areas near existing exotic plants are more likely to be invaded than areas farther away, simply because propagules must travel farther (ILAP STM 2012). Environmental conditions also affect the ability of some exotic species to invade some areas. For example, the exotic plant species *Eragrostis lehmanniana*, *Eragrostis cilianensis*, *Erodium cicutarium*, *Fumaria parviflora*, *Salsola tragus*, and *Sisymbrium irio* occur in the general region of the southwestern US and northern Mexico region (Van Devender et al. 2013). However, these plants were not found to be invasive on soils weathered from limestone near Douglas, AZ, in vegetation similar to the Chihuahuan Desert Scrub (Van Devender et al. 2013). Other exotic species are more cosmopolitan. The exotic, cosmopolitan African buffelgrass (*Pennisetum ciliare*) occurs in both the southern New Mexico and west Texas portions of the Chihuahuan Desert (USDA, NRCS 2014), including in scrubland settings. It occurs mainly below 1060 m in elevation and is a serious problem in the Big Bend area of Texas within the Chihuahuan Desert (Van Devender et al. 2013). It strongly competes with native plants for water and nutrients and creates fuel loads for fire in ecosystems naturally not prone to wildfire (Van Devender et al. 2013). A state-wide list of noxious weeds is available at <http://www.nmda.nmsu.edu/apr/noxious-weed-information/> for New Mexico and http://texasinvasives.org/plant_database/ for Texas.

Increased invasive plant dominance in general has the potential to increase fuel loads and fire frequency beyond historic levels. Most desert shrubs can be killed by fire during their seedling phase (Bock and Bock 1992a, Cable 1967, Cox et al. 1993, Glendening and Paulsen 1955, Humphrey 1949, McPherson 1995, Reynolds and Bohning 1956, Wright et al. 1976). Many of these shrubs do not produce seeds until 10+ years of age (Chew and Chew 1965, Humphrey 1958, Martin 1975, McPherson 1995). Some mature desert shrubs also are at risk of fire induced mortality when sufficient fine fuels exist because they are not able to resprout from roots or produce sufficient long-lived, fire tolerant propagules. Prescribed fire can kill creosotebush and burro weed, for example, where fuels are sufficient to carry fire (Cable 1973).

However, additional research is needed to more fully assess the resprouting behavior of desert shrubs following fire (Abella 2009) because several studies, as cited in these REA reports, discuss the importance and ability of fire as a tool to limit woody species invasion into neighboring plant communities. This is contrary to some early reports that state that many woody species in the Chihuahuan Desert resprout after fire (Kittams 1973, Worthington and Croal 1987). Shrubs that resprout return to pre-disturbance levels more quickly than species that have to regenerate from seed. Many desert perennial plants rarely establish new individuals from seed (Callison et al. 1985, Guo 2004). Differences in fire timing, behavior, and severity among burns (Abella, 2009) may account for many of the differences found among plant responses to fire damage. Cool fires are expected to cause less damage to the ability of a plant to resprout after fire (Abella 2009).

Invasive plants may have traits that make them more adapted to an area than the previous vegetation. Buffelgrass (*Pennisetum ciliare*) distribution previously had been limited by cold winter temperatures. However, a new variety of buffelgrass may increase the extent of land (e.g., cooler or higher elevation) that the species can inhabit and potential invade (Archer and Predick 2008). The ability of the exotic Lehmann lovegrass (*Eragrostis lehmanniana*) to compete with native grasses is likely due to its ability to make more use of winter precipitation and better tolerate drought than some native species (Archer and Predick 2008). Other traits of Lehmann lovegrass that contribute to its ability to maintain its current extent and spread into new territories include prolific production of seeds that are easily spread by water or wind, an ability to produce multiple seed crops in a year, and production of seeds that have different dormancy periods. This latter trait spreads germination out over time, increasing the chances of at least some seeds establishing when suitable conditions arise (Humphrey 1994).

6.3.4 Development

Chapters 2 and 3 summarize information on the types and extent of development in the U.S. portions of the Chihuahuan desert. The spatial distributions of the different types of development in the ecoregion reflect the interaction of several geographic and economic factors, as discussed in Chapter 5. Scrubland habitat has been replaced in several areas of the ecoregion by expansions of commercial development along the U.S.-Mexico border, most notably in the El Paso/Juarez area; expansions of military and security activities and infrastructure along the border and around military facilities; expansions of oil and gas production and distribution systems in and around the Permian Basin across large portions of the east side of the ecoregion; and expansions of residential and commercial development around cities and towns. Transportation infrastructure developments associated with these activities also fragment the surrounding scrublands. All these developments increase air pollution, risks of accidental fires, and pressures on wildlife; and associated demands on water supplies can lower water tables, potentially eliminating some natural watering holes for wildlife that may also have supported patches of distinctive vegetation.

6.3.5 Excessive Domestic Grazing

Excessive livestock grazing alters ecological states in Chihuahuan desert grasslands by reducing the biomass of highly palatable plant species – often consisting of grasses and herbaceous species – or in extreme instances eliminating these species entirely (see Livestock Grazing in Figure 6-2). The remaining

plants have less photosynthetic tissue to support growth and less energy to extend roots to collect soil resources. The unused resources that formerly were used by these palatable species then can be used by the less palatable plant species including woody species, which can then increase in size and the area they cover. In this way, excessive livestock grazing can promote the conversion of grassland to scrubland. Grazing pressure may lessen after the more palatable plants are greatly reduced or eliminated. However, excessive livestock grazing can change community composition sufficiently to reduce the production of propagules (e.g., seeds, stolons, rhizomatous roots) by the palatable native species of the community, and thereby reduce the ability of the historic plant community to recover – e.g., revert from scrubland back to grassland – after the grazing pressure lessens. Excessive grazing similarly can alter the composition of the grass and herbaceous assemblages in native scrubland. For example, excessive livestock grazing can drive transitions from perennial grassland to ruderal grassland states in both Chihuahuan Desert Scrub and Grassland systems (Belsky 1992, Berg et al. 1997, Drewa and Havstad 2001, ILAP STM 2012). Excessive grazing combined with increasing aridity also has helped drive the increase of scrub on former grassland in the Chihuahuan desert (Krammes 1990, Nelison 1986).

Livestock distribute plant propagules including the transport of seeds of woody species to new or disturbed areas. For example, this factor is suspected in the transition of black grama grasslands to unhealthy mesquite dunelands (Campbell 1929, Dick-Peddie 1993). This transition often results in further degradation because mesquite plants do not provide sufficient protection of the soil and thereby allow the surface soil to erode. Where the top soil horizon erodes away, the lower soil horizons often cannot support the historic biodiversity and plant community composition because of less suitable soil characteristics including less organic matter, nutrient availability, and sometimes less water holding capacity.

6.3.6 Landscape Restoration

Reducing the conversion of grasslands into scrublands is an important management concern in the U.S. portions of the Chihuahuan desert, as discussed in Chapter 5. Land managers therefore have tried a wide range of methods for driving down or eliminating shrubs from former grassland sites. Treatments commonly applied to return shrub invaded states back to grassland states include fire, chaining, and herbicide. Treatments applied at intense levels that are severe enough to kill shrub roots can cause a shrub dominated site to transition back to a grassland state (see Treatment Shrub Lethal in Figure 6-2; ILAP STM 2012). Less severe treatments do not kill shrub roots and only set the plant community back to a grassland state with live shrub roots remaining that can resprout (see Treatment Shrub Non-lethal in Figure 6-2). This leads to a faster return of shrub cover and need for more frequent treatments (see Shrub Invasion Post-treatment in Figure 6-2; ILAP STM 2012).

Creosotebush and mesquite are two widespread shrub species. As also discussed in Chapter 5, fire can limit mesquite encroachment into grasslands because fire can kill mesquite seeds and seedlings (Brown and Archer 1999, Cox et al. 1993, Peters and Gibbens 2006). However, fire does not usually kill mature mesquite, which can resprout prolifically following fire or cutting (Peters and Gibbens 2006). A mean fire

return interval of 7-10 years is needed to maintain an open grassland state without many shrubs (Griffiths 1910, Leopold 1924, McPherson 1995, Schmutz et al. 1985, Wright and Bailey 1982).

In fact, most shrubs that have become established in Chihuahuan desert grasslands can be killed by fire during their seedling phase (Bock and Bock 1992a, Cable 1967, Cox et al. 1993, Glendening and Paulsen 1955, Humphrey 1949, McPherson 1995, Reynolds and Bohning 1956, Wright et al. 1976) without resprouting (McPherson 1995). The effectiveness of this method results from the fact that many of these shrubs do not produce seed until 10+ years of age (Chew and Chew 1965, Humphrey 1958, Martin 1975, McPherson 1995). However, shrub invasion reduces the availability of fine surface fuels for wildfire, making it difficult to reintroduce fire as a shrub management tool, particularly after invading shrubs mature (McPherson 1995). Under these circumstances, land managers need to use chemical, mechanical, or other tools to return scrublands back to grassland states (McPherson 1995). However, landscape restoration treatments that result in soil disturbance also can fail. Disturbed soils promote the establishment of undesirable plant species, causing the treated site to transition to a highly undesirable state (see Treatment Soil Disturbance in Figure 6-2; ILAP STM 2012).

As noted above, the exotic Lehman lovegrass can spread into natural scrub communities, where it can increase fire fuel loads leading to fires of sufficient intensity to kill native scrub species. Unfortunately, Lehmann lovegrass is more fire tolerant than several native grasses in the Chihuahuan Desert (McGlone and Huenneke, 2004). As a result, managers worry that burning in areas invaded by Lehman lovegrass may favor rather than reduce its persistence. Some research does suggest that fire does not promote the spread of Lehman lovegrass (e.g., (McDonald and McPherson 2011, McGlone 2013), but the use of fire to control Lehmann lovegrass in native scrub communities nevertheless must also take into account the potential for harm to fire-sensitive shrubs as well. Chapter 5 (see USFS 2014) provides additional information on treatments to restore land invaded by Lehman lovegrass.

Several types of land reclamation approaches have been studied over the decades at or near the Jornada Experimental Range in the Chihuahuan Desert (Rango et al. 2002), as also discussed in Chapter 5. Many of these vegetation treatments were implemented around the 1930s and often included help from the Civilian Conservation Corps prior to World War II (Rango et al. 2002). Many of the treatment details related to these studies were not sufficiently recorded or are missing. However, retrospective analyses of aerial or satellite imagery reveal the long-term effects or lack thereof for several types of treatments. In general, contour terraces, brush water spreaders, strips grubbed free of shrubs, and mechanical root plowing and seeding did not have long-term desired treatment outcomes (Rango et al. 2002).

The analyses of the data on land reclamation at and near the Jornada Experimental Range suggest several reasons why so many treatment methods did not have the desired long-term benefits. Experimental treatments often were not applied at sufficiently large extents resulting in landscape scale hydrological and ecological processes overwhelming small treatment areas (Rango et al., 2002). Treatments were not maintained over time and this resulted in the loss of initial treatment effectiveness. Treatments sometimes were located in the wrong area to find successful outcomes from experimental treatments (Rango et al. 2002).

These findings near the Jornada Experimental Range have implications for land managers. Historical aerial and satellite imagery should be evaluated prior to applying vegetation treatments to ensure that treatments are of sufficient size to not be dominated by hydrological or ecological processes from nearby lands and that treatments are appropriately located on the desired overall landscape position away from decades earlier or current nearby factors that would lessen treatment effectiveness. These recommendations are based on the findings that some factors are more readily apparent from an aerial perspective than from the ground. These findings also suggest that long-term plans for monitoring, accurate record keeping, and repeat treatments are essential to long-term treatment success. Even though several types of treatments were unsuccessful, some experimental treatments did have long-term vegetation benefits. The treatments of water retention dikes, some fence lines, and some boundaries between disparate management practices displayed long-term treatment effectiveness (Rango et al. 2002).

Climate change can limit restoration success when the current, modified temperature and precipitation patterns no longer favor the historic plant community. Climate change is more fully addressed in the climate related sections of the REA reports. In addition, soil erosion following severe disturbances such as land development, severe wildfire, or invasion by some weedy species can alter scrubland distribution and landscape restoration potential. Where the top soil horizon erodes away, the lower soil horizons often cannot support the historic biodiversity and plant community composition because of less suitable soil characteristics including less organic matter, nutrient availability, and sometimes less water holding capacity.

6.4 Chihuahuan Desert Scrub Key Ecological Attributes

The variables that distinguish the ecological states from each other in the Chihuahuan Desert Scrub/Salt Desert Scrub STM all constitute key ecological attributes for the system. Additional key ecological attributes include critical aspects of soil condition, wildfire, and soil moisture. The list below identifies seven (7) key ecological attributes for Chihuahuan Desert Scrub CE on these criteria. *Characterizing the present condition of a system requires data on indicators for its key ecological attributes.* Indicator sources for these key ecological indicators include: Ecological Site Descriptions developed by the U.S. Department of Agriculture, Natural Resources Conservation Service (USDA NRCS 2006) for ESD R042XB014NM Loamy and R042XB035NM Gravelly Loam, LANDFIRE (2014) Biophysical Setting 2515030 Chihuahuan Loamy Plains Desert Grassland, and expert knowledge.

- **System State**
 - Perennial-Ruderal-Invasive Grass Species Composition and Age Structure
 - Herbaceous versus Woody Species Cover
 - Grass versus Shrub Composition and Age Structure
 - Vegetation Patch Size and Fragmentation
 - Soil Exposure and Stability
- **Critical Dynamics**
 - Wildfire Dynamics
 - Soil Moisture Dynamics
 - Soil Erosion

7 Pinyon-Juniper Woodlands Conceptual Model

This chapter presents the conceptual ecological model for the Pinyon-Juniper Woodlands CE. The presentation follows the structure described in Chapter 4. As defined here, “pinyon-juniper woodlands” includes pinyon-juniper savanna, pinyon-juniper shrub woodlands, and pinyon-juniper persistent woodlands, as distinguished by Gori and Bate (2007).

7.1 Sources of Conceptual Models and Vegetation Classification

The Pinyon-Juniper Woodlands conceptual model is adapted from Gori and Bate (2007) using the vegetation dynamics development tool (VDDT, ESSA Technologies Ltd.). The Madrean Archipelago REA also used this source for its pinyon-juniper woodland conceptual models (Crist et al. 2014). These woodlands occur in the Chihuahuan Desert and parts of Arizona, New Mexico, and Texas. The present adaptation included converting the VDDT model diagram into a state-and-transition conceptual model (STM) diagram similar in style to the hierarchical approach of Miller et al. (2010) using Microsoft Visio software. The STM model is not location specific but rather refers to large areas of land with the same general potential natural vegetation type (PNVT). PNVT is largely determined by biogeochemical and disturbance regime constraints.

The Pinyon-Juniper Woodlands ecological system is ecologically important and has been classified by other organizations. As noted in Chapter 3, the term, “ecological system” here refers to “... recurring groups of biological communities that are found in similar physical environments and are influenced by similar dynamic ecological processes, such as fire or flooding” (Comer et al. 2003). NatureServe refers to this CE as Madrean Pinyon-Juniper Woodland (International Ecological Classification Code CES305.797) and Madrean Juniper Savanna (CES305.730) (NatureServe 2014). The LANDFIRE classification approach, related to the NatureServe approach, uses the same classification names but assigns them as biophysical settings 2510250 and 2511160, respectively (LANDFIRE 2014). Related ecological site descriptions developed by the U.S. Department of Agriculture, Natural Resources Conservation Service (USDA NRCS 2006) include Igneous Hill and Mountain (R042XE277TX, R042XF286TX), Shallow (R070DY152NM), Gravelly (R070DY156NM), and Breaks 12-18” PZ (R070BY661TX). The present Pinyon-Juniper Woodlands conceptual model incorporates information from these publications as well.

7.2 Pinyon-Juniper Woodlands Control Model

The Pinyon-Juniper Woodlands control model (Figure 7-1) illustrates current understanding of how the main ecological components and controlling processes for this ecological system. The ecological components of the control model include landscape configuration, elevation and soil-geomorphic setting, precipitation and temperature, vegetation, natural disturbance regimes, and wildlife and habitat. The following paragraphs explain these individual model components.

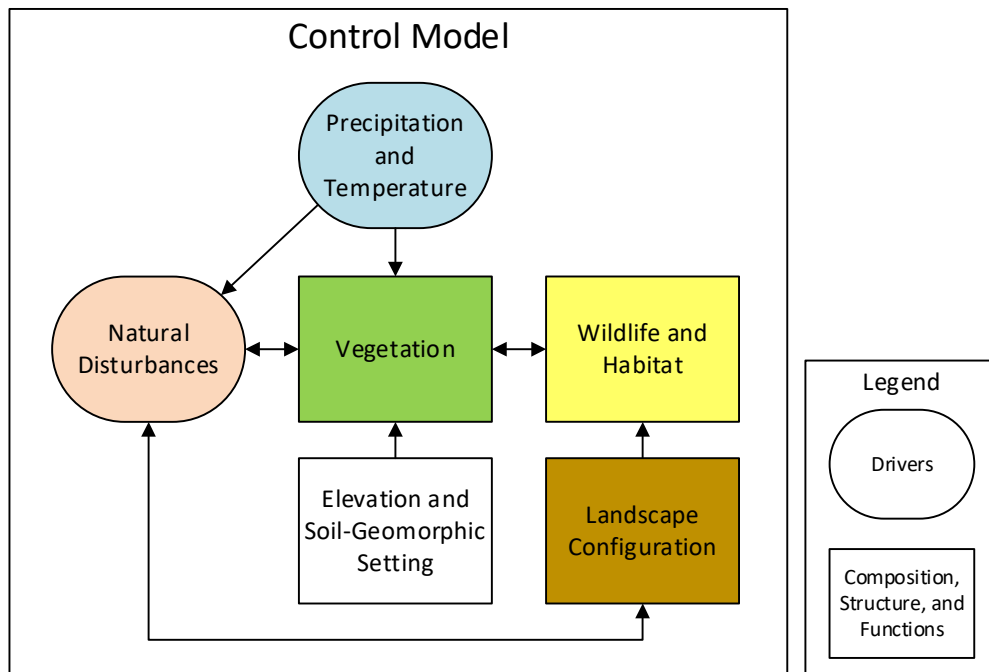
7.2.1 Landscape Configuration, and Elevation and Soil-Geomorphic Setting

Pinyon and juniper trees occur on a variety of landforms including basins, hills, and slopes from low to moderate elevations (LANDFIRE 2014). Chihuahuan Desert Pinyon-Juniper savannas and woodlands

occur between 1400 and 2200 m in elevation with savanna occurring below (LANDFIRE 2014). Pinyon-Juniper communities are often bordered by grasslands at the lower elevations (LANDFIRE 2014). Pinyon typically occurs at elevations higher than juniper whereas juniper is often more common at lower elevations because of differences in germination and mortality (Martens et al. 2001, Merkle 1952, Naylor 1964, Springfield 1976, Woodin and Lindsey 1954). However, in southern New Mexico, juniper is more common than pinyon at higher elevations, as well, likely because of greater summer precipitation in the south (Ernest et al. 1993, Gori and Bate 2007, Hill 1990, Kennedy et al. 1983, Pieper et al. 1971, Potter 1957).

Pinyon-juniper communities occur on a variety of soils. These soils include gravely loams and clayey textures with deep to shallow depths and are well to poorly drained (Springfield 1976, Woodbury 1947). Although Pinyon-Juniper communities can be found on a variety of soils, they usually occur on shallow and rocky, alkaline soils that are well drained and have little fertility (Budy and Meeuwig 1987, Evans 1988, Howell 1941, Meeuwig and Bassett 1983, Pieper 1977). Where Pinyon-Juniper occur on deep, fine or mesic soils, biomass production is usually greater along with more understory plant density and cover (Gori and Bate 2007, West 1999).

Figure 7-1. Pinyon-Juniper Woodlands control model.



7.2.2 Precipitation and Temperature

Southwestern pinyon-juniper communities occur in Arizona, New Mexico, and Texas. The southwestern U.S. generally has a bimodal precipitation pattern that in southeastern New Mexico is skewed toward summer precipitation (Gori and Bate 2007, Jurwitz and Kangieser 1978, Springfield 1976). The annual amount of precipitation varies across the distribution of pinyon-juniper woodlands. Mean annual precipitation is less in northern Arizona with 18 cm and greater further south with 63 cm (Gori and Bate 2007, Ronco 1990, West et al. 1975). Pinyon-juniper canopy cover tends to vary in direct relation to the amount of precipitation, with more cover occurring where there is greater mean annual precipitation (O'Rourke and Ogden 1969, Springfield 1976). Mean annual temperatures also follow the precipitation pattern, varying from 40 °F in the north to 61 °F in the south (Gori and Bate 2007, Ronco 1990).

7.2.3 Natural Disturbances

The most influential natural disturbances that modify pinyon-juniper community structure include climate variation, fire, insects, and seed dispersal (Gori and Bate 2007). Over thousands of years, pinyon and juniper have spread into drier, lower elevation sites often along with increased grass growth during wetter climates. Pinyon and juniper also have retreated to wetter, higher elevation sites during drier climates (Gori and Bate 2007, West 1999). Recent examples of tree death related to drought have occurred during the 20th and 21st centuries. Droughts are often accompanied by insect outbreaks. Pinyon Ips are a common insect that increase when trees are water stressed and they can increase the number of trees that die during drought. Climate not only influences mortality but also plays a large role in seed production and subsequently wildlife populations that consume tree seeds. Pinyon cone production is greater following unusually cool weather during late August and early September (Gori and Bate 2007). Trees that occasionally produce extra-large masts help ensure that seeds are dispersed by satiating birds who cache seeds that later may establish as new trees (Gori and Bate 2007).

Wildfire and climate variation together largely controlled pinyon-juniper succession and distribution prior to European settlement (Alderete 1996). Pinyon-juniper seedlings are susceptible to fire but mature trees are less susceptible, especially mature juniper trees. The major historic increases in pinyon-juniper distribution and infill across the southwestern U.S. coincide with decreased wildfire frequency in the late 1800s and early 1900s (Miller et al. 2000). Reduced fire frequency has been connected with livestock overgrazing and later anthropogenic fire suppression, which together have allowed woodlands to spread and increase in density and cover to the detriment of herbaceous cover (Gori and Bate 2007, Miller 1999). When rare stand-replacing fires occur, they often return areas invaded by pinyon-juniper trees temporarily back to annual plant dominance (restarting secondary succession; Alderete 1996, Arnold et al. 1964, Barney and Frischknecht 1974, Erdman 1970). Perennial herbaceous plants then typically reclaim dominance a few years after the stand-replacing fire, followed by a combination of perennial herbaceous and shrub plants (Alderete 1996, Arnold et al. 1964, Barney and Frischknecht 1974, Erdman 1970). The presence of shrubs and trees that resprout or produce fire stimulated seeds after fire can reduce the time until woody species again dominate a site; non-sprouting woody plants that repopulate through seeds and seedling establishment lack this capability to quickly recolonize a burned site (Barney and Frischknecht 1974, Ward 1977, Wright 1972, Wright et al. 1979). Resprouting shrubs and trees in the U.S. portions of the Chihuahuan desert include shrub live oak

(*Quercus turbinella*), rabbitbrush (*Chrysothamnus viscidiflorus*), snakeweed (*Gutierrezia sarothrae*; Gori and Bate, 2007), alligator juniper (*Juniperus deppeana*), and to a lesser extent one-seed juniper (*Juniperus monosperma*; Johnson 2002). Woody species that do not resprout following severe fire can eventually return over time when fire or other severe disturbance is not present.

The state-and-transition model section later in this chapter presents further information on the types of natural disturbances that influence pinyon-juniper community composition, distribution, and condition, including wildfire.

7.2.4 Vegetation

Pinyon-juniper woodlands and savannas occupy 7.5 million hectares (18.5 million acres) across Arizona and New Mexico and vary from low to high tree densities (Gori and Bate 2007). Common tree species include Mexican pinyon (*Pinus cembroides*), border pinyon (*Pinus discolor*), two-needle pinyon (*Pinus edulis*), alligator juniper (*Juniperus deppeana*), one-seed juniper (*Juniperus monosperma*; LANDFIRE 2014), redberry juniper (*Juniperus coahuilensis*), and Pinchot's juniper (*Juniperus pinchotii*; USDA NRCS 2016). Understory plant community composition can vary greatly but the common shrubs include manzanita (*Arctostaphylos pungens*), Apache plume (*Fallugia paradoxa*), barberry (*Berberis* spp.; LANDFIRE 2014), mountain mahogany (*Cercocarpus montanus*), cliff fendlerbush (*Fendlera rupicola*), Wright's silktassel (*Garrya wrightii*), scrub oak (*Quercus turbinella*), and other *Quercus* spp. Grass cover ranges from dense in the savanna to sparse where woody plant cover is high (LANDFIRE 2014). Typical grasses include sideoats grama (*Bouteloua curtipendula*), cane bluestem (*Bothriochloa barbinodis*), muhly grasses (*Muhlenbergia emersleyi*, *M. torreyi*; LANDFIRE 2014), blue grama (*Bouteloua gracilis*), black grama (*Bouteloua eriopoda*), hairy grama (*Bouteloua hirsuta*), bulb panicgrass (*Panicum bulbosum*), pinyon ricegrass (*Piptochaetium fimbriatum*), squirreltail (*Elymus elymoides*), and beardgrass (*Bothriochloa laguroides*; USDA NRCS 2016).

Pinyon-juniper communities have been categorized into three community types—grass savanna, shrub woodland, and persistent woodland—according to structure of the canopy, composition of the understory, and historic fire regime (Romme et al 2003). Pinyon-juniper grass savannas have low tree and shrub density with high herbaceous density (Gori and Bate 2007). Pinyon-juniper grass savannas most commonly occur on deep soils with fine textures in valley bottoms or low hills where fire is able to spread long distances but this vegetation type also can occupy rockier areas (Gori and Bate 2007). This type commonly occurs where the annual precipitation pattern is bimodal or skewed toward summer precipitation (Gori and Bate 2007). Pinyon-juniper shrub woodlands are mostly dominated by a combination of trees and shrubs but the community can transition through a range of ecological states that include periods of herbaceous, shrub, or tree dominance across time (Gori and Bate 2007). Severe fire can return the woody species dominated communities back to herbaceous dominance (Gori and Bate 2007). The shrub woodland type occurs on a range of soil types from deep soils with fine textures on valley bottoms and low hills to coarse soils where depth can be shallow to deep and terrain more rugged (Gori and Bate 2007). The annual precipitation patterns for this vegetation type are commonly skewed to more precipitation during winter than summer but not always (Gori and Bate 2007). Pinyon-juniper persistent woodlands commonly have trees of various ages with some trees >300 years of age.

These woodlands typically have high tree density and canopy cover with little herbaceous or shrub cover (Gori and Bate 2007). Pinyon-juniper persistent woodlands occur on a variety of soil types and in a variety of annual precipitation patterns but usually occurs on rocky and thin soils where fuel breaks limit fuel continuity and fire spread (Gori and Bate 2007).

7.2.5 Wildlife and Habitat

Several animals, mostly small mammals and birds, eat pinyon nuts or juniper berries (Arnold et al. 1964, Balda 1987, Gottfried et al. 1995, Johnsen 1962, McCulloch 1969, Noble 1990, Salomonson 1978, Scott and Boeker 1977, Short et al. 1977). Large mammals including mule deer (*Odocoileus hemionus*), white-tailed deer (*Odocoileus virginianus*), and elk (*Cervus canadensis*) also eat pinyon and juniper seeds and leaves along with grasses, forbs, and some shrubs that also occur in pinyon-juniper communities (e.g., mountain mahogany, Gambel's oak, wavyleaf oak, big sagebrush, and cliffrose) (Martin et al. 1961, Short et al 1977).

Seed consumers also benefit Pinyon-Juniper communities with birds being the main dispersal vector of pinyon and juniper seeds. Juniper seeds that survive consumption by birds or mammals germinate faster than seeds that were not consumed (Johnsen 1962). Birds that commonly consume and disperse pinyon seeds include Scrub jays (*Aphelocoma californica*), pinyon jays (*Gymnorhinus cyanocephalus*), Steller's jays (*Cyanocitta stelleri*) and Clark's nutcrackers (*Nucifraga columbiana*; Balda and Bateman 1971, Ligon 1978, Vander Wall and Balda 1977). Competition for pinyon cones and seeds can occur between small mammals including cliff chipmunks (*Neotamias dorsalis*), rock squirrels (*Spermophilus variegatus*), birds, and insects when there are not enough cones and seeds for all consumers (Christensen and Whitham 1993, Gori and Bate 2007).

7.3 Pinyon-Juniper Woodlands State-and-Transition Model

State-and-transition models (STMs) conceptualize historic, current, or predicted ecological conditions ("states") of vegetation and soil based on our understanding of past interactions among ecological elements, processes, and stressors. Stressors are ecological drivers that are outside their natural range of variability and uncharacteristically push ecological conditions to a degraded state. These ecological stressors and states can serve as key ecological attributes for land management agencies to monitor and gauge subsystem conditions and trends.

The Pinyon-Juniper Woodlands STM for the present REA (Figure 7-2 through Figure 7-6) presents information separately for the three system variants distinguished by Gori and Bate (2007), Pinyon-Juniper Savanna, Pinyon-Juniper Shrub Woodlands, and Pinyon-Juniper Persistent Woodlands. Additionally, the STMs for the Pinyon-Juniper Savanna and Pinyon-Juniper Shrub Woodlands system variants characterize separately their historic versus current (and predicted near-future) manifestations, following Gori and Bate (2007).

The STMs for historic versus current system conditions for the Pinyon-Juniper Savanna and Pinyon-Juniper Shrub Woodlands system variants differ primarily in the frequency of fire disturbance. Historically, prior to ca. 1880-1900, wildfire of low to mixed severity maintained pinyon-juniper grass

savannas as savannas and wildfire of mixed severity maintained pinyon-juniper shrub woodlands as shrub woodlands prior (Gori and Bate 2007). After ca. 1880 (current), wildfires of low to mixed severity became rare because livestock grazing reduced fuel loads, fire suppression limited fire spread, and drought reduced production of fuels, which in some areas led to soil erosion and insufficient understory fuel loads to carry fires (Gori and Bate 2007). Average temperatures also have warmed since the peak of the Little Ice Age in the mid-1880s (Ghil and Vautgard 1991, Miller and Wigand 1994). Along with increased drought, insect damage to trees has increased, resulting in increased tree mortality. The STM for Pinyon-Juniper Persistent Woodlands does not distinguish historic from current dynamics, because this variant primarily occurs in areas of highly uneven topography, which provides natural barriers to fire spread and soils that limit understory production that would otherwise compete with the pinyon and juniper for soil resources (Gori and Bate 2007).

Some of the pinyon-juniper grass savannas and woodlands of today in fact were grasslands prior to 1900 (Cottam and Stewart 1940, Dwyer and Pieper 1967, Gori and Bate 2007, Johnsen 1962, Leopold 1924, Miller 1999, Shaw 2006). Pinyon and juniper trees have spread into new areas and increased in density during the last 100-140 years (Cottam and Stewart 1940, Davis and Turner 1986, Gottfried and Ffolliott 1995, Huffman et al. 2006, Landis and Bailey 2005, Leopold 1924, Springfield 1976, West et al. 1975). Pinyon and juniper have reduced shrubs and grasses in some areas leaving dead shrubs and bare soil between the trees with an increased opportunity for erosion (Allen 2001, Brockway et al. 2002, Gori and Bate 2007, Jacobs et al. 2002, Leopold, 1924, West et al. 1975). As noted above, the start of pinyon-juniper spread and infill coincides with the decrease in fire frequency around 1880 to 1900 (Gori and Bate 2007). Reduced fire frequency has been associated with livestock grazing and later fire suppression that have allowed trees to increase in density and cover, with an accompanying decrease in herbaceous cover (Gori and Bate 2007, Miller 1999).

The Pinyon-Juniper Woodlands STM diagrams for the present REA (Figure 7-2 through Figure 7-6) mostly follow the STM diagram conventions as in Chapters 5 and 6, above. Rectangles represent ecological states of plant community composition. Arrows represent stressors that are ecological drivers outside their normal range of variation that transition ecological condition from one state to another. However, unlike the STM diagrams in Chapters 5 and 6, the Pinyon-Juniper Woodlands STM diagrams do not display the system states in distinct rows and columns, because the transitions among states are not easily characterized along gradients. Additionally, the Pinyon-Juniper Woodlands STM diagrams include information on stand age, because transition probabilities among pinyon-juniper states are affected significantly by this variable. “Years” in the diagrams indicates years since last stand-replacing fire. “In Growth” refers to increases in tree population density.

Multiple stressors and processes influence transitions among Pinyon-Juniper Woodlands ecological states. These stressors include the Change Agents identified for assessment in the Chihuahuan Desert REA. Among these stressors, however, wildfire stands out as the most potent driver of succession among system states. Woody plants in general are increasing in the Chihuahuan desert (Fuchs 2002), as seen in the two types of settings in which pinyon-juniper woodlands generally occur: (1) areas where pinyon-juniper woodlands have expanded into areas formerly covered by grasslands or shrublands when

fire frequency became too infrequent to kill advancing tree seedlings; and (2) areas where shallow, rocky soils or terrain limit the fuels available for fire.

Pinyon-juniper woodlands have expanded widely into former grasslands and shrublands over the last 100-140 years (Cottam and Stewart 1940, Davis and Turner 1986, Gori and Bate 2007, Gottfried and Ffolliott 1995, Huffman et al. 2006, Landis and Bailey 2005, Leopold 1924, Springfield 1976, West et al. 1975). This expansion has reduced shrubs and grasses in some areas, leaving dead shrubs and bare soil between the trees with an increased opportunity for erosion (Allen 2001, Brockway et al. 2002, Gori and Bate 2007, Jacobs et al. 2002, Leopold 1924, West et al., 1975) and potentially reduced historic plant community biodiversity.

More specifically, juniper invasion into former grasslands or shrublands usually results in competition among plants for resources, eventually resulting in a decrease in understory vegetation and potentially a decrease in plant species diversity as well. These effects vary by location. For example, the effect of juniper invasion on the understory plant community varies with the depth of the soil to a restrictive layer (Miller et al. 2005), which affects resource availability and competition among plant species. Juniper invasion is a gradual process. Juniper density and cover increase over time, with juniper eventually becoming the dominant cover type as early as 70 years after the last stand-replacing fire (Barnes and Cunningham 1987, Barney and Frischknecht 1974, Blackburn and Tueller 1970, Erdman 1970, Floyd et al. 2000, Gori and Bate 2007, Gottfried 1987, Gottfried and Severson 1993, Schott 1984). Pinyon trees usually appear later than juniper trees, if they invade. Some research suggests that pinyon trees can become established only after the development of a suitable microclimate surrounding the base of juniper trees and shrubs 60-80 years after juniper establishment (Gori and Bate 2007, Landis and Bailey 2005, Padien and Lajtha 1992). Cover by a nurse plant or woody debris also increases pinyon seed germination and seedling survival (Callaway et al. 1996, Gori and Bate 2007, Meeuwig and Bassett 1983, PAdien and Lajtha 1992). Although pinyon trees may establish later than juniper successional, pinyon rates of increase generally eventually exceed those of juniper (Blackburn and Tueller 1970, Howell 1941, Jameson 1965, Lymbery and Pieper 1983, Meagher 1943, Tausch et al 1981). However, pinyon does not always invade the areas where juniper invades and juniper stands without pinyon can be common in some areas (Allred 1996, Heerwagen 1956, Sallach 1986).

The vegetation in areas into which pinyon-juniper woodlands have expanded can transition through multiple ecological states. Severe disturbances such as severe fire can return woody-plant-dominated communities successional back to herbaceous dominance (Gori and Bate 2007). This can restart the progression from fire-tolerant herbaceous or shrub plants to eventual tree dominance, if the historic disturbance regime that formerly maintained the plant community in an herbaceous or shrubland state (prior to woodland encroachment) is not reinstated.

Pinyon-juniper persistent woodlands commonly have trees of various ages with some trees >300 years of age, in contrast to pinyon-juniper woodlands in areas of expansion. Some studies have estimated that pinyon-juniper persistent woodlands take 200-300 years to return to late successional stages after stand-replacing fire (Gori and Bate 2007), which are very rare.

The following paragraphs discuss how each Change Agent affects Pinyon-Juniper Woodland states and transitions, as illustrated in Figure 7-2 to Figure 7-6.

7.3.1 Climate Change

Climate variation historically has strongly shaped the distribution of pinyon-juniper woodlands throughout their range (Betancourt et al. 1993). This sensitivity of pinyon-juniper woodlands to climate variation strongly suggests that climate change will cause further change in the distribution of these woodlands in the ecoregion. The sensitivity of pinyon and juniper to climate variation is well documented, as a result of the suitability of the growth rings in these species for dendrochronology and dendroclimatology.

Pinyon and juniper trees during the last major ice age, ca. 40,000-11,000 years before present, grew on what we now consider desert elevations (300-1700 m) (Betancourt et al. 1993). These former distributions have little or no overlap with current distributions in the Chihuahuan desert and sometimes involved different species or subspecies (Betancourt et al. 1993). Approximately 11,000 years before present, at about the time of the Clovis-age drought (*sensu* Haynes 1991), pinyon trees died out in southern New Mexico at these lower elevations. Junipers and oaks survived longer but eventually died out from these same lower elevations around 8,000 years before present (Betancourt et al. 1993). Temperatures have been warming since the peak of the Little Ice Age in the mid-1880s (Ghil and Vautgard 1991, Miller and Wigand 1994). After about 1880, a decline in precipitation resulted in less production of fuels and also to soil erosion in some areas, reducing the ability of the affected communities to carry fires (Gori and Bate 2007). Drought thus appears to have played a dominant role in the changes in distribution of pinyon-juniper over the centuries (Betancourt et al. 1993).

Cyclical climate patterns such as El Niño events and expanded circumpolar vortices usually result in increased fall, winter, and spring precipitation in the ecoregion (Andrade and Sellers 1988, Cayan and Webb 1992, Webb and Betancourt 1992). These events also can result in slightly less summer precipitation (Sheaffer and Reiter 1985). La Niña events and contracted circumpolar vortices usually result in the opposite effect (Webb and Betancourt 1992). Historically, the amount of land burned by wildfires—including woodlands—tends to be higher during La Niña events with low precipitation, and lower during El Niño events with higher precipitation (Swetnam and Betancourt 1990). Ideally these relationships would provide guidance to land managers when considering vegetation control treatments. However, seasonal and annual precipitation patterns do not vary reliably with El Niño or La Niña cycles, making predictions risky for scheduling vegetation control treatments in pinyon-juniper woodlands based on forecasts of El Niño or La Niña (Betancourt et al. 1993).

Drought, as noted above, appears to have played a dominant role in the changes in distribution of pinyon-juniper over the centuries (see Insect/Drought in Figure 7-2 to Figure 7-6) (Betancourt et al. 1993). Pinyon and juniper trees can live for hundreds of years, which increases the opportunity for them to experience severe disturbances such as drought (Betancourt et al. 1993). Severe, long duration droughts like those that occurred during 1575-1595, 1667-1681, 1730-1750, and 1942-1956 appear to have killed many trees and reset the population structure and extent of distribution. The drought during the 1950s resulted in a die-off of pinyon trees across the lowest 100 m of elevation of its distribution in

the Sevilleta Refuge where the substrate was limestone (Betancourt et al 1993). The extreme drought of the 1950s in fact proved that many plant species in the ecoregion are susceptible to death related to drought. Drought stress resulted in the death of many range grasses (Herbel et al. 1972, Lohmiller 1963, Neilson 1986, Young 1956), saguaro cacti (*Carnegia gigantea*; Schulman 1956), and conifer plants including one-seed juniper (*Juniperus monosperma*) and Chihuahua pine trees (*Pinus leiophylla*; Marshall 1957). The 1950s drought also moved the lower ponderosa pine (*Pinus ponderosa*) boundary uphill by about one mile of elevation (Allen and Breshears 1998), an effect that lasted >40 years even though precipitation was greater during 1960-1990 than during the drought (Allen and Breshears 1998). Less severe droughts during 1996 and 2002-2003 also have resulted in the death of pinyon and some juniper trees at lower elevations (Breshears et al. 2005, Gitlin et al. 2006, Mueller et al. 2005, Ogle et al. 2000). Pinyon trees were affected across > 2 million acres during the drought in the early 2000s (Gori and Bate 2007, Shaw et al. 2005).

Other factors also may have made the effects of the drought of the mid-1990s to early 2000s on pinyon-juniper woodlands more severe throughout the southwestern U.S., including greater tree density after higher precipitation during 1978-1995, fire suppression, higher temperatures intensifying water stress, and beetle infestations (Breshears et al. 2005, Breshears and Allen 2002, Cobb et al. 1997, Gori and Bate 2007). Drought in fact influences plant stress in multiple ways (see Insect/Drought in Figures 7-2 to 7-6). Drought can weaken plants and result in increased insect infestations or pathogens. A pinyon beetle (*Ips confusus*) infestation, for example, occurred at the same time as a large dieoff of pinyon trees during the drought in the 1950s at Bandelier National Monument (Chansler 1964). Drought can increase the heat load plants experience because the soil is dry with limited soil water evaporation and associated evaporative cooling (Betancourt et al. 1993). During severe drought conditions, trees cannot take up sufficient water to account for water use, resulting in cavitation in the xylem and plant dieback (Sperry and Tyree 1988).

7.3.2 Uncharacteristic Wildfire

Fire, especially severe fire, largely determines succession in pinyon-juniper woodland communities (see Succession in Figures 7-2 to 7-6). The distribution, composition, and condition of pinyon-juniper woodlands in the Chihuahuan desert are highly sensitive to the seasonal timing, frequency, and severity of wildfire – and therefore highly sensitive to changes in these variables, as shown throughout the STM diagrams for Pinyon-Juniper Woodlands, Figures 7-2 to 7-6.

Fire frequency in the ecoregion has declined since ca. 1880, as discussed above. Wildfires of low to mixed severity have become rare because livestock grazing has reduced fuel loads, fire suppression has limited fire spread, and droughts have reduced production of fuels and also promoted soil erosion in some areas, leaving insufficient understory fuel loads to carry fires (Gori and Bate 2007). These changes have allowed pinyon-juniper trees to spread into adjacent plant communities and become more abundant in areas already invaded. Pinyon-juniper invasion often reduces herbaceous cover, especially on shallow soil and this can limit the opportunity to reintroduce the historic fire regime due to insufficient understory fuels to carry fire. These changes in vegetation structure and composition negatively affect grassland obligate wildlife while helping woodland obligate wildlife, by altering such

factors as food availability, cover, and nesting sites. Additional discussion of fire related changes in wildlife habitat are included in the wildlife conceptual models and associated narratives, Chapters 12-17.

Stand-replacing fires often return pinyon-juniper shrub and persistent woodlands back to annual plant dominance (restarting secondary succession) for approximately 3 year (see Stand-replacing fire in Figure 7-2 to Figure 7-6) (Arnold et al. 1964, Barney and Frischknecht 1974, Erdman 1970). As succession continues after a stand-replacing fire, perennial herbaceous plants typically dominate by years 5-6, followed by dominance by a combination of perennial herbaceous and shrub plants (Arnold et al. 1964, Barney and Frischknecht 1964). Shrubs that resprout or produce fire-stimulated seeds may reappear about 11 years after a stand-replacing fire (Barney and Frischknecht 1974) (Gori and Bate 2007). Examples of these types of shrubs include shrub live oak (*Quercus turbinella*), rabbitbrush (*Chrysothamnus viscidiflorus*), and snakeweed (*Gutierrezia sarothrae*) (Gori and Bate 2007). Shrubs often become dominant approximately 35 years after a stand-replacing fire with shrub dominance potentially lasting more than 100 years (Barney and Frischknecht 1974, Erdman 1970, Gori and Bate 2007, Koniak 1985, Schott 1984, Young and Evans 1978).

Juniper competition with shrubs in pinyon-juniper woodland communities usually results in shrubs beginning to die 45-65 years after a stand-replacing fire. Juniper trees usually reappear before pinyon trees approximately 11 years after a stand-replacing fire. Juniper density and cover increase over the subsequent years through in-growth or infill, with juniper becoming the dominant cover type as early as 70 years after stand-replacing fire (see In Growth in Figure 7-2 to Figure 7-6 (Barnes and Cunningham 1987, Barney and Frischknecht 1974, Blackburn and Tueller 1970, Erdman 1970, Floyd et al. 2000, Gottfried 1987, Gottfried and Severson 1993, Schott 1984). Alligator juniper can resprout after fire and may dominate sooner than 70 years (Miller 1999). As noted earlier in this chapter, the microclimate surrounding the base of juniper trees and shrubs 60-80 years after juniper establishment may create conditions particularly favorable for pinyon tree establishment (Gori and Bate 2007, Landis and Bailey 2005, Padien and Lajtha 1992). As also noted earlier in this chapter, cover by a nurse plant or woody debris also increases pinyon seed germination and seedling survival (Callaway et al. 1996, Gori and Bate 2007, Meeuwig and Bassett 1983, Padien and Lajtha 1992). Although pinyon trees may establish later than juniper following a stand-replacing fire, pinyon rates of increase generally exceed those of juniper (Blackburn and Tueller 1970, Howell 1941, Jameson 1965, Lymbery and Pieper 1983, Meagher 1943, Tausch et al. 1981). In time, pinyon trees may dominate and eventually limit juniper growth and recruitment (Erdman 1970, Gori and Bate 2007, Schott 1984, Tausch and Tueller 1977).

Tree canopy cover typically remains at <3% after a stand-replacing fire for up to 50 years (Goodrich and Barber 1999, Gori and Bate 2007). Tree canopy cover typically reaches 5-17% after 65 to 90 years following a stand-replacing fire (Arnold et al. 1964, Tress and Klopatek 1987), and reaches 30-43% cover after enough time has passed that evidence of a stand-replacing fire has weathered away (Gori and Bate 2007). Some studies have estimated that shrub or persistent woodlands take 200-300 years to return to late successional stages after a stand-replacing fire (Gori and Bate 2007).

Plant community composition, both within and surrounding pinyon-juniper woodland communities prior to a stand-replacing fire, affects the rate and pattern of succession in pinyon-juniper woodland

communities following the fire (see Figure 7-2 to Figure 7-6). Where shrubs dominate without perennial grasses before a stand-replacing fire, succession after the fire may proceed directly from annual plant to shrub dominance without an intermediate state of perennial grass dominance (Barney and Frischknecht 1974). The presence of shrubs such as rabbitbrush and oaks that resprout after fire, versus non-sprouting shrubs such as sagebrush and bitterbrush that repopulate through seeds and seedling establishment, can reduce the time until shrubs again dominate a site after a stand-replacing fire (Ward 1977, Wright 1972, Wright et al. 1979). Where pinyon and juniper trees survive a non-stand-replacing fire (less severe), these two species can return to former population levels within 20-30 year (Dwyer and Pieper 1967, Miller 1999, Schott and Pieper 1987, Tausch and Tueller 1977). Where severe disturbances affect large areas, propagules may have to arrive from distant sources to repopulate an area, with the rate of propagule dispersal influencing the rate of succession (Barney and Frischknecht 1974, Huber et al. 1999). Soil characteristics also affect the rate of succession following fire. For example, succession proceeds faster after fire on deep, mesic soils compared to thin, dry soils (Gori and Bate 2007, Graves 1917, Harper and David 1999) with less resources available for plant growth.

Succession is less well studied in pinyon-juniper grass savannas than in the other variants of pinyon-juniper woodlands, but is thought to be similar in succession to pinyon-juniper shrub and persistent woodlands with some exceptions. Multiple studies indicate that if perennial herbaceous plants dominate prior to fire in pinyon-juniper grass savannas, perennial herbaceous vegetation would be the first to return to dominance rather than annual vegetation (Barney and Frischknecht 1974, Everett 1987, Schott and Pieper 1986). Additionally, low-severity surface fire presumably occurs frequently in savannas and maintains the density of pinyon-juniper at low levels (see Surface Fire in Figure 7-2). Otherwise without fire, woody species in pinyon-juniper grass savannas would increase (Humphrey 1958, McPherson 1997). Additionally, stand-replacing disturbances appear to be rare in savannas, as evidenced by some the presence of 300-500 year old live trees (Gori and Bate 2007, Leopold 1924, McPherson 1997, Wilkinson 1971). Low availability of soil moisture may also limit pinyon-juniper density in some pinyon-juniper grass savannas (Gori and Bate 2007, Johnsen 1962, Romme et al. 2007).

Pinyon-juniper woodlands generally are not considered to be resilient to changes in wildfire regime. However, several factors can interact to make individual pinyon-juniper woodland stands more or less resilient to fire. Woodland species such as alligator juniper that readily resprout after disturbance are more resilient to fire than non-resprouting trees, so long as the fire is not severe enough to kill all woody plants and their roots. Drought and warm temperatures in the past have resulted in the death of many pinyon and juniper trees, as discussed above (Floyd et al. 2009, Shaw et al. 2005). Trees weakened or killed by drought or elevated air temperatures no longer compete as well or at all for resources with understory plants. Increases in drought frequencies and/or durations, or higher air temperatures therefore would be expected to reduce woodland resiliency to fire. Climate change could also affect the seasonal distribution of fire potential. Major fire seasons generally occur following higher than average wet periods, because such periods because of increased biomass production (i.e., fuel loads; Crimmins and Comrie 2004). The natural fire regime during the 1942-1957 drought was different than the fire regime during or shortly after the wet years of 1983-1989 with record precipitation from El Niño weather patterns (Crimmins and Comrie 2004). However, dry years are not a prerequisite for major fire

seasons, because April-June is an annually occurring dry season that is sufficient to dry out fuels enough to allow burning (Crimmins and Comrie 2004). Increases in understory vegetation, perhaps as consequences of wet weather or incursions of invasive grasses, result in greater concentrations of fine fuel loads. These loads can lead to more frequent and continuous fires, which can harm trees already weakened by other factors. Burned, weakened trees have less or no ability to resprout or produce seed after fire.

Biotic agents such as insects interact with the effects of climate on the ability pinyon-juniper woodlands to withstand or recover from disturbances like fire (Floyd et al. 2009, Gaylord et al. 2013). Several native insects and pathogens affect pinyon and juniper trees (see Insect/Drought in Figure 7-2 to 7-6). The species that attack pinyon trees include the stem and cone moths (mainly *Dioryctria albovittella*), pinyon cone moth (*Eucosma bobana*), pinyon Ips bark beetle (*Ips confusus*), pinyon twig beetles (*Pityophthorus* spp. and *Pityogenes* spp.), pinyon needle miner (*Coleotechnites edulicola*), pinyon needle scale (*Matsucoccus acalyptus*), black stain root disease (*Leptographium wagneri*), pinyon dwarf mistletoe (*Arceuthobium divaricatum*). Pinyon mortality as a result of insect injury is mainly caused by pinyon Ips beetles (Rogers 1993). Severe infestations of pinyon Ips bark beetle have been recorded in recent history during the droughts of the 1950s (Allen 1989, Swetnam and Betancourt 1998), 1996, and 2000-2003 in northern New Mexico and nearby states (Gori and Bate 2007). The density of pinyon trees and their size are also related to pinyon Ips bark beetle infestations (Negron and Wilson 2003). A smaller number of insects and pathogens affect juniper trees. These species include twig beetles (*Phloeosinus* spp.), twig pruners (*Styloxus bicolor*), western cedar borer (*Trachykele blondeli*), juniper mistletoe (*Phoradendron juniperinum*), and rusts (*Gymnosporangium* spp.) (Gottfried et al. 1995, Negron 1995, Rogers 1995, Ronco 1990, Shaw et al. 2005, Weber et al. 1999) (Gori and Bate 2007). While several species are known to affect pinyon and juniper trees, how they alter tree populations are less understood. Species such as the pinyon Ips bark beetle, twig beetles, needle miners, and needle scale usually do not cause large problems and occur in small numbers. However if the trees are stressed due to lack of water, these species can increase and may defoliate or kill trees across large areas (Allen and Breshears 1998, Furniss and Carolin 1977, Gori and Bate 2007, Gottfried et al. 1995, Hagle et al. 2003, Rogers 1995, Waring and Cobb 1992, Wilson and Tkacz 1992). In turn, trees stressed by insect infestations are more vulnerable to further stress or mortality from drought and fire.

Human activities also affect fire potential. Groundwater consumption and altered watershed soil conditions can result in lower water tables, causing additional stress to woodland plants. The building of roads and structures increases human presence and ignition risks that can result in unusually high fire frequency around developed areas. Fire suppression starting during the 1900s, especially after World War II, has reduced fire frequency (Miller et a. 2000) and increased fuel loads in many areas. The reduced frequency of low- to mid-severity fire has led to rare but severe fires like crown fires where trees have invaded. Crown fires in these areas are uncharacteristic of the historic fire regime before tree invasion. These extreme fires can lead to changes in plant community composition, wildlife habitat, soil erosion, and loss of life and property. Land managers have also applied treatments other than prescribed fire in many areas to reduce invasion by woody species and maintain herbaceous vegetation. These treatments include cutting down woody species with chainsaws or loppers, using tractors to pull

anchor chains across woodland stands (aka “chaining”) to uproot trees, shredding woody material, and using mowers or herbicides to thin woody species density. Such treatments necessarily alter fire potential in the treated areas.

Thus, as noted above and later in this chapter, the seasonal timing, frequency, and severity of wildfire has changed, and is expected to change further in the Chihuahuan desert as a result of several factors. These factors will likely result in further changes in distribution, composition, and condition of pinyon-juniper woodlands in the ecoregion.

7.3.3 Invasive Species

As noted in Chapter 5, land managers throughout the southwestern U.S. face challenges arising from woodland invasion into former grasslands by several species including from pinyon-juniper woodland expansion (Brown 1994, ILAP STM 2012). In fact, many exotic plants have spread or been introduced into the U.S. portions of the Chihuahuan desert. Some seeds have been purposefully introduced for cultural purposes such as food, conservation, forage, or decorative landscaping while others have been accidentally introduced. Introduced plants species become established in a new environment only if they are adapted to that environment, and only if the resources (e.g., soil conditions, water, etc.) they need are also available at the right times of the year. Introduced plants become invasive if they can use resources in ways that native plants do not, or if they can out-compete native plants for resources. A new plant can perhaps grow at colder or hotter temperatures than can native plants, or exploit resources at a time when those potential resources are not available to the existing plant species. For example, a new plant may be able to tap into resources at a different soil depth than can native plants, or more rapidly colonize disturbed soils than can native plants. Some of the exotic grasses and herbaceous plants introduced into the U.S. portions of the Chihuahuan desert include burdock, cocklebur, dandelion, Russian thistle, cheatgrass, and summer cypress (Allred 1996). One study estimated 88 non-native plant species can occur in pinyon-juniper woodlands (Scurlock 1993). A state-wide list of noxious weeds is available at <http://www.nmda.nmsu.edu/apr/noxious-weed-information/> for New Mexico and http://texasinvasives.org/plant_database/ for Texas.

Invasive plant species, whether exotic or native (i.e., plants native to a different but nearby ecological system), often displace at least part of the historic plant community and modify nutrient cycling, water cycling, and the disturbance regimes that used to help maintain the historic plant community (Huenneke 1995). One study estimated that, of the grass species in New Mexico, 103 or one-fourth are exotic (Allred 1993). Lehmann lovegrass is a well-known example of a grass purposefully introduced to aid rangeland management that has since become an invasive grass that can dominate rangelands. Lehmann lovegrass increases fine fuel loads that help sustain fire, which can increase fire frequency beyond historic levels and harm plant species like black gramma that are not fire resilient. Saltcedar (aka “tamarisk”) is an example of a plant that was introduced to help with erosion control in watershed drainages but has become wide-spread in New Mexico (Allred 1996) degrading riparian habitat. Mesquite is an example of a native plant that has become invasive and replaces grasslands in the Chihuahuan Desert. Mesquite invasions in some areas have converted grasslands into mesquite dunelands especially in areas with sandy soil. Pinyon-juniper species are native trees but large portions

of the current distribution of pinyon-juniper woodlands are invaded grasslands or shrublands. The effects of pinyon-juniper woodland encroachment into grasslands and shrublands include plant community modification, habitat alteration, erosion, and changes to the historic disturbance regime.

7.3.4 Development

Native Americans influenced vegetation in pinyon-juniper woodlands and also may have influenced the distribution of these woodlands prior to European settlement. Prehistoric residents of the southwestern U.S. harvested fuelwood from pinyon-juniper woodlands, for example (Betancourt and Van Devender 1981, Samuels and Betancourt 1982). Native Americans also likely transported pinyon nuts from areas of large mast production to other areas (Betancourt et al. 1993). Sporadic climate events trigger mast events in pinyon trees that provide an important food source for animals (Betancourt et al. 1993). Pinyon nuts were an important food source for Native Americans and the probable transportation of the pinyon nuts could have expanded the distribution of pinyon trees prior to European settlement where appropriate soil and climate conditions existed for pinyon seedling establishment (Betancourt et al. 1993).

Chapters 2 and 3 summarize information on the types and extent of development in the U.S. portions of the Chihuahuan desert following European arrival. The spatial distributions of the different types of development in the ecoregion reflect the interaction of several geographic and economic factors. In contrast to the impacts of development on Chihuahuan desert grasslands and scrublands, however, development in the ecoregion has not resulted in widespread conversion of former areas of pinyon-juniper woodlands to human use. Pinyon-juniper persistent woodlands historically have covered areas of rugged and shallow soil terrain with little understory vegetation. These areas occur at higher elevations and away from water courses where the most significant residential, urban, and agricultural development has occurred. These areas also do not occur along the residential-commercial-industrial development corridor along the U.S.-Mexico border, nor are they affected by oil and gas production or their associated distribution systems in and around the Permian Basin. On the other hand, fire suppression across pinyon-juniper woodlands, fragmentation of pinyon-juniper woodlands by roads and other transportation and utility infrastructure, and efforts to address pinyon-juniper woodland encroachment into grasslands have all indirectly affected ecological conditions in pinyon-juniper woodlands throughout the U.S. portions of the Chihuahuan desert. The building of roads and structures in pinyon-juniper woodlands limits the safe use of prescribed fire even though this increase in human presence increases ignition sources that may result in unusually high fire frequency around these human developments. Fire suppression starting during the 1900s, especially after World War II, generally has reduced fire frequency (Miller et al. 2000) while increasing fuel loads in many pinyon-juniper woodland areas. This in turn has led to rare but severe fires like crown fires where trees have invaded. Crown fires in these areas are uncharacteristic of the historic fire regime before tree invasion. These extreme fires can lead to changes in plant community composition, wildlife habitat, soil erosion, and loss of life and property.

7.3.5 Excessive Domestic Grazing

The major expansions of pinyon-juniper woodlands into Chihuahuan desert in the U.S. coincided with

decreased fire frequency in the late 1800s and early 1900s (Miller et al. 2000). Reduced fire frequency has been connected with excessive livestock grazing, which reduces fine fuels that help carry fire, and with later anthropogenic fire suppression that has allowed trees to spread and increase in density and cover to the detriment of herbaceous cover (Gori and Bate 2007, Miller 1999). Excessive livestock grazing also interferes with the competition for resources between desired understory plants and pinyon and juniper trees. Because herbivores selectively graze the plants they like to eat, Excessive livestock grazing can reduce the more palatable grasses, which are commonly the plant species land managers want to maintain. Excessive livestock grazing thus can weaken or eliminate the desired species and free up soil resources for less palatable and invasive woodland species that livestock graze less intensively or simply ignore. However, livestock grazing is minimal in pinyon-juniper persistent woodlands due to the lack of forage in these settings and their rough terrain. Livestock grazing in areas where pinyon-juniper woodlands have invaded grasslands and scrublands is discussed further in Chapters 5 and 6, respectively.

7.3.6 Landscape Restoration

When land managers speak of “landscape restoration” in pinyon-juniper woodlands, they generally mean restoration of grassland or scrubland conditions by reducing or removing pinyon and juniper. If pinyon and juniper are removed before trees mature and/or before woodland density increases to the point that it eliminates the desired understory vegetation, then tree control may be all that is necessary to restore the invaded land, as discussed in some detail in Chapters 5 and 6. Land managers use a variety of methods to reduce or eliminate woody vegetation from grasslands, including cutting down trees with chain saws, using tractors to pull anchor chains to uproot trees (aka “chaining”), mowing and mulching smaller woody plants, harvesting trees to use as posts or in furniture, harvesting trees for use as heating fuels, setting prescribed fires to burn non-resprouting woody plants, and applying herbicides to thin woody species density. Some of these methods may have adverse effects on existing, desired understory vegetation.

Land managers in fact often must consider what kind of plant community will develop following treatments undertaken to control woodland invasion. In general, the plant community present at the time of treatment eventually will return. If shrubs dominate without perennial grasses before fire, then after stand-replacing fire, succession may proceed from annual plant to shrub dominance skipping perennial grass dominance (Barney and Frischknecht 1974). If perennial herbaceous plants dominate prior to fire, perennial herbaceous vegetation may be the first plant community to return to dominance rather than annual vegetation (Barney and Frischknecht 1974, Everett 1987, Schott and Pieper 1986). However, there are exceptions to this general rule. For example, an unusually severe crown fire could kill all woody vegetation and nearly all other plants, leaving the area open to invasive plants and erosion. This could change the trajectory of plant community development.

Multiple factors affect the pattern and rate of plant community succession in pinyon- juniper woodlands following a severe treatment or disturbance. Pinyon and juniper trees surviving non-stand-replacing fire (less severe) can return to former population levels within 20-30 years (Dwyer and Pieper 1967, Miller 1999, Schott and Pieper 1987, Tausch and Tueller 1977). When severe disturbances are large,

propagules may have to arrive from long distances to repopulate an area, with the rate of propagule dispersal and source distances influencing the rate of succession (Barney and Frischknecht 1974, Huber et al. 1999). The rate of succession also depends on soil characteristics. Succession, for example, proceeds faster after fire on deep, mesic soils compared to thin, dry soils (Gori and Bate 2007, Graves 1917, Harper and Davis 1999) which have fewer resources available for plant growth.

Multiple factors also affect the potential for landscape restoration in areas invaded by pinyon-juniper woodlands and/or put current or future restoration efforts at risk of failure. Restored areas that lack an appropriate fire regime eventually will revert to pinyon-juniper savannas and shrub woodlands. The effects of fire on pinyon-juniper communities are illustrated in Figure 7-2 to Figure 7-6 and discussed in detail above. Historic fire frequency limited pinyon-juniper encroachment by killing most of the encroaching juvenile trees while they were susceptible to fire. After trees mature, they are resistant to burning except under extreme fire behavior. Without historic fire return intervals or repeated applications of woody species control techniques, trees eventually return to dominance.

Woody species invasion in many places in the southwestern U.S. have led to desertification with increasingly uneven distributions of soil resources. During desertification, soil resources accumulate around the base of woody plants while spaces between woody plants lose herbaceous cover and potentially also lose top soil. Over decades of tree invasion, desired herbaceous species can be reduced, especially on shallow soil, to a point where their propagules are insufficient to repopulate the site after woody species have been controlled through restoration management practices. Land managers therefore must respond to this lack of historic propagule pressure by desired species by artificially reseeding treated areas while also controlling invasive species. Invasive species (e.g., Lehmann lovegrass), if present, can respond comparatively quickly to increases in soil resource availability after the cessation of soil resource uptake by treated woody species. When invasive species dominate resource uptake in an area, it is difficult for desired species to establish and compete against plants that have larger or faster growing root systems. However, artificial reestablishment of native plants from seed in desert environments is problematic because of limited and variable precipitation. Consequently, land managers must begin efforts to begin treatments very soon after woody plants have begun to invade a site, and must begin follow-up efforts to reestablish desired cover soon after removal of the woody plants, before desired plant cover and propagule pressure is lost.

Climate change also may limit the potential for successful restoration to reduce woody species invasion. Dry winter and spring weather combined with subsequent wet late-summer or early-fall weather favor black grama grass seedling establishment. However, these weather patterns were more common in the late 1800s than the 1900s. The shift of summer precipitation to winter during the 1900s could explain some of the shift from black grama grass to shrub dominance, with winter rain tending to infiltrate deeper into the soil and thereby favor species with extensive root systems such many shrubs and also pinyon and juniper. Some of the vegetational shift from black grama grass to mesquite shrubs has been recorded on both grazed and ungrazed desert grasslands (Burgess 1995, Hennessy et al. 1983).

Excessive livestock grazing also can limit potential for restoring grasslands. Excessive livestock grazing in treated areas can also reduce the benefits of the treatments, for example by reducing the ability of

grasses to compete for resources and thereby increase the potential for wood plants to reestablish.

7.4 Pinyon-Juniper Woodlands Key Ecological Attributes

The variables that distinguish the ecological states from each other in the Pinyon-Juniper Woodlands STMs constitute key ecological attributes for the system. Additional key ecological attributes include critical aspects of wildfire and insect injury dynamics. The list below identifies nine (9) key ecological attributes for Pinyon-Juniper Woodlands CE on these criteria. *Characterizing the present condition of a system requires data on indicators for its key ecological attributes.*

- **System State**
 - Herbaceous versus Woody Cover Percent
 - Native versus Exotic Plant Cover Percent
 - Pinyon-Juniper Cover Percent
 - Pinyon-Juniper Stand Age
 - Vegetation Patch Size and Fragmentation
 - Soil Exposure and Stability
- **Critical Dynamics**
 - Wildfire Dynamics
 - Insect Injury Dynamics
 - Soil Erosion

Figure 7-2. Historic Pinyon-Juniper Savanna state-and-transition model.

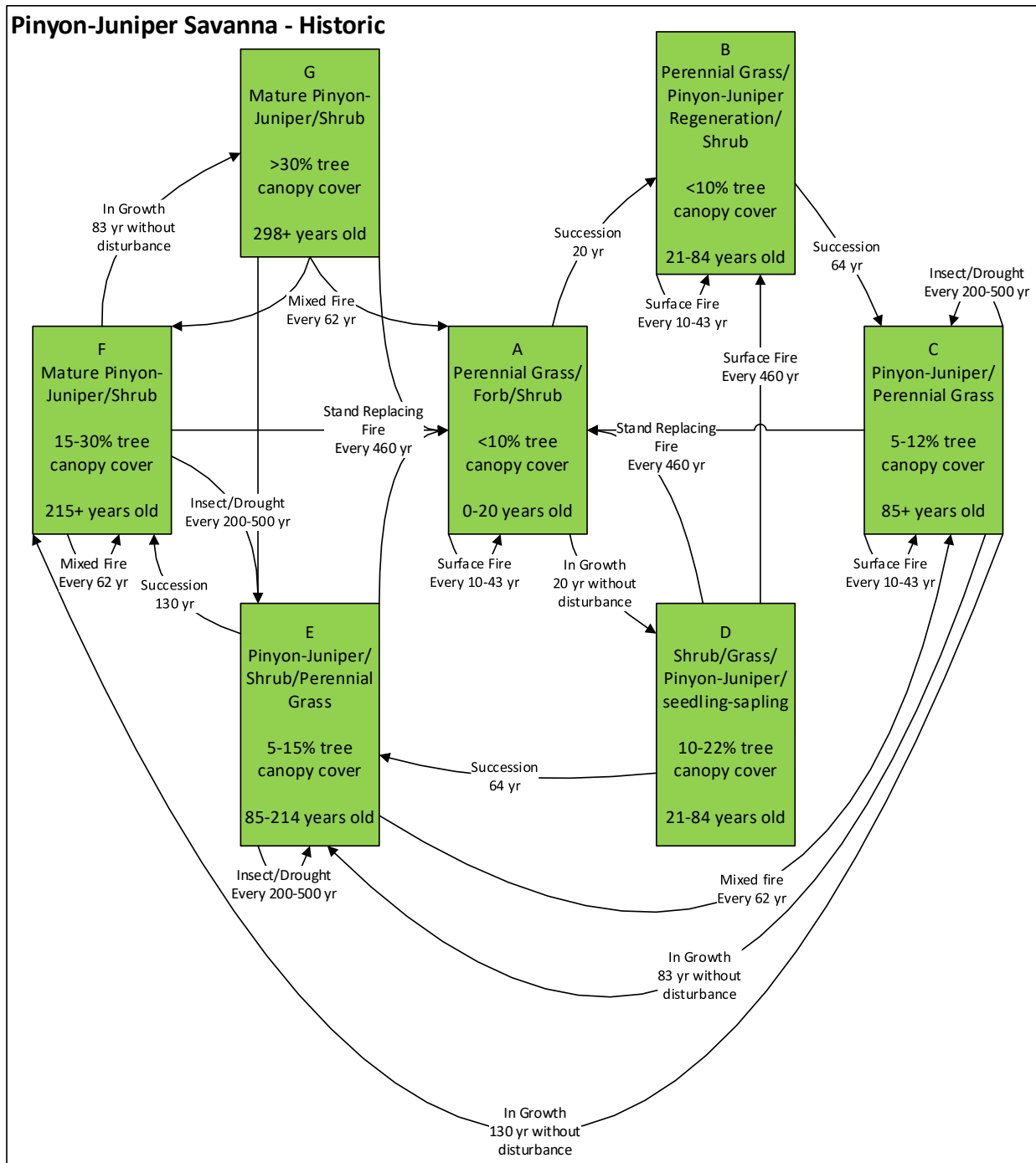


Figure 7-3. Current Pinyon-Juniper Savanna state-and-transition model.

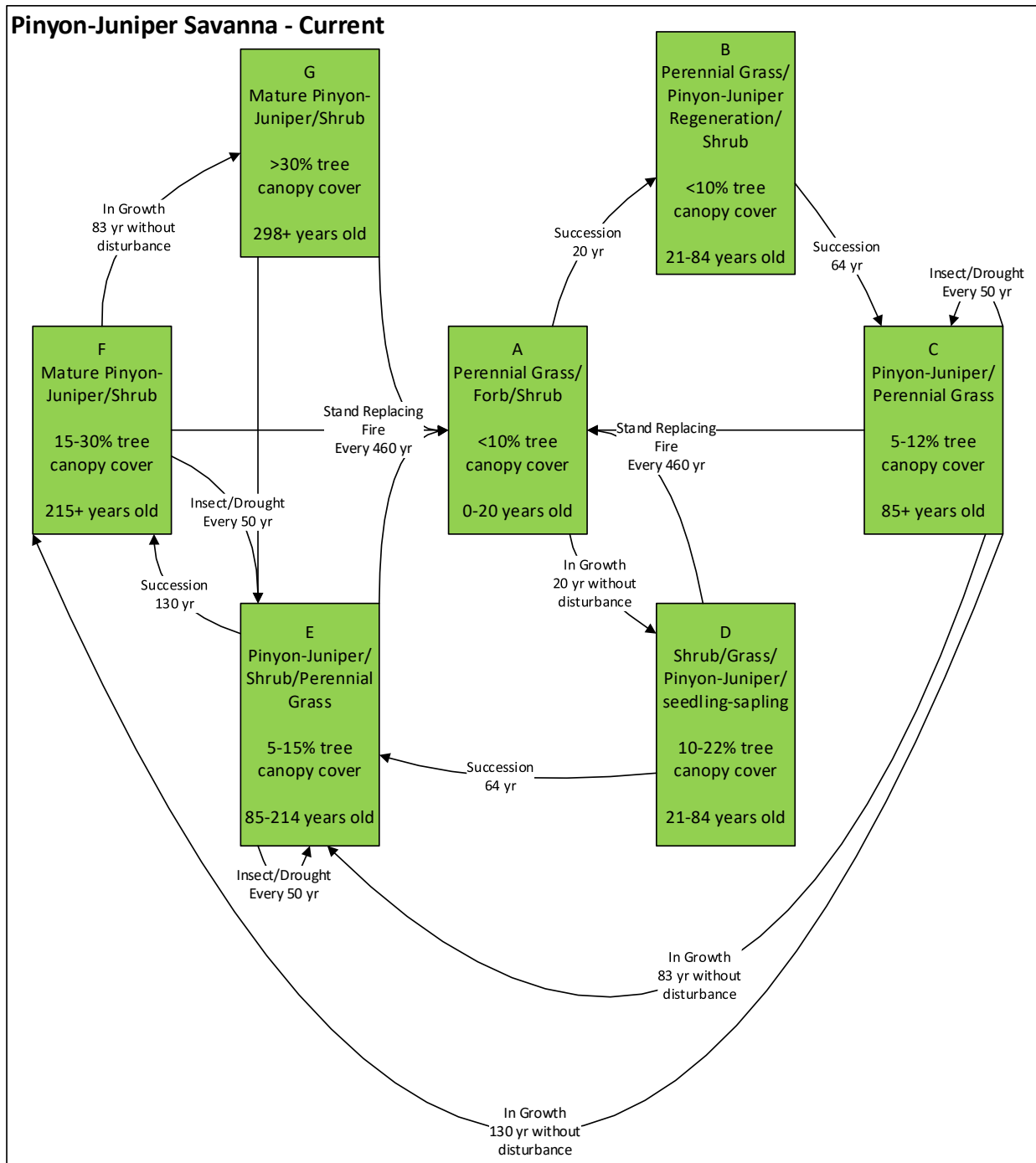


Figure 7-4. Historic Pinyon-Juniper Shrub Woodland state-and-transition model.

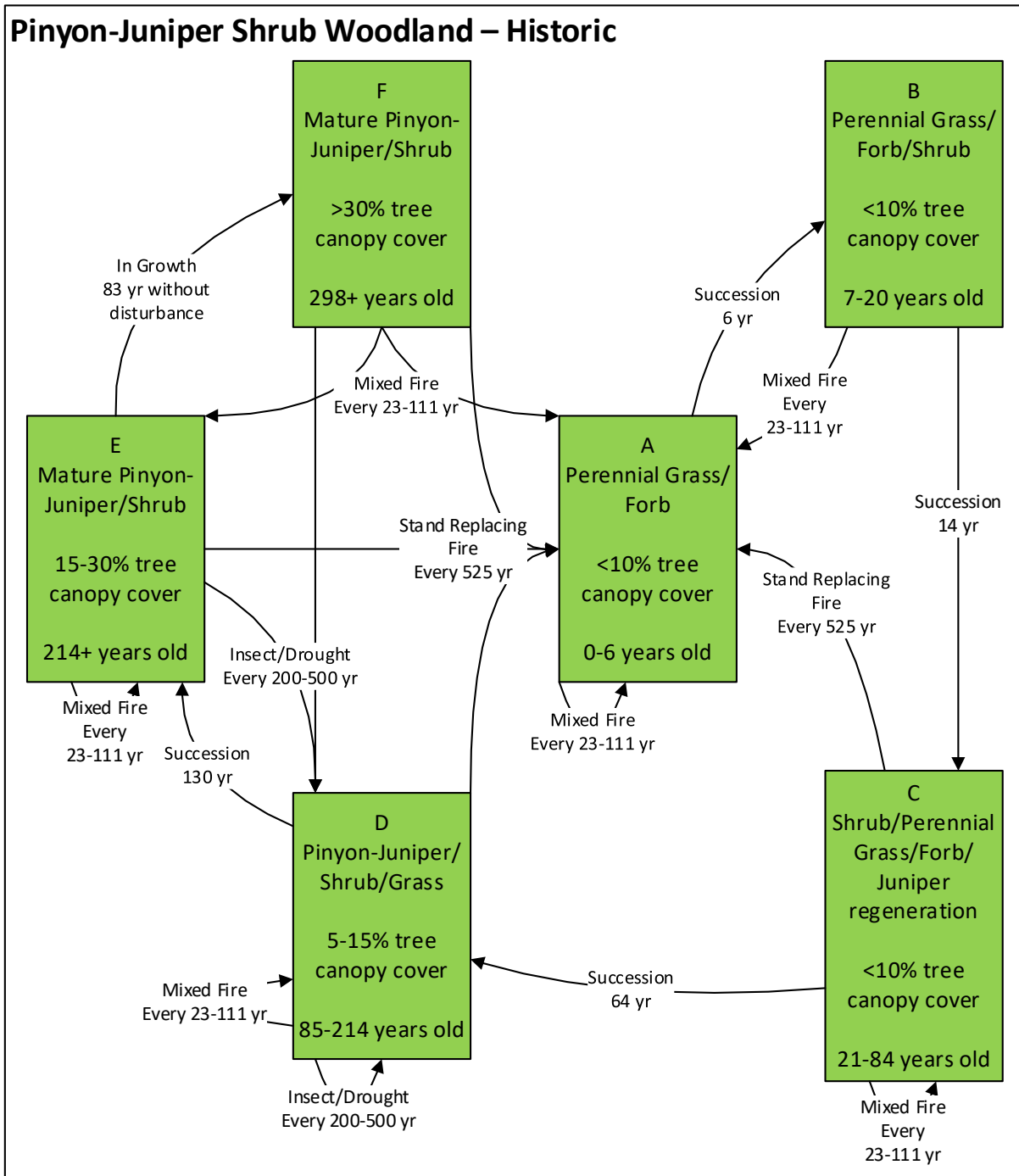


Figure 7-5. Current Pinyon-Juniper Shrub Woodland state-and-transition model.

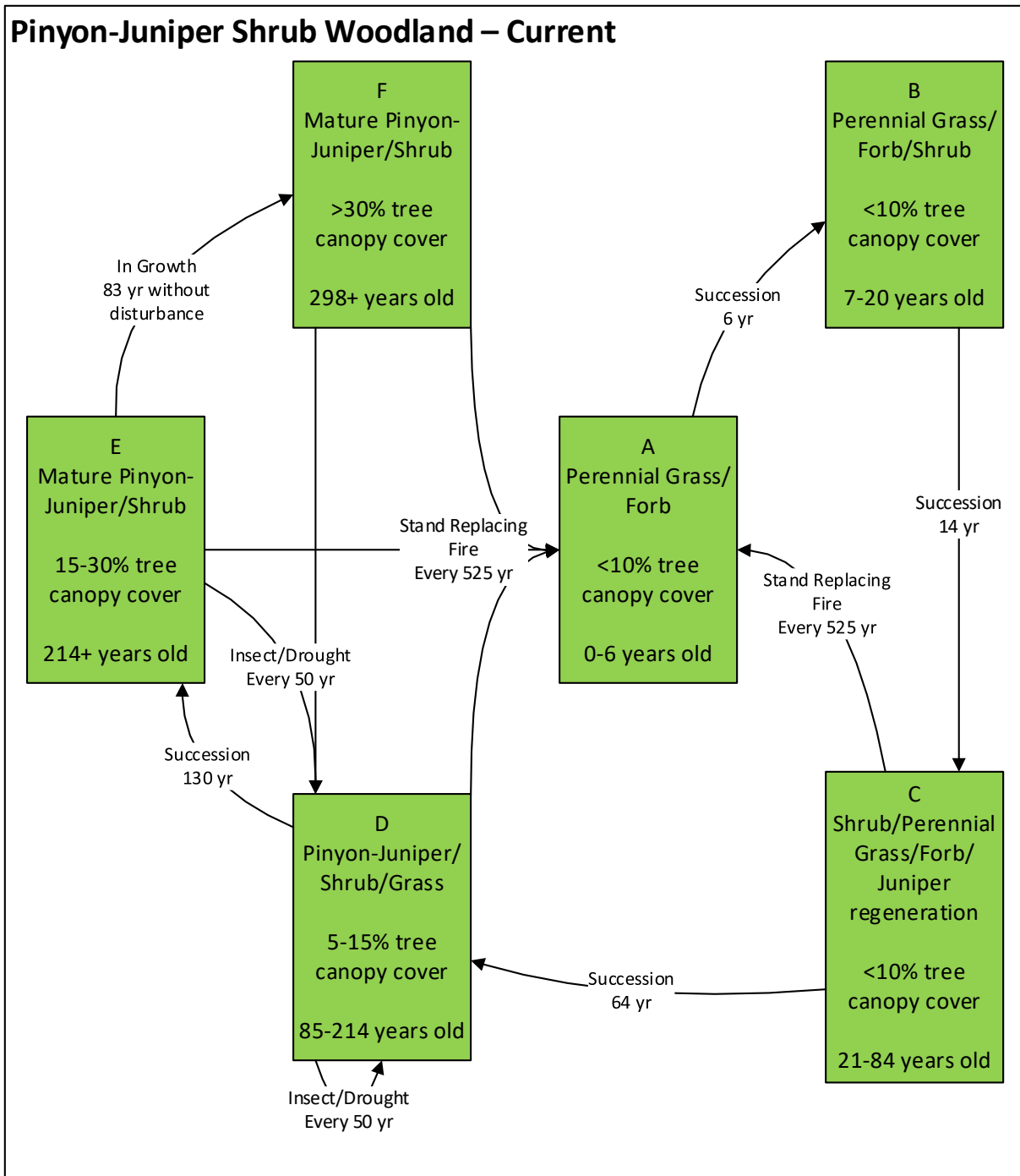
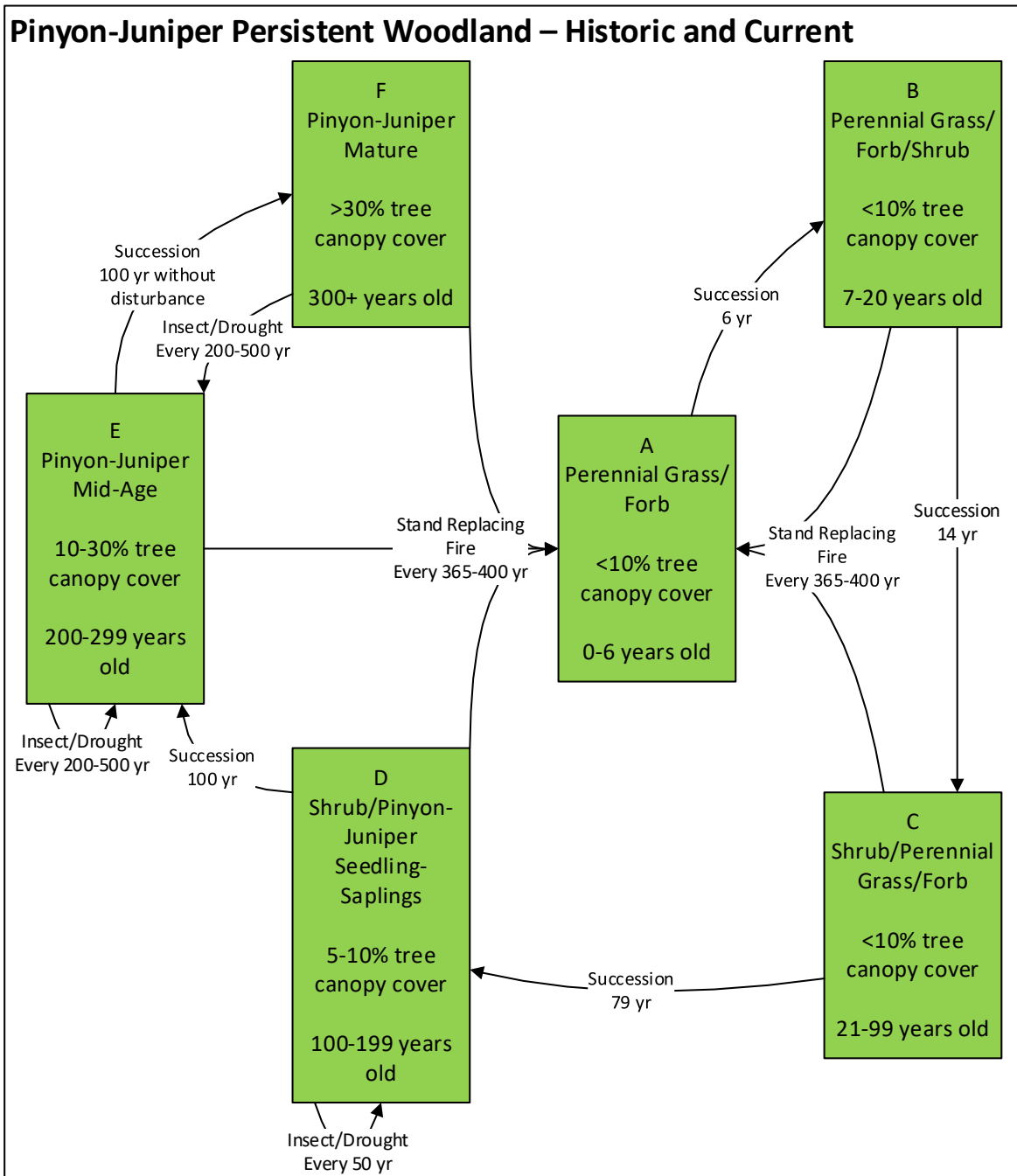


Figure 7-6. Historic and Current Pinyon-Juniper Persistent Woodland state-and-transition model.



8 Perennial Streams Conceptual Model

This chapter presents the conceptual ecological model for the Montane-Headwater Perennial Streams and Lowland-Headwater Perennial Streams CEs. A single control model and a single stressor (DLO) model address the two perennial stream types together. Montane- and lowland-headwater perennial streams have different spatial distributions; and their hydrologic and geochemical differences can select for different assemblages of aquatic and riparian fauna and flora. Their hydrologic and geochemical differences also can make them vulnerable to different – although often overlapping – impairments resulting from human activities. However, the two stream types have the same critical ecological processes, critical environmental elements, and drivers. They simply differ in the relative importance of these processes, elements, and drivers. The two system types also have the same types of ecological outcomes. Additionally, neither stream type exists in biological isolation. Both may be connected downstream to larger stream and river networks, year-round or during floods along the large rivers downstream. Where present, these connections reduce biological differences between the two types. The presentation of the perennial streams conceptual model follows the structure described in Chapter 4.

8.1 Montane- vs Lowland-Headwater Perennial Streams

Chapter 3 describes the differences between the two types of perennial stream ecological systems selected as CEs for the Chihuahuan Desert REA. The following paragraphs review these descriptions and provide examples of streams representative of each type, and discuss the fish species native to these and the other aquatic ecological system CEs.

8.1.1 Perennial Stream System Type Descriptions and Examples

Montane-headwater perennial streams in the ecoregion originate in higher-elevation, montane settings. The elevation of these settings results in higher rates of precipitation than occur at lower elevations across the surrounding valley floors, with some of the precipitation occurring as snowfall. Streams that originate in these settings receive their water as runoff from both rainfall and snowmelt, as groundwater drainage from shallow montane soils and montane bedrock aquifers, and at discrete tributary springs, some of which may be affected by geothermal activity. Cooler air temperatures, cold-air drainage along stream valleys, and montane riparian vegetation canopies help maintain relatively cool water temperatures. However, water temperature may rise or fall during periods of low-flow during the summer or winter, respectively, through the influence of high and low air temperatures. The montane topographic setting results in steeper stream gradients and higher flow velocities on average, than found in streams of comparable size in lowland settings. The water spends only a relatively short time in contact with the vegetative litter, soils, and bedrock of the montane setting, and so contains relatively low concentrations of dissolved matter. However, montane-headwater perennial streams in the ecoregion may receive discharge from groundwater sources as these streams emerge from uplands to flow through lowland settings, giving their lower reaches a combination of montane- and lowland-headwater stream hydrologic, thermal, and chemical characteristics.

Examples of montane-headwater perennial streams in the ecoregion include:

- East Fork Gila River, originating in the Black Range and flowing SW to the Gila River mainstem. (This stream conventionally is not considered part of the Chihuahuan Desert ecoregion. However, as described in Chapter 2, the analysis extent for the present REA includes all watersheds identified by a fifth-level (10-digit) Hydrologic Unit Code (HUC) that lie within or overlap the boundaries of the Level-III ecoregion. The analysis extent for the present REA consequently includes portions of the East Fork.)
- Mimbres River, NM, originating in the Black Range and flowing generally S into the Guzmán Basin, a closed (endorheic) basin.
- Rio Hondo, NM, originating in the Sierra Blanca range and flowing E to the Pecos River. It has several separately-named montane-headwater perennial tributaries, including Rio Ruidoso and Rio Bonito.
- Rio Penasco, NM, originating in the southern Sacramento Mountains and flowing E to the Pecos River.
- Delaware River, NM/TX, originating in the Guadalupe Mountains in TX and flowing NE into NM to the Pecos River.

Lowland-headwater perennial streams in the ecoregion, in contrast, originate around the bases of mountains or in surrounding valleys. Streams that originate in these settings receive their water primarily from discharges of groundwater from lower-montane bedrock, basin-fill, and other larger-scale aquifers, often at discrete springs. The groundwater discharged at such springs in the ecoregion originates as recharge at higher elevations, but may spend years, decades, or longer moving through the groundwater system before re-emerging. As a result, the water in each resulting stream emerges with a distinct but comparatively constant temperature year-round, controlled by the temperature in the aquifers through which the water has passed, some of which may be affected by geothermal activity. The water in each resulting stream also emerges with a distinct pattern of concentration of dissolved matter, controlled by the geochemistry of the groundwater pathways along which the water has traveled. Finally, because of their geological and topographic settings, lowland-headwater streams have relatively low gradients with relatively constant rates of baseflow year-round. Short-term hydrologic disturbance is rare because of the low contribution of runoff to the hydrologic regime, although flooding along large river-floodplain systems downstream can cause flooding along the lower reaches of tributary streams.

Examples of lowland-headwater perennial streams in the ecoregion include:

- Black River, NM, originating in five springs – Geyser, Washington, Rattlesnake, Blue, and Castle Springs (Cowley and Sublette 1987) – and flowing E-NE to join the Pecos River at Malaga, NM, south of Carlsbad, NM.
- Toyah Creeks, TX, with perennial discharge originating largely at the Salmon Springs complex, flowing to the Pecos River.
- Independence Creek, TX, originating at Chandler and Caroline Springs and flowing E to the Pecos River.
- Diamond-Y Draw, TX, originating at Diamond-Y Spring and flowing generally E to the Pecos River.

- Alamito, Terlingua, and Tornillo Creeks, TX, originating in canyons and flowing generally S to the Big Bend and Lower Canyons of the Rio Grande (IBWC 2006).

8.1.2 Fishes of the U.S. Portion of the Chihuahuan Desert Ecoregion

The montane-headwater perennial streams, lowland-headwater perennial streams, large rivers, and springs of the ecoregion can differ significantly in their characteristic and endemic fish species. Table 8-1 lists the native fish species of the ecoregion and their associations with perennial streams, large rivers, springs, and closed basins. Although the Pecos River flows into the Rio Grande, some species native to the Rio Grande are not also native to the Pecos River (e.g., the Rio Grande sucker), and vice versa (e.g., the Pecos pupfish). The Gila River basin is part of the Colorado River basin, lying west of the Continental Divide. Its fish assemblage therefore differs significantly from those of the Rio Grande and Pecos River basins east of the divide. Several fish species native to the rivers east of the Continental Divide that today occur in the Gila River basin are considered non-native species in the latter basin: western mosquitofish, channel catfish, flathead catfish, black bullhead, fathead minnow, and largemouth bass (Gori et al. 2014). The endorheic Mimbres River, which lies immediately along the eastern side of the Continental Divide and flows into the closed Guzmán Basin, shares fish species with both the Colorado River and Rio Grande basins.

Table 8-1. Fishes native to the U.S. waters of the Chihuahuan desert ecoregion, by drainage and type of water body.

Species	Common Name	Gila River Mainstem	Gila River Tributary	Mimbres River	Rio Grande Mainstem	Rio Grande Tributary	Tularosa Basin	Pecos River Mainstem	Pecos River Tributary	Springs
<i>Agosia chrysogaster</i>	Longfin dace	P	P	P						
<i>Anguilla rostrata</i>	American eel				E			E		
<i>Aplodinotus grunniens</i>	Freshwater drum				P			P		
<i>Astyanax mexicanus</i>	Mexican tetra				P	P		P	P	P
<i>Atractosteus spatula</i>	Alligator gar							E		
<i>Campostoma anomalum pullum</i>	Central stoneroller					P		E	P	
<i>Campostoma ornatum</i>	Mexican stoneroller				P	P				
<i>Carpiondes carpio</i>	River carpsucker				P	P		P	P	
<i>Catostomus clarkii</i>	Desert sucker	P	P							
<i>Catostomus insignis</i>	Sonora sucker	P	P							
<i>Catostomus plebeius</i>	Rio Grande sucker		P	P	P	P				P
<i>Cichlasoma cyanoguttatum</i>	Rio Grande cichlid					P		P	P	
<i>Cycleptus elongatus</i>	Blue sucker				P			P	P	
<i>Cyprinella formosa</i>	Beautiful shiner			E						
<i>Cyprinella lutrensis</i>	Red shiner				P	P		P	P	P
<i>Cyprinella proserpina</i>	Proserpine shiner					P		P	P	P
<i>Cyprinella venusta</i>	Blacktail shiner				P	P		P	P	P
<i>Cyprinodon bovinus</i>	Leon Springs pupfish							E	P	P
<i>Cyprinodon elegans</i>	Comanche Springs pupfish								P	P
<i>Cyprinodon pecosensis</i>	Pecos pupfish							P	P	P
<i>Cyprinodon tularosa</i>	White Sands pupfish						P			
<i>Dionda episcopa</i>	Roundnose minnow							P	P	P
<i>Dorosoma cepedianum</i>	Gizzard shad				P	P		P		
<i>Etheostoma grahami</i>	Rio Grande darter							P	P	P
<i>Etheostoma lepidum</i>	Greenthroat darter							E	P	P

Species	Common Name	Gila River Mainstem	Gila River Tributary	Mimbres River	Rio Grande Mainstem	Rio Grande Tributary	Tularosa Basin	Pecos River Mainstem	Pecos River Tributary	Springs
<i>Fundulus zebrinus</i>	Plains killifish				P	P		P	P	P
<i>Gambusia affinis</i>	Western mosquitofish				P	P		P	P	P
<i>Gambusia clarkhubbsi</i>	San Felipe gambusia					P				P
<i>Gambusia gaigei</i>	Big Bend gambusia									P
<i>Gambusia nobilis</i>	Pecos gambusia								P	P
<i>Gambusia speciosa</i>	Tex-Mex gambusia					P				P
<i>Gila intermedia</i>	Gila chub		P							P
<i>Gila nigra</i>	Headwater chub		P							
<i>Gila nigrescens</i>	Chihuahua chub			P						
<i>Gila pandora</i>	Rio Grande chub					P		P	P	
<i>Gila robusta</i>	Roundtail chub		E							
<i>Hybognathus amarus</i>	Rio Grande silvery minnow				P			E		
<i>Ictalurus furcatus</i>	Blue catfish				P			P		
<i>Ictalurus lupus</i>	Headwater catfish				P	P		P	P	
<i>Ictalurus punctatus</i>	Channel catfish				P	P		P	P	
<i>Ictiobus bubalus</i>	Smallmouth buffalo				P			P		
<i>Lepisosteus oculatus</i>	Spotted gar				P			E		
<i>Lepisosteus osseus</i>	Longnose gar				P			P	P	
<i>Lepomis aurtus</i>	Redbreast sunfish					P		P	P	P
<i>Lepomis cyanellus</i>	Green sunfish				P	P		P	P	P
<i>Lepomis gulosus</i>	Warmouth (possibly not native)								P	P
<i>Lepomis macrochirus</i>	Bluegill				P	P		P	P	
<i>Lepomis megalotis</i>	Longear sunfish				P			P	P	P
<i>Lucania parva</i>	Rainwater killifish							P	P	P
<i>Macrhybopsis aestivalis</i>	Rio Grande speckled chub				P	P		P		
<i>Meda fulgida</i>	Spikedace	P	P							
<i>Menidia beryllina</i>	Inland silverside				P			P		
<i>Micropterus salmoides</i>	Largemouth bass				P			P	P	
<i>Morone chrysops</i>	White bass				P			P		
<i>Moxostoma congestum</i>	Gray redbreast				P			P	P	
<i>Notropis amabilis</i>	Texas shiner							P	P	P
<i>Notropis braytoni</i>	Tamaulipan shiner				P	P		P	P	
<i>Notropis buchanani</i>	Ghost shiner							P		
<i>Notropis chihuahua</i>	Chihuahua shiner				P	P				
<i>Notropis jemezianus</i>	Rio Grande shiner				P			P		
<i>Notropis orca</i>	Phantom shiner				E			E		
<i>Notropis simus</i>	Bluntnose shiner				E			P		
<i>Notropis stramineus</i>	Sand shiner				P			P	P	
<i>Oncorhynchus clarki virginalis</i>	Rio Grande cutthroat trout					P				
<i>Oncorhynchus gilae</i>	Gila trout	E	P	P						
<i>Percina macrolepida</i>	Bigscale logperch				P			P		
<i>Pimephales promelas</i>	Fathead minnow				P	P		P	P	
<i>Pimephales vigilax</i>	Bullhead minnow				P	P		P	P	
<i>Platygobio gracilis</i>	Flathead chub				P			P		
<i>Poeciliopsis occidentalis</i>	Gila topminnow		P							
<i>Ptychocheilus lucius</i>	Colorado pikeminnow	E								
<i>Pylodictis olivaris</i>	Flathead catfish				P	P		P		
<i>Rhinichthys cataractae</i>	Longnose dace				P	P		P		
<i>Rhinichthys cobitis</i>	Loach minnow	P	P							
<i>Rhinichthys osculus</i>	Speckled dace	P	P							
<i>Xyrauchen texanus</i>	Razorback sucker	E								

Notes: "P" = reported present in one or more publications; "E" = reported formerly present but now thought to be extirpated.

Sources: Hubbs et al. 1977; Miller 1977; Minckley 1985; Cowley and Sublette 1987; Garrett and Matlock 1991; Edwards 1997;

Species	Common Name	Gila River Mainstem	Gila River Tributary	Mimbres River	Rio Grande Mainstem	Rio Grande Tributary	Tularosa Basin	Pecos River Mainstem	Pecos River Tributary	Springs
Hoagstrom and Brooks 1999; Propst 1999; Minckley and DeMarais 2000; Edwards et al. 2002, 2003; Cowley et al. 2003; Echelle et al. 2003; Hoagstrom 2003, 2009; Hubbs 2003, 2014; Edwards et al. 2004; NMDGF 2006; Dudley and Platania 2007; Paroz and Propst 2007, 2009; Zymonas and Propst 2007; Hubbs et al. 2008; Propst et al. 2009; Carman 2010; Pilger et al. 2010; Sallenave et al. 2010; Burkhead 2012; Heard et al. 2012; Hendrickson and Cohen 2012; USFWS 2010; Hanna et al. 2013; Garrett and Edwards 2014; Miyazono 2014; NatureServe 2014; Propst 2016.										

A small number of fish species have native ranges that closely approach but do not or only minimally extend into the Chihuahuan Desert Level-III ecoregion in the U.S. (see Chapter 2 for definition of ecoregion). These species conventionally are not considered native to the ecoregion. However, as described above and in Chapter 2, the analysis extent for the present REA includes all watersheds identified by a fifth-level (10-digit) Hydrologic Unit Code (HUC) that lie within *or overlap* the boundaries of the Level-III ecoregion. As a result, Table 8-1 includes four fish species that are not conventionally considered native to the ecoregion (Propst 2016): ghost shiner *Notropis buchmanii*, Rio Grande cutthroat trout *Oncorhynchus clarki virginalis*, Gila trout *Oncorhynchus gilae*, and flathead chub *Platygnathia gracilis*.

8.2 Sources of Information

The perennial stream systems control and stressor models integrate information from numerous sources: (1) the Stream and Riparian control model presented in Miller et al. (2010); (2) the conceptual models for riparian-stream conservation elements developed for the Madrean Archipelago REA immediately to the west of the Chihuahuan Desert ecoregion (Crist et al. 2014); (3) type descriptions (NatureServe 2014) for the terrestrial ecological system types recognized for the riparian corridors in the ecoregion (Dinerstein et al. 2001, NMDGF 2006, Connally, ed. 2012a; 2012b); (4) a large literature on the riparian and aquatic species and communities of the ecoregion and the natural and anthropogenic factors shaping their distribution and condition (see citations below and in Appendix 1); and (5) review comments provided by Dr. David L. Propst on a draft of this chapter (Propst 2016).

The ecological system types identified for the riparian corridors in the Chihuahuan Desert ecoregion (Dinerstein et al. 2001, NMDGF 2006, Connally, ed. 2012a; 2012b, NatureServe 2014) include:

- North American Warm Desert Lower Montane Riparian Woodland and Shrubland (International Ecological Classification Code CES302.748).
- North American Warm Desert Riparian Mesquite Bosque (CES302.752).
- North American Warm Desert Riparian Woodland and Shrubland (CES302.753).
- Western Great Plains Riparian (mixed upland and wetland) (CES303.956).
- North American Arid West Emergent Marsh (CES300.729)

The terrestrial ecological system types associated with riparian corridors in the Chihuahuan Desert ecoregion (Dinerstein et al. 2001, NMDGF 2006, Connally, ed. 2012a; 2012b, NatureServe 2014) also include North American Arid West Emergent Marsh (CES300.729) and North American Warm Desert

Ciénega (CES302.747). However, these two system types also are associated with the Springs-Emergent Wetlands CE (see Chapter 10) and therefore are not uniquely characteristic of riparian corridors in the ecoregion.

8.3 Perennial Stream Systems Control Model

Figure 8-1 shows the control model for the Chihuahuan Desert perennial stream systems. The control model shows drivers and system components in greater detail than the overarching Chihuahuan Desert wet system conceptual model. System components consist of pivotal physical, biological, and ecological characteristics of a resource, its abundance, and its distribution. Anthropogenic drivers are colored orange, to distinguish them from natural drivers (grey). As in the overarching wet system model, arrows simply represent relationships in which one model component affects or influences another. The stressor model presented later in this chapter addresses the details of these relationships.

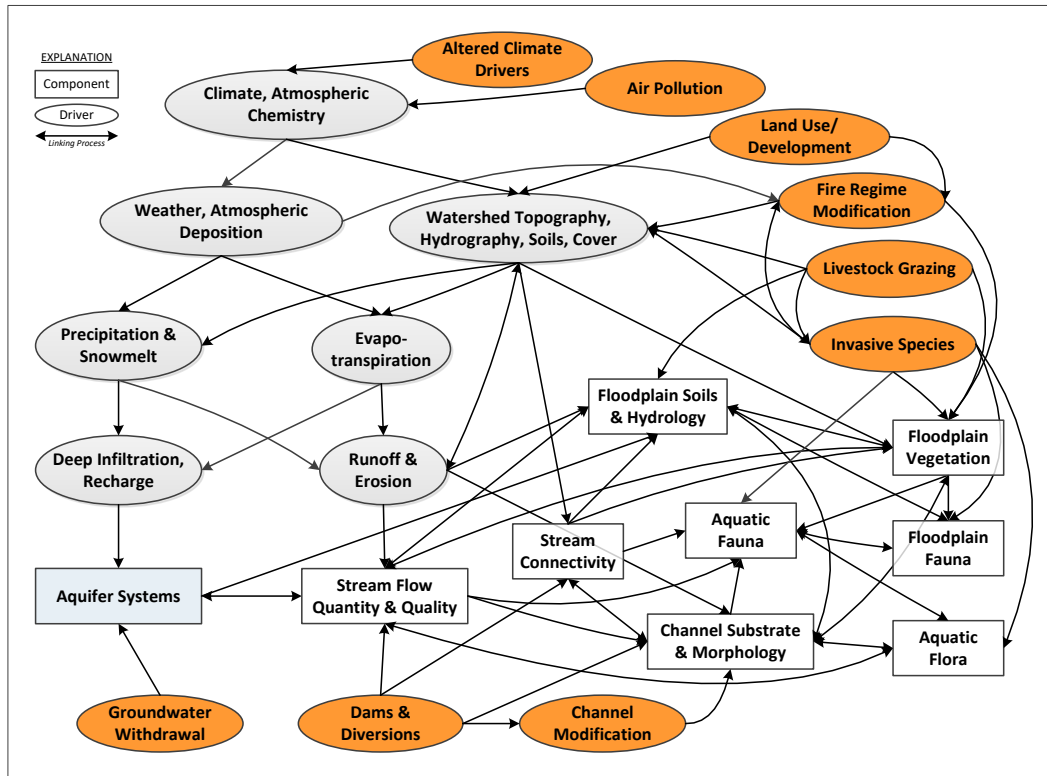
The perennial stream systems control model specifically identifies the following system components:

- **Floodplain Soils & Hydrology** refers to the mineralogy, hydrology, hydrochemistry, organic matter content, structure, stability, and biotic composition of the alluvial soils of the riparian zone. These aspects of floodplain soils both affect and are affected by other components, including stream flow quantity and quality, stream connectivity, channel substrate & morphology, and floodplain vegetation.
- **Stream Flow Quantity & Quality** refers to the daily, seasonal, annual, and longer-term variability in water flow, dissolved and suspended matter constituents of the water in the stream, and water temperature and pH. These aspects of stream flow quantity and quality affect aquatic fauna and flora; and both affect and are affected by floodplain soils and channel morphology. High-flow pulses of stream discharge, in turn, disturb soils and vegetation along stream reaches where the stream remains connected to the floodplain.
- **Stream Connectivity** refers to the ability of organisms, seeds, eggs, sediment, large woody debris, and other types of solid matter to move from one perennial reach to another, either by being carried by the currents or through self-locomotion. Falls, rapids, and dry reaches may pose barriers to up-downstream locomotion and downstream transport. This component also addresses the ability of water and solid matter to move between the stream and its floodplain. Dams, artificial levees, channelization, and channel incision inhibit stream-floodplain connectivity.
- **Channel Substrate & Morphology** refers to the gradient, lateral and longitudinal geometry, and stability and dynamism of the stream channel; and substrate structure. Channel substrate and morphology affect aquatic fauna and flora; and both affect and are affected by floodplain soils and hydrology, floodplain vegetation, stream flow quantity and quality, and stream connectivity.

Floodplain Vegetation refers to the distribution, density, composition, and structure of the floodplain vegetation community. Floodplain vegetation in turn affects aquatic fauna and flora; and both affects and is affected by floodplain soils and hydrology. Floodplain vegetation also affects stream flow quantity & quality: (1) transpiration along the riparian corridor draws water from the stream into the floodplain alluvial aquifer, reducing stream flow; (2) floodplain vegetation creates shade during warmer months while deciduous leaf-loss during cooler months reduces shade, both of which moderate water temperature; (3) tree limbs and trunks that fall into streams contribute to aquatic habitat complexity;

and (4) vegetative litter from the floodplain provides crucial inputs of organic matter to the stream food web.

Figure 8-1. Chihuahuan desert perennial stream systems control model.



- **Floodplain Fauna** refers to the distribution, biomass, composition, food-web interactions, and other impacts of fauna that use riparian habitat for some or all of their life histories, including insects, reptiles, birds, and mammals. Floodplain fauna both affect and are affected by floodplain vegetation and floodplain soils and hydrology.
- **Aquatic Flora** refers to the distribution, biomass, composition, and food-web interactions of stream and benthic flora, including phytoplankton, periphyton, and emergent vegetation. Aquatic flora affect and are affected by stream flow quantity and quality, channel substrate and morphology, and aquatic fauna.
- **Aquatic Fauna** refers to the distribution, biomass, composition, and food-web interactions of stream and benthic fauna, including zooplankton; aquatic macroinvertebrates, including insect larvae; reptiles and amphibians; and fishes, including native endemic species. This system component also addresses other aquatic and semi-aquatic vertebrates such as beaver. Aquatic fauna are affected by floodplain vegetation, the coarse and fine litter from which provides habitat and food; by stream connectivity, which affects the ability of aquatic fauna to move within the flow network; by stream flow quantity and quality; and by channel substrate and morphology. Aquatic fauna also both affect and are affected by aquatic flora; and by floodplain fauna, some of which may prey on aquatic fauna and others of which (e.g., some insects) may spend parts of their life cycles as aquatic fauna.

Environmental components and natural drivers that shape these system components in turn include the following:

- **Aquifer Systems** provide the water for baseflow in Chihuahuan Desert perennial streams in the form of spring discharge and diffuse groundwater seepage. Groundwater-surface water interactions shape stream hydrology, temperature, and chemistry; and are the most crucial process shaping flow persistence along individual stream reaches.
- **Runoff & Erosion** across watershed surfaces deliver not only surface water to stream channels but also sediment, particulate organic matter, and dissolved inorganic and organic matter. Runoff is the most crucial driver shaping extreme high-flow events and overbank flooding of the riparian zone, which together strongly shape channel morphology and floodplain soils and their dynamics. Elevated pulses of stream discharge from runoff also recharge alluvial aquifers.
- **Watershed Topography, Hydrography, Soils, and Cover** mostly affect perennial streams indirectly, through its effects on watershed processes that shape water movement, chemistry, temperature; watershed soil erosion and deposition; and the transport of sediment and organic matter. However, this environmental element also affects perennial streams directly by controlling the geometry of the drainage channel network (watershed hydrography) and its connectivity; and by shaping the potential for upland wildfires to spread into the riparian zone.

The following anthropogenic drivers shape these system components, environmental elements, and natural drivers:

- **Groundwater Withdrawal** alters aquifer system storage and flow gradients in ways that can alter groundwater-surface water interactions along affected stream reaches, thereby altering stream flow quantity and quality.
- **Dams & Diversions** remove surface water from the stream channel, thereby altering stream flow and channel morphology (e.g., wetted area). The construction of stream diversion structures also results in channel modification (see below); and dams and dry reaches created by stream diversions alter stream connectivity.
- **Channel Modification** reshapes channel morphology to better suit human use of perennial streams and their floodplains, for example to stabilize channel geometry at a road crossing or in areas of intensive recreational activity, or to stabilize floodplain agricultural field borders.
- **Invasive Species** alter the composition of the floodplain and aquatic biotic communities. Invasive species can also alter ecological processes such as herbivory and predation on native species, competition for food and habitat among native aquatic fauna, the structure of the aquatic food web, evapotranspiration, stream chemistry, and floodplain soil chemistry and structure. Invasive species also can affect perennial streams indirectly by altering watershed ground cover, soils, and wildfire regimes.
- **Livestock Grazing** can alter floodplain vegetation through herbivory; and can alter both floodplain soils and channel morphology through trampling. Livestock grazing also can affect perennial streams indirectly through its impacts on upland soils and ground cover, thereby affecting watershed processes; and by serving as a vector for the introduction of non-native species into a locality.
- **Fire Regime Modification**, both through wildfire management and through the effects of altered watershed vegetation and climate, alter the frequency, timing, and severity of wildfires across a landscape. Such changes can affect floodplain vegetation, both directly through changes in the riparian wildfire regime and indirectly through the effects of upland wildfire on the spread of invasive species. Fire regime modifications also affect perennial streams indirectly by altering

land surface permeability and soil vulnerability to erosion. Such changes to the land surface affect watershed processes such as infiltration, runoff, and sediment transport and deposition.

- **Land-Use/Development** alters watershed cover, land surface permeability, soil vulnerability to erosion, and releases of chemical pollutants into both watershed soils and water courses, with effects that cascade through the entire hydrologic system. Land use and development also shapes wildfire management policies and actions.

The perennial stream systems control model also recognizes the impacts of climate change and air pollution on the stream-riparian system. These drivers affect stream-riparian ecological condition indirectly, through their effects on weather and atmospheric deposition, and the cascading effects of these changes on upland soils and cover and watershed processes.

8.4 Perennial Stream Systems Stressor Model

Table 8-2 presents and defines the drivers, critical environmental elements, critical ecological processes, and ecological outcomes that characterize the perennial stream systems stressor model. The stressor model follows the methodology described in Chapter 4.

Table 8-2. Chihuahuan desert perennial stream systems stressor model drivers, critical environmental elements, critical ecological processes, and ecological outcomes.

Model Component	Definition
Drivers	
Air Temperature Regime	The pattern of variation in air temperature, including daily, seasonal, annual, and longer-term variation; and the magnitude, frequency, timing, and duration of maxima and minima.
Atmospheric Deposition	The pattern of variation in the deposition of potential pollutants from the atmosphere onto the land and water surfaces of the Chihuahuan Desert ecoregion, including variation in pollutant types, and rates of wet, dry, and total deposition.
Domestic Grazing Management	The pattern of management of the spatial distribution, timing, duration, frequency, and density of domestic livestock grazing.
Fire Management	The pattern of management of the spatial distribution, timing, and frequency of wildfire suppression and prescribed burns. (The fire regime, affected by both natural dynamics and management decisions, is treated as a critical environmental element, below).
Non-Native Species Introductions	The types, origins, and patterns of introduction (where, when, how) of non-native species into the ecoregion. This driver does not include domesticated livestock or species intentionally introduced by fish and game managers for recreational sport.
Precipitation & Snowmelt Regime	The form (rain, ice, snow) and pattern of variation in precipitation, including daily, seasonal, annual, and longer-term variation in magnitude, frequency, timing, and rate (intensity); the annual pattern of variation in the rate and timing of snowmelt; and the chemistry of the precipitation.
Sport & Nuisance Species Management	The pattern of management of sport and nuisance species spatial distributions and densities, including official management by governmental agencies and private management by individuals and non-governmental organizations; and including species rearing, releasing, monitoring, control, and removal; and also including removal of non-native species and habitat restoration to benefit native species.

Model Component	Definition
Water Management & Use	The pattern of management of surface and groundwater storage, movement, and use (where, when, at what magnitudes) by public agencies, private organizations, and private individuals, controlled by structures such as dams, diversions, well fields, conveyances, and levees, including “return flows” from agricultural, industrial, and municipal use. (In other ecoregions, this driver would also address management of water for navigation and/or hydropower generation, neither of which pertains within the U.S. portion of the ecoregion.)
Watershed & Riparian Land Development	The pattern of development of the land surface to support human activities, involving intentional modification of vegetation, soils, or topography and/or construction and maintenance of structures and engineered surfaces; pollutants released by the associated human activities; and riparian habitat restoration.
Critical Environmental Elements	
Channel Reach-Scale Morphology	The overall shape and stability of stream channels at the multi-kilometer scale within a watershed, including the types, abundance, relative confinement, and spatial and temporal distributions of (a) natural features such as confluences/deltas, linear reaches, braiding, bends and meanders, falls and rapids; and (b) artificial features such as dams and other barriers, channel control structures, and levees.
Fire Regime	The pattern of spatial distribution, extent, severity (intensity), timing, and frequency of fire on the landscape, as affected both by natural fuel and ignition dynamics and by management actions including prescribed burns.
Fluvial Network Connectivity	The capacity of a stream network to support the natural downstream transport of matter such as sediment, large woody debris and other plant litter, seeds and other propagules, plankton, and larger aquatic organisms; and the natural upstream and downstream movement of aquatic organisms such as fishes, as determined by the spatial distribution of natural and artificial features that may prevent or inhibit such transport or movement.
Runoff Regime	The pattern of variation in the amount of water flowing from a watershed into a stream, including daily, seasonal, annual, and longer-term variation; and including the frequency, timing, and duration of particular flow rates or stages such as “floods.”
Runoff Water Quality	The chemical properties of the water that runs off a watershed into a stream, including temperature, pH, turbidity, and concentrations of dissolved and suspended constituents swept in off the watershed; and the patterns of variation in these properties, including daily, seasonal, annual, and longer-term variation in their magnitudes.
Water Impoundments & Diversions	The distribution and size of stream reaches that have been converted from natural fluvial conditions into either impounded (lacustrine) conditions or zones of diversion (distributary branches) by dams or other engineered structures, including those constructed by beavers.
Watershed Erosion	The pattern of daily, seasonal, annual, and longer-term variation in the amount (mass and volume) and particle size distribution of sediment eroded off the surface of a watershed and transported into its streams suspended in the watershed runoff.
Watershed Ground Cover	The composition of the surface of a watershed in terms of the abundances and spatial distributions of classes of vegetated, disturbed, and artificial surfaces that differ in their permeability to water infiltration, hydraulic roughness to water runoff, ability to inhibit soil erosion, and provision of shade – the latter of which can affect runoff temperatures and snowmelt). Dissolved and particulate organic matter produced by watershed ground cover and swept by watershed runoff into streams helps support the aquatic food web within the stream.
Watershed-Scale Groundwater Dynamics	The locations and rates of recharge of precipitation to groundwater systems; the storage volumes, inter-connections, and flow path lengths and duration of the aquifers that comprise the groundwater system(s) of a watershed; the geochemical and hydrothermal dynamics of these groundwater systems; and the locations and rates of discharge to the ground surface from these groundwater systems.

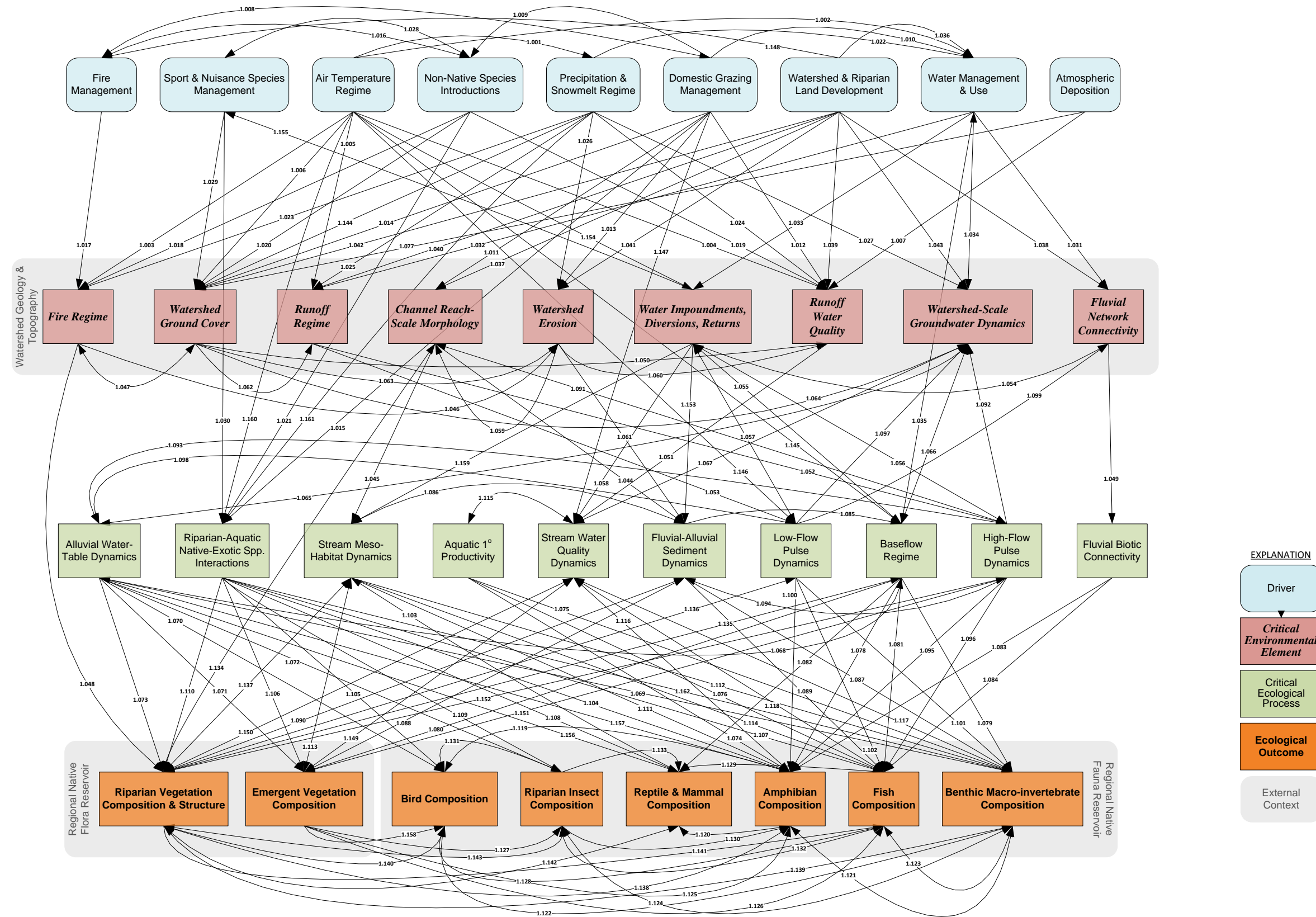
Model Component	Definition
Critical Ecological Processes	
Alluvial Water-Table Dynamics	The pattern of daily, seasonal, annual, and longer-term variation in the elevation of the alluvial water table of riparian reach, including the frequency, timing, and duration of particular water table elevations such as the “X-year” maxima or minima.
Aquatic Primary (1°) Productivity	The pattern of daily, seasonal, annual, and longer-term variation in the rate of primary (<i>aka</i> autochthonous) production of biomass through photosynthesis by organisms such as algae and aquatic and emergent plants. Autochthonous production differs from allochthonous production, the contributions of organic matter from the watershed and riparian corridor to the stream. The stressor model addresses the latter as a component of runoff water quality and the impacts of riparian vegetation on stream conditions.
Baseflow Regime	The pattern of seasonal, annual, and longer-term variation in the stage of the water in a stream when it is fed only by groundwater discharge without any inputs from runoff.
Fluvial Biotic Connectivity	The downstream transport of large woody debris and other plant litter, seeds and other propagules, phytoplankton and zooplankton, and larger aquatic organisms including fish larvae; and the upstream and downstream movement of aquatic organisms such as fishes, as characterized by properties such as the timing, rates, and distances of transport or movement.
Fluvial-Alluvial Sediment Dynamics	The pattern of erosion, deposition, and storage of sediment along a stream and between the stream and its floodplain, as characterized by properties such as the annual and longer-term sediment mass balance of a given stream reach; patterns of channel and floodplain aggradation and degradation; and changes in wetted area.
High-Flow Pulse Dynamics	The pattern of seasonal, annual, and longer-term variation in the stage and rate of discharge of a stream when it is fed by large pulses of runoff, as characterized by properties such as the magnitude, timing, duration, and hydrograph shape of the annual maximum flow; and the magnitude, timing, duration, and hydrograph shape of long-term effective, bankfull, and flood flows (approx. 1.5-year; 2-year; and less frequent flows such as the 10-year, 50-year, and 100-year “7-day” high flows).
Low-Flow Pulse Dynamics	The pattern of seasonal, annual, and longer-term variation in the stage and rate of discharge of a stream during periods drought, as characterized by properties such as the magnitude, timing, and duration of the annual minimum flow; and the magnitude, timing, and duration of long-term 10-year, 50-year, and 100-year “7-day” low flows).
Riparian-Aquatic Native-Exotic Species Interactions	The ways, magnitudes, and spatial and temporal extent to which native and exotic riparian and aquatic species compete for habitat space, food, and other materials; prey on each other; infect or otherwise harm each other; or interact beneficially (mutualism).
Stream Meso-Habitat Dynamics	The frequencies of creation and destruction of meso-habitat features such as bends, side channels, backwaters, islands, bars, pools, eddies, riffles, stranded snags (large woody debris), and bank overhangs in/along the channel; and the spatial extent and persistence of such features and their substrates.
Stream Water Quality Dynamics	The physical and chemical properties of the water in a stream, including temperature, pH, turbidity, and concentrations of different types of dissolved and suspended particulate matter, both inorganic and organic; and the patterns of variation in these properties, including daily, seasonal, annual, and longer-term variation in their magnitudes.
Ecological Outcomes	
Amphibian Composition	The taxonomic, functional, and size composition; spatial and temporal distribution; abundance; health; and activity level of the amphibian assemblages of a stream corridor.
Benthic Invertebrate Composition	The taxonomic, functional, and size composition; abundance; spatial and temporal distribution; and activity level of the aquatic invertebrate assemblage of a stream, including biofilms and periphyton, phytoplankton, zooplankton, insect larvae, crayfish, and mollusks.
Bird Composition	The taxonomic, functional, and size composition; spatial and temporal distribution; abundance; health; and activity level of the avifaunal assemblage of a riparian corridor.

Model Component	Definition
Emergent Vegetation Composition	The taxonomic composition; size range; spatial and temporal distribution; and abundance of emergent (aquatic) vegetation along a stream.
Fish Composition	The taxonomic, functional, and size composition; spatial and temporal distribution; abundance; health; and activity level of the fish assemblages of a stream.
Reptile & Mammal Composition	The taxonomic, functional, and size compositions; spatial and temporal distributions; abundances; health; and activity levels of the assemblages of reptiles and mammals that occupy or visit the riparian zone.
Riparian Insect Composition	The taxonomic, functional, and size composition; abundance; spatial and temporal distribution; and activity level of the insect assemblage of the riparian zone.
Riparian Vegetation Composition & Structure	The taxonomic composition; size range; spatial and temporal distribution; health; vertical above-ground (e.g., canopy) and below-ground (e.g., rooting) structure; and abundance of vegetation along a riparian corridor. This outcome includes the shade and organic matter, including large woody debris, provided by riparian vegetation to the stream.

Figure 8-2 shows the stressor model for the perennial stream systems in the Chihuahuan Desert ecoregion, built using the system model components shown in Table 8-2. It displays all the system model components listed in Table 8-2, along with their causal relationships. Appendix 1 describes and presents the rationale for including every causal relationship in the stressor model, with citations for each causal relationship, providing a comprehensive presentation of the stressor model. Figure 8-2 indicates the presence or absence of causal relationships between the system model components but does not indicate the potential magnitude or other characteristics of these relationships, as explained in Chapter 4.

The stressor model (1) identifies the causal relationships that have affected how the condition of the system likely has changed in the past, in response to changes in its drivers; and (2) provides a means for articulating hypotheses about how the condition of the system will likely change in response to changes in its drivers. The second capability of stressor models is crucial for expressing individual management questions as hypotheses, as discussed in detail below. As discussed and illustrated below, the stressor model makes it clear: (a) which critical environmental elements would likely be affected by a change in one or more particular drivers, including change agents; (b) which critical ecological processes would likely be affected by the cascading effects of these changes in critical environmental elements; and (c) which system characteristics (ecological outcomes) would likely be affected by the cascading effects of these changes in critical environmental elements and ecological processes. By doing so, in turn, the stressor model also highlights those components of the model—drivers, environmental elements, ecological processes, and ecological outcomes—that demand indicator data.

Figure 8-2. Chihuahuan desert perennial stream systems stressor model.



As defined earlier, a causal relationship exists when a change in one component of the system results in a change in some other component. Change in the first component is said to “cause a change in the second component. Each chain of causation, from driver to outcome, describes how the condition of the system likely has changed in the past, or likely would change in the future, in response to changes in its drivers.

Figure 8-2 also includes three gray, background fields. One, labeled “Watershed Geology & Topography,” surrounds the critical environmental elements of the model. This first background field indicates that the critical environmental elements of the system derive from and are shaped by long-term dynamics of watershed geology and topography. However, the stressor model does not attempt to capture these longer-term dynamics. The second and third background fields, labeled “Regional Native Faunal Species Reservoir,” and “Regional Native Plant Species Reservoir,” surround the faunal and floral ecological outcomes of the system, respectively. These two background fields indicate that the ecological outcomes of the system stressor model depend in part on the compositions of the regional reservoirs of native animal and plant species. However, again, the stressor model also does not attempt to capture these larger-scale dynamics.

The following paragraphs discuss how each Change Agent affects Chihuahuan Desert Montane-Headwater Perennial Streams and Lowland-Headwater Perennial Streams, as articulated in the perennial stream systems stressor model. A “sub-model” diagram for each Change Agent presents a simplified version of the master stressor model diagram, showing only the direct and indirect effects of the Change Agent of interest. The text accompanying each sub-model diagram summarizes the information from Appendix 1 concerning the impacts of each Change Agent, and includes selected citations from the more detailed list of citations for each causal link presented in Appendix 1. For each Change Agent, the paragraphs summarize present understanding of: (a) which critical environmental elements would likely be affected—directly or indirectly—by each Change Agent and (b) which critical ecological processes would likely be affected—directly or indirectly—by the cascading effects of these changes in critical environmental elements. However, the ecological characteristics of the system (ecological outcomes) affect each other and are affected by critical environmental elements and ecological processes in the same way regardless of which Change Agent is involved in altering these elements and ecological processes. For this reason, the presentation below begins with a discussion of the interactions between critical ecological processes and ecological outcomes, and among ecological outcomes.

8.4.1 Critical Ecological Processes and Ecological Outcomes

The perennial stream systems stressor model includes eight ecological outcomes: amphibian composition, benthic invertebrate composition, bird composition, emergent vegetation composition, fish composition, reptile & mammal composition, riparian insect composition, and riparian vegetation composition & structure. Table 8-2 above, defines these eight model components. These eight ecological outcomes directly affect each other in numerous ways—and therefore also affect each other indirectly in even more numerous ways—as shown in Figure 8-2 and documented in Appendix 1. For example (see Appendix 1 for full presentation):

- The composition and density of riparian and emergent vegetation affect native benthic invertebrate, amphibian, and fish assemblage composition in the stream by providing habitat such as substrates and cover and/or by providing food to various life stages (e.g., Allan 1995, Propst 1999, Rosen and Caldwell 2004, Bateman et al. 2008a, Wallace and Anderson 2008, USFWS 2009, Kinzli and Myrick 2010, Boeing et al. 2014, Forstner et al. 2014).
- Riparian vegetation composition and structure affect terrestrial riparian insects and vertebrate fauna by providing food and habitat options, which differ depending on the composition and structure of the riparian vegetation (e.g., Bateman et al. 2008a; 2008b; 2008c; 2009, Malcom and Radke 2008, Andersen and Shafroth 2010, Ruth et al. 2010, Wild 2011, Merritt and Bateman 2012, Brand et al. 2013, Minckley et al. 2013, Refsnider et al. 2013, Forstner et al. 2014, Gibson and Olden 2014, Smith and Finch 2014).
- Emergent vegetation provides nesting and feeding habitat for waterfowl. The types and quality of this habitat varies with the composition, spatial and temporal distribution, and abundance of emergent vegetation along streams including in floodplain, thereby affecting the composition of the bird assemblage along streams (e.g., NMDGF 2006, Lougheed and Rodriguez 2008, Malcom and Radke 2008, Ruth et al. 2010, Merritt and Bateman 2012, Brand et al. 2013, Minckley et al. 2013, Smith and Finch 2014).
- Riparian insects provide food options for birds, reptile and mammals, and amphibians along stream corridors – options that differ depending on what insects are available, at what times, and in what abundances. The composition and abundance of the riparian insect assemblage therefore can affect the composition of the bird, reptile and mammal, and amphibian assemblages along the stream corridors (Hunter et al. 1985, Johnson and Haight 1985, Kozma and Mathews 1997, Skagen et al. 1998, Krueper et al. 2003, Hinojosa-Huerta et al. 2004, Rosen and Caldwell 2004, Makings 2005, Skagen et al. 2005, Price et al. 2005, Rosen 2005, Rosen et al. 2005, Stromberg et al. 2005, Brand et al. 2006, NMDGF 2006, Stromberg et al. 2006, Bateman et al. 2008a; 2008b, Levick et al. 2008, Cerasale and Guglielmo 2010, Ruth et al. 2010, Nagler et al. 2011, McCluney and Sabo 2012; 2014, Oring et al. 2013, Flesch 2014, Forstner et al. 2014). Freshwater fish may sometimes also consume riparian insects that fall into the water (e.g., Tyus and Minckley 1988), although this has not been documented in the present ecoregion.
- Amphibians along stream corridors provide food options for birds and reptiles and mammals – options that differ depending on what amphibians are available, at what times, and in what abundances. The composition and abundance of the amphibian assemblage therefore can affect the composition of the bird and reptile and mammal assemblages along the stream corridors (e.g., Mora et al. 2002, Schmitt et al. 2005, White et al. 2006, Bateman et al. 2009; 2013).
- Fish in perennial streams also provide food options for birds and reptiles and mammals – options that differ depending on what fish are available, at what times, and in what abundances. The composition and abundance of the fish assemblage therefore can affect the composition of the bird and reptile and mammal assemblages along the stream corridors (e.g., Schmidly and Ditton 1978, MacRae et al. 2001, Mora et al. 2002, Schmitt et al. 2005, Bateman et al. 2009, Forstner et al. 2014).
- Benthic invertebrates along perennial streams provide food options for some birds – options that again vary depending on what invertebrates are available, at what times, and in what abundances. The composition and abundance of the benthic invertebrate assemblage therefore can affect the composition of the bird assemblages along the stream corridors. Additionally, birds that consume benthic invertebrates can bio-accumulate contaminants (e.g., organochlorines, mercury) that these invertebrates have bio-accumulated through their own diets and exposures. This can result in high body loads of such contaminants in the birds, to such

high levels that this impairs health and reproduction (e.g., MacRae et al. 2001, White et al. 2006).

- Benthic invertebrates along perennial streams also provide food options for some fish and amphibians – options that again vary depending on what invertebrates are available, at what times, and in what abundances. In turn, feeding pressure by fish and amphibians can affect the composition and abundance of the benthic invertebrate assemblage, with these pressures varying depending on what fish and amphibians are present, at what times, and in what abundances. Consequently, the composition and abundance of the benthic invertebrate assemblage can both affect and be affected by the composition of the bird assemblages along the stream corridors (e.g., Allan 1995, Karr and Chu 1999, Stoddard et al. 2005, Witte 2005, Bergeron et al. 2011, Luce et al. 2012, Rolls et al. 2013).
- Riparian insect larvae are important constituents of the benthic invertebrate assemblage. The composition and abundance of the benthic invertebrate assemblage therefore both affects and is affected by the composition of the riparian insect assemblages (e.g., Allan 1995, Karr and Chu 1999, Stoddard et al. 2005, Wallace and Anderson 2008).

The perennial stream systems stressor model includes ten critical ecological processes that directly affect the ecological outcomes discussed above, shown in the following order in Figure 8-2: alluvial water-table dynamics, riparian-aquatic native-exotic species interactions, stream meso-habitat dynamics, aquatic 1^o productivity, stream water quality dynamics, fluvial-alluvial sediment dynamics, low-flow pulse dynamics, baseflow regime, high-flow pulse dynamics, and fluvial biotic connectivity. Table 8-2, above, defines these ten model components. These ten critical ecological processes directly affect – and in some cases are also affected by – the eight ecological outcomes in numerous ways, as shown in Figure 8-2 and documented in Appendix 1. Alterations to these critical ecological processes as a result of changes in drivers and critical environmental elements necessarily lead to altered ecological outcomes. The following paragraphs provide examples of the interactions of the ten critical ecological processes with ecological outcomes in relatively unaltered systems (see Appendix 1 for full presentation):

- Alluvial water-table dynamics directly affect riparian and emergent vegetation dynamics by affecting the depth to the water table, and the vegetation reciprocally affects water table dynamics through evapotranspiration (Scott et al. 2004, Lite and Stromberg 2005, Price et al. 2005, Stromberg et al. 2005, Leenhouts et al., eds. 2006, Stromberg et al. 2006, Baillie et al. 2007, Scott et al. 2008, Katz et al. 2009, Doody et al. 2011, Nagler et al. 2011). Alluvial water-table dynamics directly affect the composition of the bird, riparian insect, reptile and mammal, and amphibian assemblages by affecting soil moisture and humidity levels across the floodplain, which may be crucial factors affecting habitat quality for these species (Bateman et al. 2008b; 2009, Levick et al. 2008, Stromberg et al. 2005, Brand et al. 2006, Stromberg et al. 2006, Lougheed and Rodriguez 2008, Cerasale and Guglielmo 2010, Ruth et al. 2010, McCluney and Sabo 2012; 2014, Oring et al. 2013, Flesch 2014, Mosher and Bateman 2016). Finally, alluvial water-table dynamics directly affect the availability of hyporheic habitat for some benthic invertebrates (Hancock et al. 2005, Boulton et al. 2010, Tockner et al. 2010).
- Riparian-aquatic native-exotic species interactions have pervasive effects across all ecological characteristics of perennial streams, as discussed later in this chapter (see Invasive Species, below).

- Stream meso-habitat dynamics establish the physical habitat template, and the dynamics of that template, for all aquatic and semi-aquatic species – emergent vegetation, amphibians, semi-aquatic mammals, fishes, and benthic invertebrates – and their interactions with each other (e.g., Pease et al. 2006, Stromberg et al. 2006, Dudley and Platania 2007; 2011, Higgins and Strauss 2008, Lougheed and Rodriguez 2008, Propst et al. 2008, Magaña 2009, Hoagstrom et al. 2010, Kinzli and Myrick 2010, Theobald et al. 2010, USFWS 2010, Dean and Schmidt 2011, Nagler et al. 2011, Karatayev et al. 2012, Connally, ed. 2012a, Haase et al. 2012, Heard et al. 2012, Booth et al. 2013, Gido et al. 2013, Jones and Woods, eds. 2013, Garrett and Edwards 2014, Sandoval-Solis and McKinney 2014, Worthington et al. 2014).
- Aquatic 1° (primary or autochthonous) productivity strongly affects the abundance of benthic invertebrates, amphibians, and fish that feed on the resulting biomass (e.g., Kupferberg 1997, USFWS 2009, Luce et al. 2012, Turner and Edwards 2012, Magaña 2013, Rolls et al. 2013, Wellard Kelly et al. 2013, Boersma et al. 2014, East 2015, Wood et al. 2016, Propst 2016).
- Stream water quality dynamics directly affect the composition and abundance of both riparian and emergent vegetation by affecting the availability of dissolved nutrients, and salts and potentially harmful chemicals; and the vegetation reciprocally can affect the concentrations of salts and potentially harmful chemicals as well (e.g., Deloach et al. 2000, Mainston and Parr 2002, Seiler et al. 2003, Lite and Stromberg 2005, Shafroth et al. 2005, USEPA 2005, Bhattacharjee et al. 2006, Chipps et al. 2006, Mitsch and Gosselink 2007, Shafroth et al. 2008, Johnston et al. 2009, Rooney and Bayley 2010).
- Stream water quality dynamics directly affect the composition and abundance of the benthic invertebrate, amphibians, and fish assemblages, because all aquatic fauna are sensitive to variation in water temperature, pH, turbidity, salinity, and concentrations of specific chemical constituents such as metals and organochlorines. These properties of the water can affect organism health, development, reproduction, feeding activities, and vulnerabilities to predation; can cause them to depart from or avoid affected stream reaches; or, if they are in fact adapted to extreme conditions of water chemistry, can allow them to safely occupy affected stream reaches at the expense of other species (Cowley and Sublette 1987, Allan 1995, Edwards 1997, Karr and Chu 1999, Propst 1999, MacRae et al. 2001, Edwards et al. 2002, Cowley et al. 2003, Hoagstrom 2003, Calamusso 2005, Stoddard et al. 2005, NMDGF 2006, White et al. 2006, Zymonas and Propst 2007, Gregory and Hatler 2008, Hoagstrom 2009, Theobald et al. 2010, Witte 2005, Connally, ed. 2012a, Heard et al. 2012, Karatayev et al. 2012, Jones and Woods, eds. 2013, Garrett and Edwards 2014, Hubbs 2014, Miyazono 2014). Additionally, benthic invertebrates may bio-accumulate contaminants and pass them up the food chain, as discussed above.
- Stream water quality dynamics also affect the benthic invertebrate, amphibians, and fish assemblages indirectly by affecting aquatic primary productivity and the concentrations of allochthonous organic matter carried into the stream by runoff (Allan 1995, e.g., White et al. 2006, Zymonas and Propst 2007, Hoagstrom 2009, Theobald et al. 2010, Gregory and Hatler 2008, Heard et al. 2012, Karatayev et al. 2012, Jones and Woods, eds. 2013, Garrett and Edwards 2014, Hubbs 2014, Miyazono 2014).
- Fluvial-alluvial sediment dynamics affect the ecological characteristics of perennial streams directly by affecting the overall stability and particle size distributions of habitat substrates, for which different benthic invertebrate and fish species have different preferences and tolerances (Magaña 2009, Herbst and Cooper 2010, Hoagstrom et al. 2010, Kinzli and Myrick 2010, Theobald et al. 2010, Heard et al. 2012, Karatayev et al. 2012, Jones and Woods, eds. 2013). The erosion, transport, and deposition of sediment along a riparian-stream corridor also affect

habitat quality for riparian and emergent vegetation of a stream: Erosive disturbances of submerged and exposed alluvial soils reset succession in the disturbed areas, while rooted vegetation in turn can stabilize alluvial soils so that they resist erosive disturbance (Schmidly and Ditton 1978, Stromberg et al. 2006, Loughheed and Rodriguez 2008, Theobald et al. 2010, Dean and Schmidt 2011, Nagler et al. 2011, Connally, ed. 2012a, Sandoval-Solis and McKinney 2014). Fluvial-alluvial sediment dynamics also affect the ecological characteristics of perennial streams primarily by affecting two other critical ecological processes, baseflow and stream meso-habitat dynamics.

- High-flow and low-flow pulse dynamics affect the ecological characteristics of perennial streams primarily by affecting two other critical ecological processes, alluvial water table dynamics and fluvial-alluvial sediment dynamics. However, high-flow and low-flow pulse dynamics also affect the ecological characteristics of perennial streams directly. Different fish, amphibian, and benthic invertebrate species differ in their need for high-flow pulses as triggers for reproduction or movement, in their ability to cope with the extremes of velocity and turbulence associated with extreme-high-flow pulses, and in their ability to tolerate and recover from extended low-flow pulses (Propst 1999, Schmidt et al. 2003, Stromberg et al. 2005, Dudley and Platania 2007, Zymonas and Propst 2007, Bateman et al. 2008a; 2008b; 2008c, Levick et al. 2008, Propst et al. 2008, Small et al. 2009, Perkin et al. 2010, Turner et al. 2010, Stefferud et al. 2011, Bogan and Boersma 2012, Gido and Propst 2012, Heard et al. 2012, Karatayev et al. 2012, Gido et al. 2013, Jones and Woods, eds. 2013, Rolls et al. 2013, Bogan et al. 2014a; 2014b, Boersma et al. 2014, Hubbs 2014, Jaeger et al. 2014, Miyazono 2014, Sandoval-Solis and McKinney 2014, Propst 2016).
- High-flow pulse dynamics also affect and are affected by the riparian and emergent vegetation of perennial streams. The vegetation affects channel and floodplain hydraulic roughness, which affects high-flow discharge velocity and stage. Conversely, high-flow pulses can remove vegetation patches, disturb succession in others, or establish of new patches. Some riparian plants such as cottonwood and willow also require high-flow pulses to create the conditions necessary for seed germination and seedling establishment (Shafroth and Beauchamp 2006; Hultine et al. 2007; Stromberg et al. 2007, 2012; Katz et al. 2009; Merritt et al. 2010; Doody et al. 2011; Poff et al. 2011; Merritt and Bateman 2012; Moore and Owens 2012).
- Outside of runoff events, the seasonal magnitude of baseflow, in combination with stream morphology, determines the extent of the wetted perimeter and patterns of water depths and velocities in a stream. Baseflow also affects water temperatures, and can also affect water quality when the source of the baseflow lies within deeper aquifers. All these factors affect the availability and suitability of stream habitat for emergent vegetation, benthic invertebrates, amphibians, and fishes; and also affect the attractiveness of a stream reach for visitation or use by reptiles and mammals outside of runoff episodes (Deason 1998, Propst 1999, Schmidt et al. 2003, McFarland et al. 2004, Makings 2005, Stromberg et al. 2005, Witte 2005, Dudley and Platania 2007, Zymonas and Propst 2007, Bateman et al. 2008a; 2008b, Loughheed and Rodriguez 2008, Propst et al. 2008, Katz et al. 2009, Small et al. 2009, Perkin et al. 2010, Turner et al. 2010, Heard et al. 2012, Karatayev et al. 2012, Rogalski and Skelly 2012, Jones and Woods, eds. 2013, Rolls et al. 2013, Forstner et al. 2014, Hubbs 2014, Miyazono 2014, Sandoval-Solis and McKinney 2014).
- Fluvial biotic connectivity affects the taxonomic and genetic composition of the amphibian and, most particularly, the fish assemblage, by affecting the ability of these fauna to escape stream reaches at risk of becoming isolated by interruptions in fluvial network connectivity, and/or their ability to subsequently return to reaches that were emptied or isolated by such interruptions (Cowley et al. 2003, Benda et al. 2004a; 2004b, Meyer et al. 2007, Fullerton et al. 2010, Propst

1999, Pringle 2003, Alò and Turner 2005, Dudley and Platania 2007, Zymonas and Propst 2007, Levick et al. 2008, Hoagstrom et al. 2010, Perkin et al. 2010, Theobald et al. 2010, Turner and List 2007, Heard et al. 2012, Miyazono 2014, Pilger et al. 2015).

One critical environmental element, the fire regime, also directly affects one of the ecological outcomes discussed above: Fire through a riparian corridor can significantly affect the vegetation and heat the stream water sufficiently to cause significant mortality among the stream biota. Uncharacteristic wildfire frequency and intensity in riparian corridors therefore can alter the frequency, intensity, and spatial extent of such mortality events (e.g., Gresswell 1999, Brown et al. 2001, Brunelle and Minckley 2002, Nagler et al. 2011, Luce et al. 2012, Whitney et al. 2015; 2016).

Finally, several of the critical ecological processes affect each other, as described in Appendix 1: Alluvial water table dynamics both affect and are affected by the baseflow regime; fluvial-alluvial sediment dynamics affect both the baseflow regime and stream meso-habitat dynamics; high-flow pulses affect both alluvial water-table dynamics and fluvial-alluvial sediment dynamics; low-flow pulse dynamics also affect alluvial water-table dynamics; and stream water quality dynamics both affect and are affected by aquatic 1° productivity.

8.4.2 Climate Change

The final report for the Chihuahuan Desert REA will include a discussion of current forecasts of the ways in which climate change will affect the ecoregion and its ecological resources. The present chapter presents only the conceptual model of the causal relationships and outcomes for perennial streams that potentially will be affected.

Figure 8-3 presents the stressor model for perennial streams in the U.S. portion of the ecoregion, simplified to show only those causal relationships that potentially will be affected by changes in the air temperature and precipitation and snowmelt regimes. Appendix 1 presents the rationale and citations for every causal link shown in the diagram.

Climate change will affect the perennial streams in the U.S. portion of the ecoregion through its effects on two drivers in the perennial stream systems stressor model: the air temperature regime; and the precipitation and snowmelt regime. Changes to these two drivers may be termed “first-order” impacts of climate change. Changes in these regimes may include changes in annual and seasonal averages, in the timing and magnitude of annual and seasonal extreme temperatures, and in the timing and magnitude of precipitation.

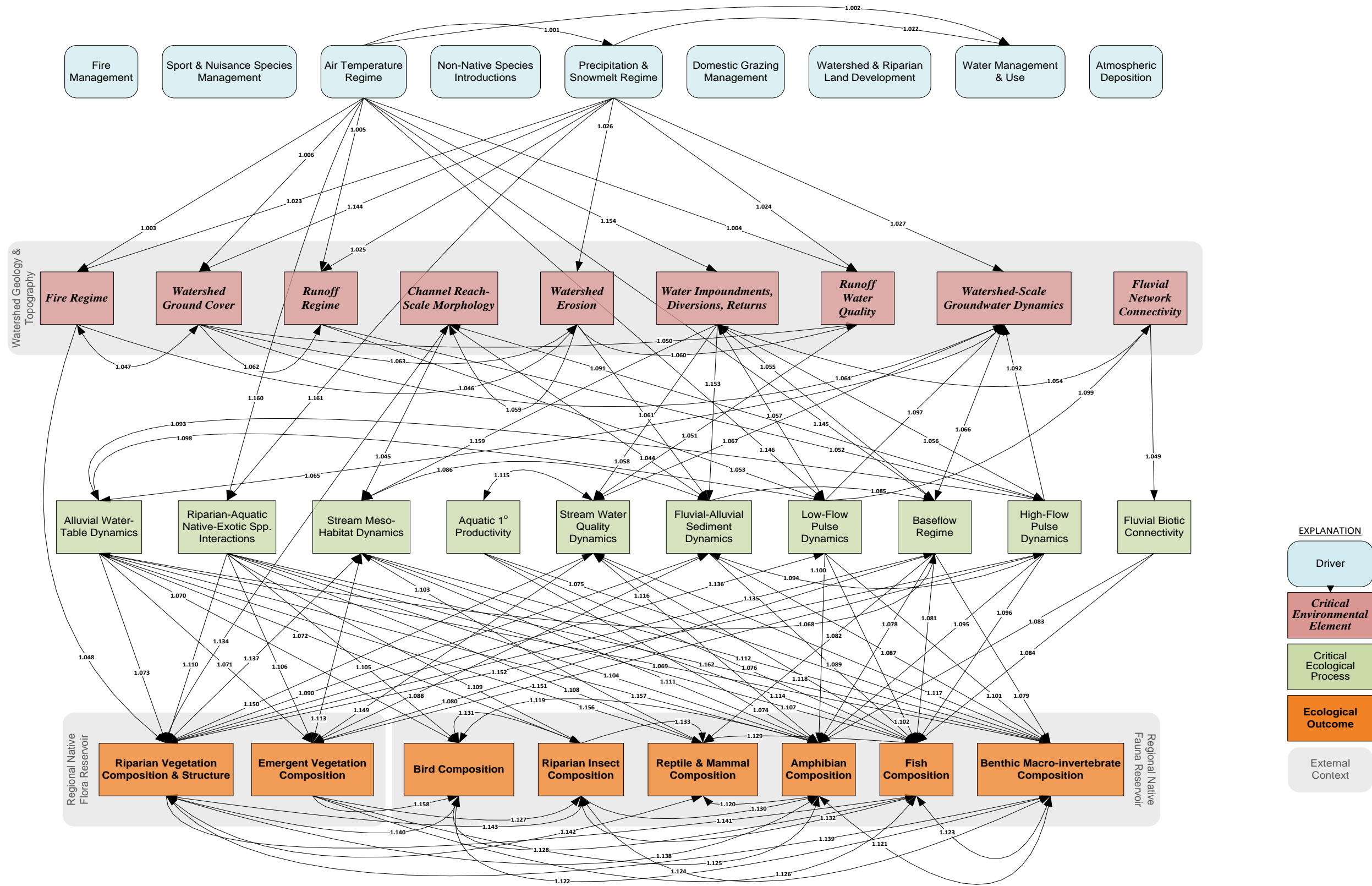
Changes in the air temperature and precipitation and snowmelt regimes in turn will directly affect seven critical environmental elements: the fire regime; watershed ground cover; runoff regime; watershed erosion; water impoundments, diversions, returns; runoff water quality; and watershed-scale groundwater dynamics. The air temperature regime also affects the precipitation and snowmelt regime. Specifically, air temperature affects whether precipitation falls as rain or snow, whether precipitation even reaches the ground or evaporates as it falls (termed “virga” precipitation), and how much water runs off or infiltrates following precipitation versus simply evaporating.

Changes in the air temperature and precipitation and snowmelt regimes also will directly affect another driver, water management and use. Specifically, changes in air temperatures and precipitation will affect annual and seasonal water supply and demand. The resulting changes in water management and use will have their own, further effects on the runoff regime, watershed erosion, and watershed-scale groundwater dynamics; on the operation of dams and diversions; and so forth. These impacts would add to the impacts of climate change on the perennial streams in the Chihuahuan Desert ecoregion. The potential impacts of changes in water management and use are examined further, later in this chapter, in the discussion of the impacts of development.

The impacts of changes in the air temperature and precipitation regimes on the seven critical environmental elements noted above – fire regime; watershed ground cover; runoff regime; watershed erosion; water impoundments, diversions, returns; runoff water quality; and watershed-scale groundwater dynamics – may be termed “second-order” impacts of climate change. Examples of these second-order effects include the following:

- Changes in air temperature and precipitation patterns will affect the probability of wildfires directly along perennial stream riparian corridors (see uncharacteristic wildfire, below) (D'Antonio and Vitousek 1992, Pyne et al. 1996, see Chapters 2-3 and 5-7).
- Changes in air temperature and precipitation patterns will affect watershed ground cover by affecting the types, density, and rates of mortality of upland vegetation across a watershed (see Chapters 2-3 and 5-7).
- Changes in precipitation will affect the runoff regime by altering the timing, amounts, forms, and rates of accumulation of the precipitation on the watershed surface (Dunne and Leopold 1978, Freeze and Cherry 1979, Rango 2006, see Chapters 2-3).
- Changes in the intensity of rainfall events (e.g., the maximum rainfall rate within a storm) will affect the rate and spatial extent of soil erosion caused by individual storm events.
- Changes in air temperature patterns will affect water impoundments by affecting the rate of evaporative losses from reservoir storage (Dunne and Leopold 1978)
- Changes in air temperatures and precipitation patterns will affect runoff water quality by affecting water temperature – which affects other aspects of water quality – and the relative concentrations of soluble matter transported in the runoff. Changes in air temperature and precipitation patterns will also affect the rate at which salts accumulate across soil surfaces as a consequence of natural evaporative processes, and therefore the rate at which such salts are available for dissolution and transport in runoff, further affecting runoff water quality (Manahan 1991, see Chapters 2-3).

Figure 8-3. Chihuahuan desert perennial stream systems stressor model: Potential impacts of climate change.



- Recharge to regional groundwater systems mostly takes place at higher elevations across the mountains and foothills of the ecoregion, and varies both with the amount of precipitation received and whether the precipitation occurs as rain or snow. Melting snow recharges more effectively than does rainfall. Changes in precipitation therefore will affect the spatial distribution and rates of recharge, which will affect watershed-scale groundwater dynamics (e.g., Stonestrom et al., ed. 2007, Wolaver et al. 2008, USBR 2011, Szykiewicz et al. 2012; 2015a; 2015b, Friggens et al. 2013a, Sheng 2013, Friggens and Woodlief 2014, Jaeger et al. 2014, Eng et al. 2016, Meixner et al. 2016).

The second-order effects of changes in the air temperature and precipitation regimes will also include direct impacts to three critical ecological processes:

- Changes in air temperature and precipitation will directly affect riparian-aquatic native-exotic species interactions. Air temperature affects water demand in plants and thermal regulation in land animals, and native species may differ in their abilities to adjust to changes in air temperature patterns compared to non-native species. Similarly, precipitation directly along riparian corridors affects water availability for both plants and land animals along the corridors. Native species may differ in their abilities to adjust to changes in precipitation patterns compared to non-native species (e.g., Price et al. 2005, Enquist et al. 2008, Jones et al. 2010, Nagler et al. 2011, Friggens et al. 2013a; 2013b, Friggens and Woodlief 2014).
- Changes in air temperature will affect baseflow and low-flow pulse dynamics by affecting the rates of evaporation of surface water and evapotranspiration by phreatophytes along riparian corridors. Long-term changes to watershed-scale groundwater dynamics, as a result of changes in recharge (see above), also will affect baseflow and low flows (see below) (e.g., Scott et al. 2004; 2008, Price et al. 2005, Stromberg et al. 2006, Serrat-Capdevila et al. 2007, Hatler et al. 2009, Kennedy and Gungl 2010, Friggens and Woodlief 2014).

These second-order effects, in turn, will have third-order effects on each other, on critical ecological processes, and directly on some ecological outcomes; and these in turn will have effects on other critical ecological processes and ecological outcomes, through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above). For example (see Appendix 1 for full presentation):

- Changes in air temperature and precipitation patterns will affect the magnitude, timing, duration, and hydrograph shape of the runoff events responsible for high-flow pulses (Dunne and Leopold 1978, Gordon et al. 1992).
- Changes in air temperature and precipitation patterns will affect baseflow not only directly through the effects of altered air temperatures (see above), but also indirectly through the effects of altered precipitation on watershed-scale groundwater dynamics. However, the effects on baseflow resulting from altered groundwater dynamics will take decades to centuries to emerge, because of the lengths of the groundwater flow paths that deliver water to some streams, particularly lowland perennial streams. On the other hand, baseflow in higher-elevation headwater streams may depend on discharge from relatively small, shallow aquifers. The effects of changes in regional precipitation patterns on recharge in these settings could take only a few years to affect baseflow (e.g., Heitmuller and Williams 2006, Webb and Leake 2006, Magruder et al. 2009, Kennedy and Gungl 2010, Theobald et al. 2010).

- Through their effects on the runoff regime, runoff water quality, and watershed erosion, changes in watershed ground cover will affect several critical ecological processes. The affected processes will include stream fluvial-alluvial sediment dynamics (e.g., Grant et al. 2003, Schmidt et al. 2003, NMDGF 2006, Hoagstrom et al. 2008, Porter et al. 2009, Theobald et al. 2010, USFWS 2010, Garrett and Edwards 2014, Wohl et al. 2015). The affected processes also will include stream water quality dynamics, including water temperatures and concentrations of dissolved and particulate organic matter (Allan 2004).
- Changes in low-flow pulses may also affect fluvial network connectivity, which in turn determines fluvial biotic connectivity, for streams that become disconnected from a valley or basin stream network during such pulses because some stream reaches run dry (e.g., Propst et al. 2008, Gido et al. 2013, Acreman et al. 2014, Bogan et al. 2014a, Jaeger et al. 2014, Sabo 2014, Fuller et al. 2015, Murphy et al. 2015, see also Chapters 2-3).

8.4.3 Uncharacteristic Wildfire

The fire regime of the U.S. portion of the Chihuahuan desert has changed as a result of the interaction of several drivers. Chapters 2 and 3 discuss the causes and consequences of uncharacteristic wildfire across the ecoregion in general, and Chapters 5-7 discuss the causes and consequences of altered fire regimes specifically across the terrestrial ecological systems of the region.

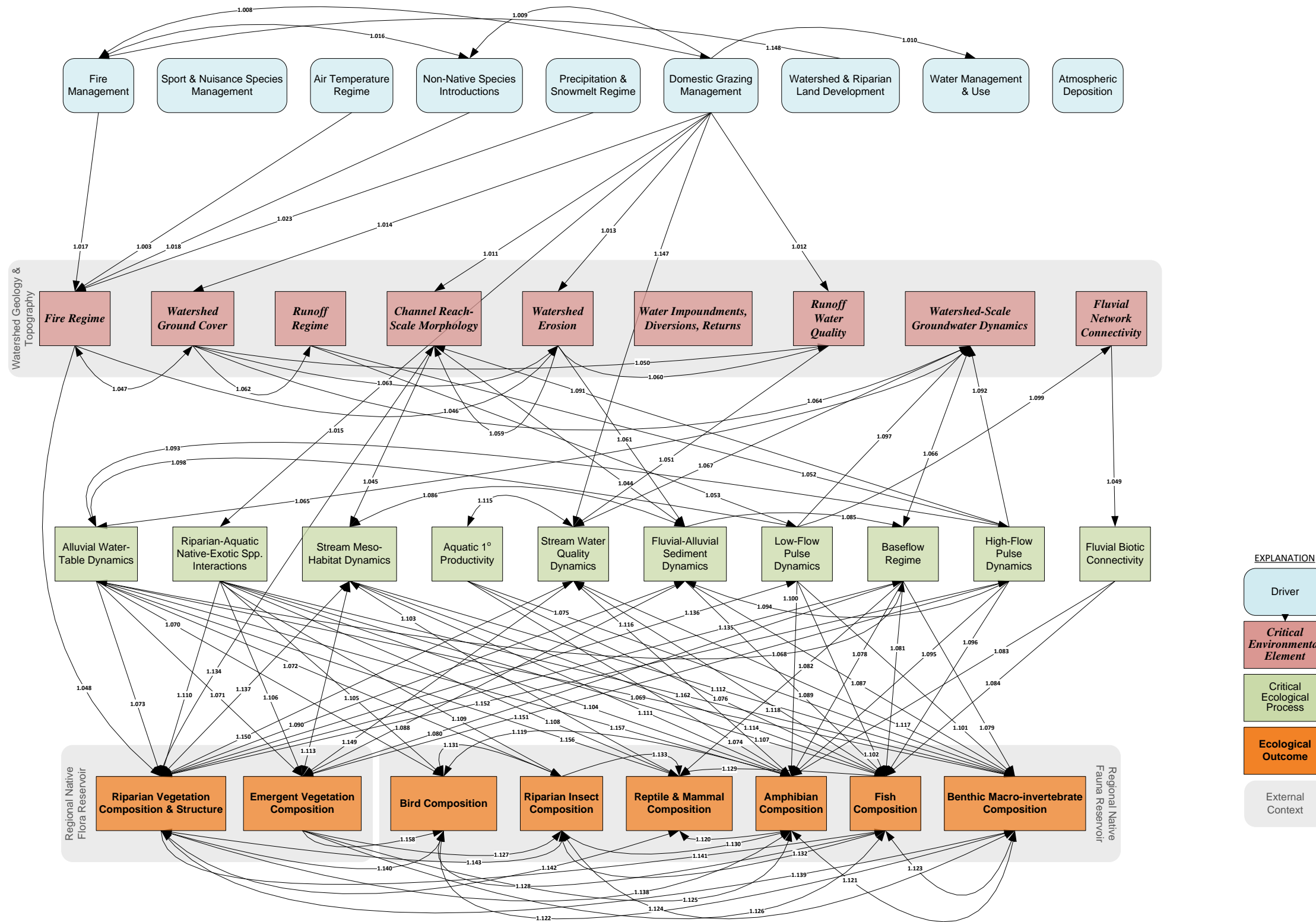
Figure 8-4 presents the stressor model for perennial streams in the U.S. portion of the ecoregion, simplified to show only those causal relationships that potentially will be affected by uncharacteristic wildfire and excessive domestic grazing. Appendix 1 presents the rationale and citations for every causal link shown in the diagram. As shown in Figure 8-4 four drivers directly affect wildfire patterns across the U.S. portion of the ecoregion in ways that, in turn, affect perennial stream systems: fire management, the air temperature regime, non-native species introductions, and the precipitation regime. Three other drivers indirectly affect wildfire patterns across the U.S. portion of the ecoregion. Domestic grazing management affects fire management and non-native species introductions; and watershed and riparian land development affects fire management. Non-native species introductions also affect fire management.

Figure 8-4 shows that watershed ground cover, a critical environmental element, also strongly affects wildfire patterns across the U.S. portion of the ecoregion (see Chapters 2-3 and 5-7). Watershed ground cover in turn is affected by numerous drivers. Chapters 5-7 provide detailed discussions of the drivers that shape watershed ground cover across the U.S. portion of the ecoregion and the ways in which watershed ground cover in turn affects wildfire and vice versa.

Uncharacteristic wildfire affects the perennial streams in the Chihuahuan Desert ecoregion at two scales: (1) indirectly through its effect on larger watershed dynamics that affect streams and their riparian corridors; and (2) through its direct impacts along individual riparian corridors, as shown in the sub-model diagram, Figure 8-4.

Uncharacteristic wildfire at the watershed scale alters (a) watershed ground cover, (b) the vulnerability of upland soils to erosion, and (c) the availability of burned and unburned particulate organic matter and soluble nutrients from ash for downhill transport in runoff (see Chapters 5-7).

Figure 8-4. Chihuahuan desert perennial stream stressor model: Potential impacts of uncharacteristic wildfire and excessive domestic grazing.



These changes in watershed characteristics in turn affect other critical environmental elements in the perennial stream stressor model at the watershed scale, including the runoff regime (which also affects infiltration and recharge), runoff water quality, and the transport of eroded sediment and organic matter downhill toward stream channels, as noted above in the discussion of the potential impacts of climate change. In turn, the alterations to these critical environmental elements affect numerous critical ecological processes. For example (see Appendix 1 for full presentation):

- Alterations to the runoff regime and to watershed-scale groundwater dynamics will result in changes to low-flow pulse dynamics, baseflow, alluvial water-table dynamics, and high-flow pulse dynamics (see discussion of the potential impacts of climate change, above). Further, changes in low-flow pulses may also affect fluvial network connectivity, for streams that become disconnected from a valley or basin stream network during such pulses because some stream reaches run dry (again, see discussion of the potential impacts of climate change, above).
- Changes in runoff water quality will affect stream water quality dynamics, while changes in the rate of delivery of sediment from watershed to stream will stream fluvial-alluvial sediment dynamics, which in turn will affect stream meso-habitat dynamics (again, discussion of the potential impacts of climate change, above).
- Uncharacteristic wildfire at the watershed scale will also affect the frequency and intensity of wildfire directly along riparian corridors (e.g., Gresswell 1999, Brown et al. 2001, Brunelle and Minckley 2002, Stromberg et al. 2009a; 2009b, Theobald et al. 2010, Luce et al. 2012, Whitney et al. 2015; 2016).

These changes in critical ecological processes in the perennial stream systems will in turn affect numerous ecological outcomes through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above).

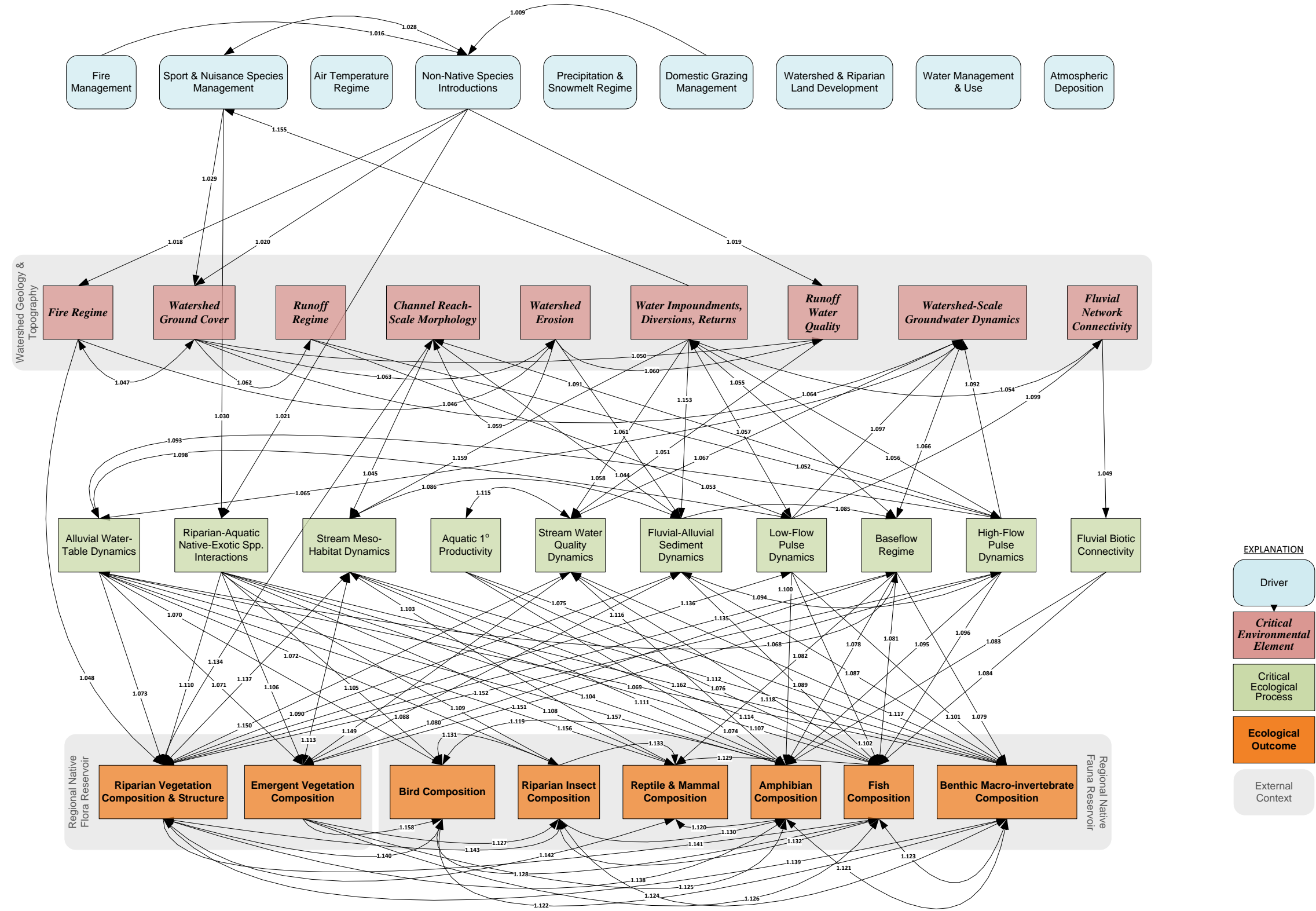
8.4.4 Invasive Species

Chapters 2-3 discuss the ecological consequences of non-native species across the U.S. portion of the ecoregion in general. Figure 8-5 presents the stressor model for perennial streams in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by invasive species. Figure 8-5 includes two drivers that address the ways in which non-native species affect perennial streams in the U.S. portion of the ecoregion: non-native species introductions; and sport and nuisance species management. Appendix 1 presents the rationale and citations for each causal link shown in the diagram.

Together, native species introductions and sport and nuisance species management significantly shape the ecological status of perennial streams across the ecoregion. The two affect each other and in turn are directly or indirectly shaped in part by two other drivers, fire management, and domestic grazing management, the effects of which are discussed separately above and below, respectively.

Water management and use also indirectly affects sport and nuisance species management, as also discussed below (see Development): decisions on impoundment management take into consideration needs for sport and nuisance species management.

Figure 8-5. Chihuahuan desert perennial stream stressor model: Potential impacts of invasive species and landscape restoration.



The native stream and riparian fauna and flora of the ecoregion in turn are affected by their interactions with non-native species and, potentially, by management actions taken to control some non-native species. For example (see Appendix 1 for full presentation):

- Non-native diseases of aquatic fauna, such as introduced fish parasites and the now-widespread chytridiomycosis, harm native fish and amphibians along perennial streams in the ecoregion; and blooms of golden alga in main channels and off-channel ponds (e.g., seasonal ponds on floodplains) may poison native amphibians and native fish (Propst 1999, Rosen and Caldwell 2004, Witte 2005, Zymonas and Propst 2007, Propst et al. 2009, USFWS 2009; 2010, Bean et al. 2010, Israël et al. 2014).
- Non-native aquatic fauna, including non-native sport and nuisance fishes, the non-native American bullfrog, non-native crayfish, and Asian clam (*Corbicula fluminea*) may compete with native aquatic fauna along perennial streams for habitat space and materials, including food; prey on native aquatic fauna; or interbreed with them, destroying their genetic integrity. Stocking of sport fishes, including native species, also alters food web dynamics (Hubbs et al. 1977, Cowley and Sublette 1987, Rinne and Minckley 1991, Rosen et al. 1994; 2005, Fuller et al. 1999, Gido and Propst 1999;2012, Propst 1999, Edwards et al. 2002; 2003, Propst and Gido 2004, Rosen and Caldwell 2004, Kapuscinski and Patronski 2005, Clarkson et al. 2005, Turner and List 2007, Gregory and Hatler 2008, Propst et al. 2008, Paroz et al. 2009, Hoagstrom et al. 2010, Pilger et al. 2010, Turner et al. 2010, Stefferud et al. 2011, Franssen et al. 2015, Heard et al. 2012, Martinez 2012, Gido et al. 2013, Moody and Sabo 2013, Rolls et al. 2013, USFWS 2011; 2013, Hershler et al. 2014, McCluney and Sabo 2014, Miyazono 2014, Whitney et al. 2014, Hedden et al. 2016).
- The introduction and spread of non-native dry-land and riparian vegetation across a watershed can alter the fire regime of the watershed and the riparian corridors of perennial streams within the watershed, as noted above and discussed in detail in Chapters 5-7. Non-native vegetation across watershed can alter runoff water quality by altering soil salt buildup and the types of organic matter swept into the runoff, thereby altering stream water quality. And native and non-native vegetation across the watersheds also can have different abilities to anchor soils (see Chapters 5-7), altering watershed soil erosion dynamics, sediment inputs to perennial streams, and runoff water quality.
- Vegetation across watersheds contributes dissolved and particulate organic matter to runoff. Alterations to watershed vegetation therefore may alter the amounts, decomposability, and chemistry of plant litter available to contribute to the allochthonous inputs to streams carried in watershed runoff (e.g., Vannote et al. 1980, Allan 1995; 2004, Hauer et al. 2002, Babler et al. 2011, Kominoski and Rosemond 2012).
- Non-native aquatic and riparian plant species may interact directly with native aquatic and riparian species, for example by outcompeting native species for space and water, as do Russian olive (*Elaeagnus angustifolia*) and tamarisk (aka salt cedar) (*Tamarisk* spp.) giant reed (*Arundo donax*), and hydrilla (*Hydrilla verticillate*). (DeBano et al., eds. 1995, Bell et al. 1999, Krueper et al. 2003, Scott et al. 2004, Lite and Stromberg 2005, Makings 2005, Price et al. 2005, Rosen 2005, Bateman et al. 2008a; 2008b, Cornell et al. 2008, Levick et al. 2008, Stromberg et al. 2009a; 2009b, Dean and Schmidt 2011, Doody et al. 2011, Nagler et al. 2011; 2012, Fain et al. 2014).
- The non-native nutria (*Myocastor coypus*), a semi-aquatic rodent, alters aquatic and shoreline habitat and vegetation, affecting habitat and food resources for native fauna and affecting both native and non-native emergent vegetation (Milholland et al. 2010).

These changes in ecological outcomes in the perennial stream systems will in turn affect numerous other ecological outcomes through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above).

8.4.5 Development

Chapters 2-3 discuss the ecological consequences of land and water development across the U.S. portion of the ecoregion in general. Figure 8-6 presents the stressor model for perennial streams in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected two drivers that address the impacts of development on this CE: (1) watershed and riparian land development, and (2) water management and use. Figure 8-6 shows the causal relationships through which these two drivers directly or indirectly affect every critical environmental element represented in the perennial stream systems stressor model, changes in which will affect almost every critical ecological process and ecological outcome. Appendix 1 presents the rationale and citations for each causal link shown in the diagram.

Changes in watershed and riparian land development will directly affect seven critical environmental elements: watershed ground cover; runoff regime; channel reach-scale morphology; watershed erosion; runoff water quality; watershed-scale groundwater dynamics; and fluvial network connectivity (see Appendix 1 and Chapters 2-3 and 5-7). Similarly, changes in water management and use will directly affect four critical environmental elements in the perennial stream systems stressor model: runoff regime; water impoundments, diversions, returns; watershed groundwater dynamics; and fluvial network connectivity (see Appendix 1 and Chapters 2-3). The impacts of water management and use thus compound the effects of watershed and riparian land development on three critical environmental elements: runoff regime; watershed groundwater dynamics; and fluvial network connectivity. In fact, watershed and riparian land development necessarily affects water management and use itself. Many if not most forms of land development in the ecoregion require a water supply to sustain the activities associated with the development. Watershed and riparian land development also affects another driver, fire management. Fire management decisions also must take into account the types and locations of developed land, as these evolve across the landscape (see Uncharacteristic Wildfire, above). Water management and use, in turn, indirectly affects the driver, sport and nuisance species management by affecting the availability of water in stream reaches and in impoundments suitable for sport and nuisance species.

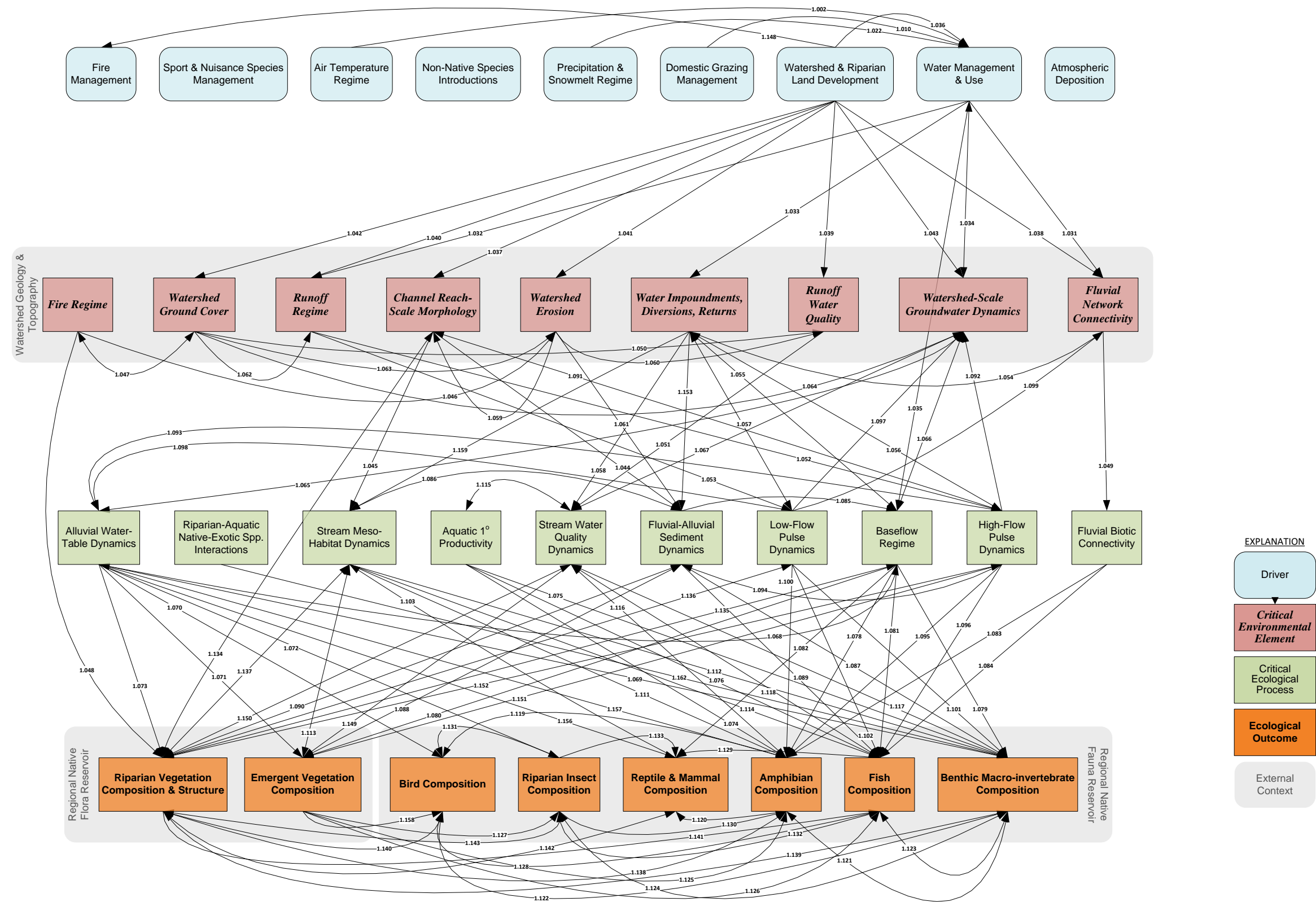
The impacts of changes in watershed and riparian land development and in water management and use on the eight critical environmental elements noted above – watershed ground cover; runoff regime; channel reach-scale morphology; watershed erosion; water impoundments, diversions, returns; watershed groundwater dynamics; runoff water quality; watershed-scale groundwater dynamics; and fluvial network connectivity – may be termed “first-order” impacts of these two drivers. Examples of these first-order effects include the following (see Appendix 1 for full presentation):

- Land development across watersheds necessarily affects watershed hydrologic function by altering watershed ground cover, erosion, runoff rates and water quality, and groundwater recharge. For example, expansion of impermeable land surface across a watershed increases

runoff rates and reduces recharge, other things being equal. Developed lands generate greater point-source and diffuse surface water pollution, unless the development includes effective runoff pollution control systems (Dunne and Leopold 1978, Gordon et al. 1992, Pepper et al. 1996, Levings et al. 1998, Belsky et al. 1999, Stonestrom et al., ed. 2007, Gregory and Hatler 2008, Levick et al. 2008, Theobald et al. 2010, NMOSE 2013, see Chapters 2-3 and 5-7).

- Water management in the ecoregion includes the interception of runoff by water-control features designed to support flood control or local water use (e.g., in stock ponds or upland reservoirs). Such features necessarily reduce watershed runoff volume and/or affect the timing and magnitude of movement of runoff from a watershed into its stream channels; and also trap sediment that would otherwise be transported further downstream to contribute to stream fluvial-alluvial sediment dynamics.

Figure 8-6. Chihuahuan desert perennial stream stressor model: Impacts of land and water development.



- Land development across watersheds also may involve the construction of road and railroad crossings over streams or the burying of stream courses in underground culverts that reduce fluvial network connectivity (Diebel et al. 2015, Fuller et al. 2015).
- Riparian land development may artificially shape and stabilize channel reach-scale morphology by: confining stream channels within artificial channel control structures; artificially “hardening” channel banks to prevent flows from reshaping their morphology; incorporating levees that eliminate natural flood dynamics crucial to riparian ecology; and even eliminating riparian habitat altogether (Theobald et al. 2010, Poff et al. 2011).
- Water management and use in the ecoregion have long involved the construction and operation of dams, diversions, and return-flow systems, as discussed in Chapter 2. The natural precipitation regime of the ecoregion, as also discussed in Chapter 2, produces a large pulse of runoff in the winter and early spring, and a smaller, more variable pulse in the late summer. The dams in the ecoregion store this water for use on the different time-table of human activities, including crop irrigation and urban and industrial consumption. Much of the water diverted for human use is also consumed, returning only a modest fraction back to the surface water system. The dams also alter fluvial network connectivity (Dunne and Leopold 1978, NMOSE 2013, see also Chapters 2-3).
- The storage and release of water from dams in the ecoregion are highly regulated, including by international agreements between the U.S. and Mexico. However, this latter topic applies primarily to the large river-floodplain systems of the ecoregion and therefore is discussed in Chapter 9.
- Water management and use in the ecoregion also have long involved the construction and operation of groundwater wells. Local, state, and bi-national water management policies and practices across the ecoregion determine how much water is consumed from the aquifers of the ecoregion and where and when it is withdrawn (e.g., George et al. 2011, NMOSE 2013, see also Chapters 2-3).
- Stream diversions can directly alter fluvial network connectivity by depleting stream flows sufficiently to create seasonal or permanent dry reaches (Fuller et al. 2015).
- Point-source and diffuse pollution affects stream water quality dynamics by introducing or increasing the concentrations of anthropogenic compounds and altering stream temperature. Water pollutants potentially can include industrial chemicals including brine wastes, agricultural chemicals including nutrients and biocides, and household chemicals including pharmaceutical compounds. Many pollutants and their effects can persist for some distance downstream of their places of inflow (Dunne and Leopold 1978, Pepper et al. 1996, Levings et al. 1998, Belsky et al. 1999, Gregory and Hatler 2008, Levick et al. 2008, NMOSE 2013, see Chapters 2-3).

These first-order effects of development, in turn, will have second-order effects on each other, on critical ecological processes, and directly on some ecological outcomes; and these in turn will have effects on other critical ecological processes and ecological outcomes. For example:

- Dams fragment the fluvial network. The impacts of individual dams on fluvial network connectivity depend on the location, design, and operation of each dam. Design factors that affect the impacts of individual dams on fluvial network connectivity include dam height, overflow rates and frequencies, the presence/absence and design of hydropower turbines or diversion structures, and the presence/absence and design of accommodations for fish passage (Pringle 2003, Fuller et al. 2015, see also Chapters 2-3).

- Changes to watershed runoff regimes alter stream hydrology, particularly annual and seasonal discharge and all aspects of high-flow and low-flow dynamics, as discussed earlier (see Climate Change).
- The operations of dams, diversions, and return-flow systems also together significantly alter annual and seasonal discharge and all aspects of high-flow and low-flow dynamics downstream. For example, dams store and release water to accomplish specific water management objectives, which may include flood control, water supply, and/or hydropower generation. The rates and timing of releases therefore may differ greatly from the natural flow regime for a given stream. Pulses of spring rain/snowmelt may be stored to supply water demands during the hot summer months, thereby eliminating the natural spring high-flow pulse of a stream. Similarly, impoundment of summer storm runoff may exaggerate the natural low-flow pattern in a dammed stream. And impoundment releases to meet varying downstream water demands may replace the naturally stable baseflow of a stream with a more variable artificial baseflow (Williams and Wolman 1984, Collier et al. 1996, Graf 1999; 2006, Smakhtin 2001, Poff and Hart 2002, Richter and Thomas 2007, Fitzhugh and Vogel 2011).
- Dam operations directly affect stream water quality in several ways. Unless extremely shallow, stream impoundments typically become thermally stratified, with the colder bottom layer (hypolimnion) also often becomes anoxic, with further consequences for lake chemistry. Dams may release impounded water from the warmer epilimnion (upper layer), the hypolimnion, or some combination of the two. The released water therefore typically is thermally and chemically highly altered from natural conditions (Baxter 1977, Miyamoto et al. 2007).

These changes in critical ecological processes for the perennial stream systems will in turn affect numerous ecological outcomes through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above).

8.4.6 Excessive Domestic Grazing

Chapters 2-3 discuss the ecological consequences of excessive domestic grazing across the ecoregion in general, and Chapters 5-7 discuss its consequences specifically across the terrestrial ecological systems of the larger landscape. Figure 8-4 presents the stressor model for perennial streams in the U.S. portion of the ecoregion, simplified to show only those causal relationships that potentially will be affected by uncharacteristic wildfire and excessive domestic grazing. Appendix 1 presents the rationale and citations for every causal link shown in the diagram. As shown in Figure 8-4 domestic grazing management in the U.S. portion of the ecoregion affects water and fire management practices. Ranchers across the ecoregion provide water for their livestock by damming intermittent streams to form small ponds, diverting perennial streams to watering structures, and drilling small wells to supply watering tanks. As also shown in Figure 8-4, domestic grazing management in the U.S. portion of the ecoregion both affects and is affected by the spread of non-native vegetation, as discussed in detail in Chapter 5-7. For example, grazed livestock act as vectors for spreading non-native grasses.

In addition to its impacts on water management and use, fire management, and the spread of non-native vegetation, excessive domestic grazing affects the perennial streams in the U.S. portion of the ecoregion at two geographic scales: (1) indirectly through its effect on larger watershed dynamics that affect streams and their riparian corridors; and (2) directly through the immediate impacts of excessive grazing along individual riparian corridors. For example (see Appendix 1 for full presentation):

- Excessive domestic grazing alters (a) watershed ground cover, (b) the vulnerability of upland soils to erosion, and (c) runoff water quality – the latter effect arising from animal liquid and solid wastes. These changes in watershed characteristics in turn affect other critical environmental elements in the perennial stream stressor model at the watershed scale, including the runoff regime (which also affects infiltration and recharge), runoff water quality, and the transport of eroded sediment downhill toward stream channels. In combination, these changes in watershed dynamics affect numerous critical ecological processes and their ecological outcomes in the perennial stream systems (Wohl 2006, Theobald et al. 2010, Poff et al. 2011, see Chapters 2-3 and 5-7).
- Livestock may feed on native and non-native riparian vegetation; rest under riparian canopy cover; trample native streambank biota and burrows while moving in/out of stream waters; and compete with native herbivores. Excessive trampling by livestock can destabilize stream banks and channel substrates, thereby directly altering channel reach morphology and consequently stream meso-habitat dynamics. Livestock that graze, rest, and move in and along streams deposit their wastes directly alongside and into streams, directly altering stream water quality (Schmidly and Ditton 1978, Medina and Martin 1988, El-Hage and Moulton 1998, Belsky et al. 1999, Propst 1999, BLM 2000, Krueper et al. 2003, Gordon and Meentemeyer 2006, Wohl 2006, Levick et al. 2008, Lucas et al. 2009, Herbst and Cooper 2010, Theobald et al. 2010, Poff et al. 2011, USFWS 2012, Garrett and Edwards 2014, Cole and Cole 2015, see Chapters 2-3).

These changes in critical environmental elements and critical ecological processes for the perennial stream systems in turn affect numerous ecological outcomes through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above).

8.4.7 Landscape Restoration

Chapters 2 and 3 discuss the types of landscape restoration projects taking place in the U.S. portion of the ecoregion, some of which currently or in the future could affect perennial stream systems. Ongoing and potential landscape (i.e., habitat) restoration efforts along perennial streams and their riparian corridors include efforts to remove non-native aquatic and riparian species, and to ensure the replacement of removed non-native vegetation with native vegetation. Specifically such efforts include or potentially could include the following (see Figure 8-5 and Appendix 1):

- Restoration of stream habitat through the control or removal of non-native fishes, removal or improvement of road-crossing barriers, exclusion of livestock, and re-introduction of beaver and locally extirpated fish (e.g., Kapuscinski and Patronski 2005, NMDGF 2006, Dudley and Platania 2007, Hoagstrom et al. 2008, USFWS 2009; 2010; 2012, Wild 2011, Januchowski-Hartley et al. 2013; 2014, BEEC 2014, Gibson and Olden 2014, Hershler et al. 2014, Pilger et al. 2015). (Captive breeding and re-introductions of native species is not considered a type of landscape or habitat restoration, and so is not included here).
- Restoration of riparian and wetland habitat along perennial streams through flow restoration, exclusion of livestock, and/or removal of non-native vegetation, with or without active restoration of native vegetation in its place (Farley et al. 1994, Stromberg 1998; 2001, Belsky et al. 1999, BLM 2000, Krueper et al. 2003, Bateman et al. 2008a; 2008b; 2008c; 2009, Malcom and Radke 2008, Shafroth et al. 2008, Abelho and Molles 2009, Katz et al. 2009, Smith et al. 2009, Stromberg et al. 2009a; 2009b, Theobald et al. 2010, Nagler et al. 2011, Poff et al. 2011, Watts and Moore 2011, Brand et al. 2013, Reynolds et al. 2014, Cole and Cole 2015, Mosher and

Bateman 2016). Non-native species targeted for control or removal include salt cedar (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*), *Arundo donax*, an invasive reed, and Phragmites (*Phragmites* spp.), another invasive reed. Removal methods may include biological control, prescribed fire, mechanical removal, and chemical control, and these methods can have their own effects on riparian biota (e.g., Nagler et al. 2012, Goolsby et al. 2016).

8.5 Perennial Stream Systems Key Ecological Attributes

All ecological outcomes and critical ecological processes in a system stressor model constitute key ecological attributes for the system. The list below identifies 18 key ecological attributes for Chihuahuan Desert perennial stream systems based on these criteria. *Characterizing the present condition of a system requires data on indicators for its key ecological attributes.* The definitions for the key ecological attributes are the same as the definitions for these model components presented above.

- **Ecological Outcomes**
 - Amphibian Composition
 - Benthic Macroinvertebrate Composition
 - Bird Composition
 - Emergent Vegetation Composition
 - Fish Composition
 - Reptile & Mammal Composition
 - Riparian Insect Composition
 - Riparian Vegetation Composition & Structure
- **Critical Ecological Processes**
 - Alluvial Water-Table Dynamics
 - Aquatic Primary (1^o) Productivity
 - Baseflow Regime
 - Fluvial Biotic Connectivity
 - Fluvial-Alluvial Sediment Dynamics
 - High-Flow Pulse Dynamics
 - Low-Flow Pulse Dynamics
 - Riparian-Aquatic Native-Exotic Spp. Interactions
 - Stream Meso-Habitat Dynamics
 - Stream Water Quality Dynamics

9 Large River-Floodplain Systems Conceptual Model

This chapter presents the conceptual ecological model for the Large River-Floodplain Systems CE. The presentation follows the structure described in Chapter 4.

As explained in Chapter 3, the Large River-Floodplain Systems Conservation Element (CE) includes only three rivers, the Gila River, Pecos River, and Rio Grande. These three rivers differ in several ways from the two perennial stream system CE types discussed in Chapter 8. First, most of the annual discharges of water and transported matter in these three rivers originate outside the ecoregion, as discussed in Chapter 2—the Gila in the Mogollon Mountains to the north and northwest of the ecoregion; the Pecos in the Sangre de Cristo range of the southern Rocky Mountains to the north; and the Rio Grande both in the southern Rocky Mountains to the north and in the mountains of the Rio Conchos basin to the southwest, in Mexico. The seasonal patterns of discharge of the three large rivers, shaped by weather conditions outside the ecoregion, also differ from the seasonal patterns of discharge of the perennial streams that originate entirely within the U.S. portion of the ecoregion. In addition, large dams and diversions within the U.S. portion of the ecoregion fragment, regulate, and deplete the Rio Grande and Pecos River within the ecoregion, as discussed in Chapter 2.

Prior to their regulation, the flows of water and sediment along these two rivers within the U.S. portion of the ecoregion maintained more complex channels, much larger, frequently inundated floodplains with extensive wetlands, and much larger alluvial aquifers than associated with any montane- or lowland-headwater stream in the ecoregion. Their longer flow distances from their headwaters also resulted in higher water temperatures and higher concentrations of dissolved matter, both conditions exacerbated by river regulation. Natural short-term disturbances include riparian fire, which may originate in the surrounding uplands, droughts, and both inundation and sediment erosion and deposition by floods. As a result, these three large rivers support – or once supported – aquatic and riparian fauna and flora adapted to large, warm-water river settings, active river-floodplain exchanges of water and nutrients, flood cycles and disturbances, and more extensive riparian wetland and woodland communities. The section in Chapter 8 on “Fishes of the U.S. Portion of the Chihuahuan Desert Ecoregion” identifies the fish native to the Gila and Pecos Rivers and to the Rio Grande and their tributaries. The floodplain wetlands of the Rio Grande and Pecos River also continue to provide stopover or over-wintering habitat for numerous migratory bird species, some in very large numbers.

The waters of the Rio Grande and Pecos River within the U.S. portion of the ecoregion, and their arable alluvial soils, have long attracted human settlement and development. The Rio Grande has experienced considerable watershed and floodplain development upstream outside the U.S. portion of the ecoregion in both the U.S. and Mexico, as well. The U.S. and Mexico share use of the Rio Grande.

9.1 Sources of Information

The large river-floodplain systems control and stressor models integrate information from numerous sources:

- (1) The Stream and Riparian control model presented in Miller et al. (2010).
- (2) The conceptual models for riparian-stream conservation elements developed for the Madrean Archipelago rapid ecoregional assessment immediately to the west of the Chihuahuan Desert ecoregion (Crist et al. 2014).
- (3) Type descriptions (NatureServe 2014) for the terrestrial ecological system types recognized for the riparian corridors in the ecoregion. The terrestrial ecological system types associated with the riparian corridors along the large rivers of the ecoregion include the North American Warm Desert Riparian Woodland and Shrubland (International Ecological Classification Code CES302.748) North American Warm Desert Riparian Mesquite Bosque (CES302.752), North American Warm Desert Riparian Woodland and Shrubland (CES302.753), Western Great Plains Riparian (mixed upland and wetland) (CES303.956), and North American Arid West Emergent Marsh (CES300.729) (Dinerstein et al. 2001, NMDGF 2006, Connally, ed. 2012a; 2012b, NatureServe 2014). The term, “terrestrial ecological systems” here refers to “... recurring groups of [terrestrial, including wetland] biological communities that are found in similar physical environments and are influenced by similar dynamic ecological processes, such as fire or flooding” (Comer et al. 2003).
- (4) A large literature identifying the distinctive assemblages of fish and amphibians in the Gila River, Pecos River, and Rio Grande, and the natural and anthropogenic factors shaping the composition of these assemblages. Table 8-1 in Chapter 8 lists the fish species of the ecoregion and their associations with Gila River, Pecos River, and Rio Grande, along with their associations with perennial streams and springs. The Gila River basin, part of the Colorado River basin, lies west of the Continental Divide and its fish assemblage therefore differs significantly from that of the Rio Grande-Pecos River basin east of the Divide. Table 3-2 in Chapter 3 lists the amphibians recognized within the U.S. portion of the ecoregion, with notes on their habitat associations and state-level conservation status in New Mexico and Texas.
- (5) A large literature on the hydrology, history of hydrologic and geomorphic alteration, and potential for eco-hydrologic restoration along the Gila River, Pecos River, and Rio Grande (e.g., Winemiller and Anderson 1997, Fullerton and Batts 2003, Schmidt et al. 2003, Bhattacharjee et al. 2006, NMDGF 2006, Rango 2006, Dudley and Platania 2007, Gregory and Hatler 2008, Lougheed and Rodriguez 2008, Hoagstrom 2009, Magaña 2009, Small et al. 2009, Kinzli and Myrick 2010, Dean and Schmidt 2011, USBR 2012);
- (6) A large literature on the ecology of large alluvial rivers in general (e.g., Schumm 1985, Bayley 1995, Sparks 1995, Kondolf 1997, Tockner et al. 2000, Trush et al. 2000, Tockner and Stanford 2002, Ward et al. 2002, Church 2006, Sheldon and Thoms 2006, Thoms 2006, Moyle and Mount 2007, Shafroth et al. 2010, Opperman 2012).
- (7) The conceptual ecological model developed for the perennial stream CEs for the ecoregion, presented in Chapter 8. Although ecological distinct, the three large rivers share many aquatic and riparian species and ecological characteristics in common with their perennial tributaries, share the same watersheds within the U.S. portion of the ecoregion, and are significantly shaped by the discharges and ecological conditions of their tributaries within the U.S. portion of the ecoregion.

9.2 Large River-Floodplain System Control Model

Figure 9-1 shows the control model for the Chihuahuan desert large river-floodplain system. The control model again shows drivers and system components in greater detail than the overarching Chihuahuan desert wet system conceptual model. As in the control models for the Chihuahuan desert perennial

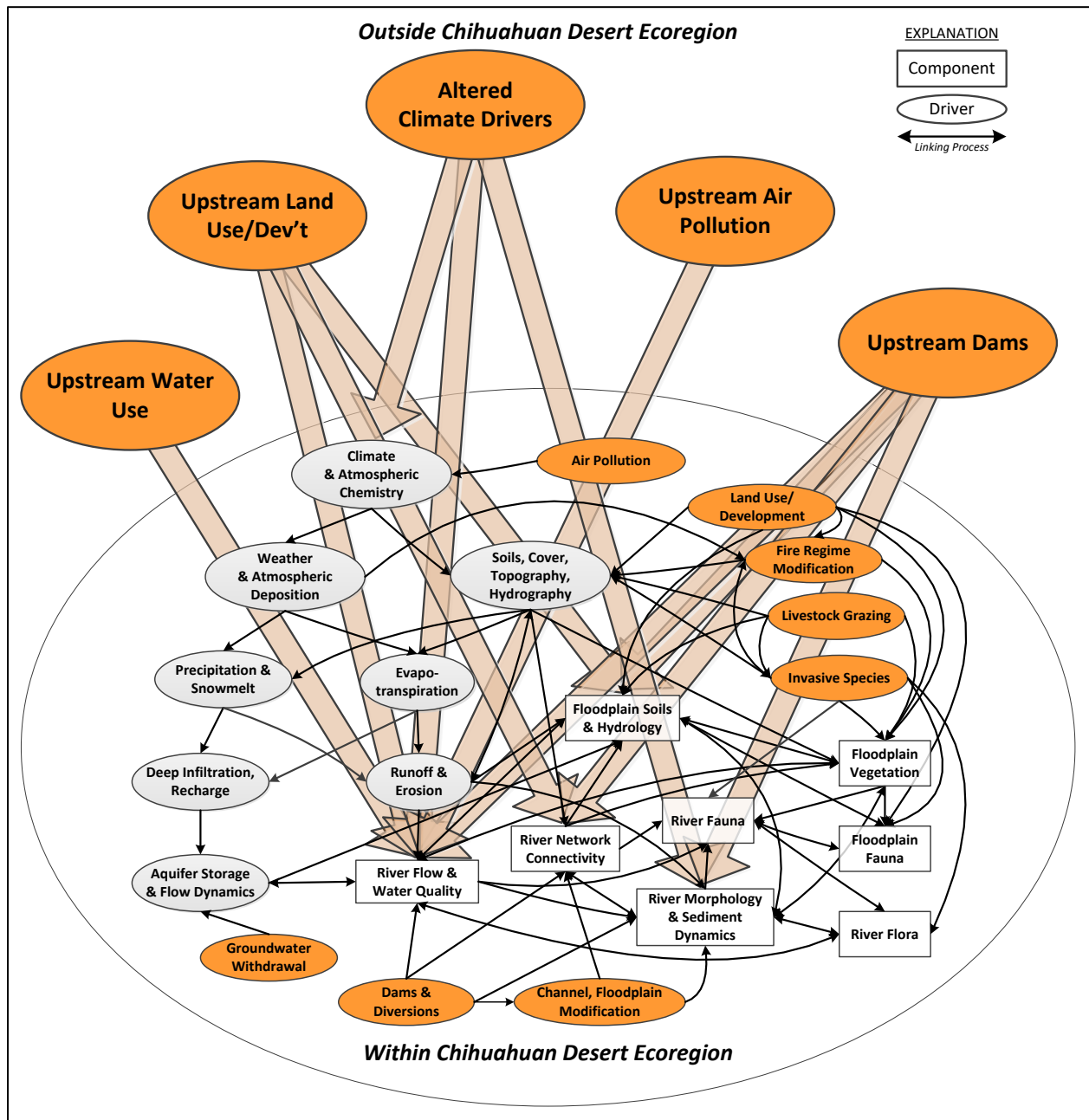
stream systems, Chihuahuan desert large-river system control model components consist of pivotal physical, biological, and ecological characteristics of the resource, its abundance, and its distribution. Anthropogenic drivers are colored orange, to distinguish them from natural drivers (grey). As in the overarching wet system model, arrows simply represent relationships in which one model component affects or influences another. The stressor model addresses the details of these relationships.

The control model for the large river-floodplain system has two parts. The first part of Figure 9-1 consists of the large ellipse and its contents, and addresses dynamics that take place within the U.S. portion of the ecoregion. This part closely resembles the control model for the perennial stream systems in the ecoregion, presented in Chapter 8. The second part of Figure 9-1 consists of five model components placed outside the large ellipse, identifying drivers that operate on the portions of the watersheds of the three large rivers that lie outside the U.S. portion of the ecoregion. These additional drivers affect watershed and river conditions outside the ecoregion across the Gila River headwaters in west-central New Mexico; the middle and upper Rio Grande basins in New Mexico and Colorado and the Rio Concho basin in Mexico; and the upper Pecos River basin in New Mexico. These outside drivers also affect each other, but the control model does not include those additional interactions.

The Chihuahuan Desert large river-floodplain system control model specifically identifies the following system components:

- **River Flow & Water Quality** refers to the daily, seasonal, annual, and longer-term variability in river discharge, dissolved and suspended matter constituents, turbidity, and water temperature and pH. These aspects of river flow and water quality affect river fauna and flora; and both affect and are affected by floodplain soils and river morphology. High-flow pulses of river discharge, in turn, disturb soils and vegetation along river reaches where the river remains able at least occasionally to inundate its floodplain.
- **River Network Connectivity** refers to the ability of organisms, seeds, eggs, sediment, large woody debris, and other types of solid matter to move from one river reach to another, either by being carried in the river currents or through self-locomotion. Falls, rapids, and dry reaches naturally may impede up-downstream locomotion and downstream transport. This component also addresses the ability of water and solid matter to move between the river and its floodplain. Such river-floodplain interactions affect not only floodplain ecology but also river hydrology: floodplain inundation attenuates river flood peaks and extends the duration of high-flow events. Dams, artificial levees, channelization, and channel incision inhibit river network connectivity.
- **River Morphology & Sediment Dynamics** refers to the gradient, lateral and longitudinal geometry, and stability and dynamism of the river channel, including sediment erosion, transport, and deposition. Channel morphology and sediment dynamics affect river fauna and flora; and both affect and are affected by floodplain soils and hydrology, floodplain vegetation, river flow and water quality, and river network connectivity.
- **Floodplain Soils & Hydrology** refers to the spatial and temporal variability in the texture, structure, chemistry, and stability of the soils of the floodplain; floodplain inundation; and alluvial aquifer storage (depth to water) and chemistry. These dynamics both affect and are affected by river flow and water quality, river morphology and sediment dynamics, and floodplain vegetation; and also affect which faunal species may use or occupy the floodplain.

Figure 9-1. Chihuahuan desert large river-floodplain system control model.



- Floodplain Vegetation** refers to the distribution, density, composition, and structure of the floodplain vegetation community. Floodplain vegetation in turn affects river fauna and flora; and both affects and is affected by floodplain soils and hydrology. Floodplain vegetation also affects river flow & water quality, in several ways: (1) transpiration along the riparian corridor draws water from the river into the floodplain alluvial aquifer, reducing river discharge; (2) floodplain vegetation creates shade during warmer months while deciduous leaf-loss during cooler months reduces shade, both of which moderate water temperature along shorelines, side channels, and backwaters; (3) tree limbs and trunks that fall into rivers contribute to aquatic habitat

complexity; and (4) vegetative litter from the floodplain that falls into or gets swept into the river by floodwaters provides crucial inputs of organic matter to the river food web.

- **Floodplain Fauna** refers to the distribution, biomass, composition, food-web interactions, and other impacts of fauna that use riparian habitat for some or all of their life histories, including insects, reptiles, birds, and mammals. Floodplain fauna both affect and are affected by floodplain vegetation and floodplain soils and hydrology.
- **River Flora** refers to the distribution, biomass, composition, and food-web interactions of phytoplankton, periphyton, and submerged and emergent vegetation in and along a river course. Together, they comprise the in-situ primary productivity (autochthonous productivity) of the river. River flora affect and are affected by river flow and water quality, river morphology and sediment dynamics, and river fauna.
- **River Fauna** refers to the distribution, biomass, taxonomic and functional composition, and food-web interactions of the river faunal community, including zooplankton; aquatic macroinvertebrates, including insect larvae; reptiles and amphibians; and fishes, including native endemic species. This system component also addresses other aquatic and semi-aquatic vertebrates such as beaver and the invasive nutria. River fauna are affected by floodplain vegetation, the coarse and fine litter from which provides habitat and food; by river network connectivity, which affects the ability of river fauna to move within the flow network; by river flow and water quality; and by river morphology and sediment dynamics. River fauna also both affect and are affected by river flora; and by floodplain fauna, some of which may prey on river fauna and others of which (e.g., some insects) may spend parts of their life cycles as river fauna.

Several environmental components and natural drivers within the Chihuahuan Desert ecoregion also directly shape these large river-floodplain system components:

- **Aquifer Storage & Flow Dynamics** determine the amount of water stored in the bedrock and basin fill aquifers of the Chihuahuan Desert ecoregion, which in turn determines the likelihood that water from these aquifers may recharge the alluvial aquifers associated with the large rivers in the ecoregion or, in some locations, may discharge directly into a large river. Such groundwater-surface water interactions along the valleys of the three large rivers in the Chihuahuan Desert ecoregion help shape the baseflow hydrology, temperature, and chemistry of these rivers.
- **Runoff & Erosion** across watersheds surfaces deliver not only surface water to the large rivers from within the ecoregion but also sediment, particulate organic matter, and dissolved inorganic and organic matter. Runoff is the most crucial driver operating within the ecoregion to shape extreme high-flow events and overbank flooding of the riparian zone. Such high-flow events and overbank flooding also recharge alluvial aquifers.
- **Soils, Cover, Topography, and Hydrography** affect the large rivers of the ecoregion directly by controlling the large-scale geometry of the river channel network (watershed hydrography) within each river basin, including its natural connectivity; and by shaping the potential for upland wildfires to spread into the riparian zone, affecting floodplain vegetation. Chihuahuan Desert soils, cover, topography, and hydrography also affect the large rivers of the ecoregion indirectly, through their effects on watershed processes that shape water movement, chemistry, temperature across watersheds; watershed soil erosion and deposition; and the transport of sediment and organic matter across watersheds.

Several anthropogenic drivers operating within the Chihuahuan Desert ecoregion further shape these system components and natural drivers:

- **Groundwater Withdrawal** within the Chihuahuan Desert ecoregion alters aquifer system storage and flow gradients in ways that can alter groundwater-surface water interactions along affected river reaches, thereby altering river baseflow and water quality.
- **Dams & Diversions** within the Chihuahuan Desert ecoregion remove surface water from the large rivers of the ecoregion, thereby altering river flow and morphology (e.g., wetted area). The construction of surface water diversion structures also results in channel modification (see below); and dams and dry reaches created by surface water diversions alter river network connectivity.
- **Channel & Floodplain Modification** within the Chihuahuan Desert ecoregion reshapes river channel morphology to better suit human use of the large rivers of the ecoregion and their floodplains, for example to stabilize channel geometry at a highway or railroad bridge crossing or in areas of intensive recreational activity, or to stabilize floodplain agricultural field borders. Channel and floodplain modification also include the construction of levees and drainage systems to prevent river flood pulses from inundating the floodplain and to remove unwanted water from the floodplain surface and soils to enhance human use of the floodplain.
- **Invasive Species** alter the composition of the floodplain and river biotic communities. Invasive species can also alter ecological processes such as herbivory and predation on native species, competition for food and habitat among native aquatic fauna, the structure of the river food web, evapotranspiration, stream chemistry, and floodplain soil chemistry and structure. Invasive species also can affect rivers indirectly by altering watershed ground cover, soils, and wildfire regimes.
- **Livestock Grazing** within the Chihuahuan Desert ecoregion can alter floodplain vegetation within the ecoregion through herbivory; and can alter both floodplain soils and river shoreline morphology through trampling. Livestock grazing also can affect rivers in the ecoregion indirectly through its impacts on upland soils and ground cover, thereby affecting watershed processes; and by serving as a vector for the introduction of non-native species into a locality.
- **Fire Regime Modification** within the Chihuahuan Desert ecoregion, both through wildfire management and through the effects of altered watershed vegetation and climate, alter the frequency, timing, and severity of wildfires across a landscape. Such changes can affect floodplain vegetation, both directly through changes in the riparian wildfire regime and indirectly through the effects of upland wildfire on the spread of invasive species. Fire regime modifications also affect the large rivers of the ecoregion indirectly by altering land surface permeability and soil vulnerability to erosion, which in turn affect watershed processes such as infiltration and runoff.
- **Land-Use/Development** within the ecoregion alters watershed cover, land surface permeability, soil vulnerability to erosion, and releases of chemical pollutants into both watershed soils and water courses, with effects that cascade through the entire hydrologic system. Land use and development also shapes wildfire management policies and actions.
- **Air Pollution**, i.e., air pollution within the ecoregion, affects the types and rates of deposition of air pollutants (atmospheric deposition) across the ecoregion, with cascading effects on (a) upland soils and cover and (b) the chemistry of surface runoff and groundwater recharge across the watersheds of the ecoregion.
- **Altered Climate Drivers** affect aquatic and floodplain conditions along the large rivers of the Chihuahuan Desert ecoregion indirectly, through their effects on the weather and its variation across the ecoregion. Changes in the weather, including changes in air temperature and precipitation patterns, directly affect watershed processes within the ecoregion, including evapotranspiration, precipitation form, snowpack formation and snowmelt, infiltration and

recharge, and watershed runoff and erosion. Changes in watershed processes in turn have cascading effects on upland soils and cover, groundwater storage, and river flows.

Finally, five anthropogenic drivers operating outside the Chihuahuan Desert ecoregion further shape large-river system components and natural drivers within the ecoregion:

- **Altered Climate Drivers** have effects outside (as well as inside) the Chihuahuan Desert ecoregion that affect aquatic and floodplain conditions along the large rivers within the ecoregion. These effects are indirect, involving impacts on the weather and its variation across the upstream catchments of the three Chihuahuan Desert large rivers outside the ecoregion. Again, changes in the weather, including changes in air temperature and precipitation patterns, directly affect watershed processes within these upstream catchments, including evapotranspiration, precipitation form, snowpack formation and snowmelt, infiltration and recharge, and watershed runoff and erosion. Changes in watershed processes across the upstream catchments of the Gila River, Pecos River, and Rio Grande in turn have cascading effects on the patterns of discharge of water and transported sediment from these upstream catchments into the Chihuahuan Desert ecoregion.
- **Upstream Water Use**, across the catchments of the three large rivers that lie outside the U.S. portion of the ecoregion, including both surface and groundwater consumption, reduce the amount of water that these rivers deliver to the U.S. portion of the ecoregion from these upstream catchments. The effects vary by season, and may also be affected by dam operations along these rivers both within and outside the ecoregion (see below, Upstream Dams).
- **Upstream Land Use and Development** affect watershed processes across the catchments of the three large rivers that lie outside the U.S. portion of the ecoregion, including causing changes to the rates and timing of evapotranspiration, runoff, and recharge across these upstream catchments. Development along the floodplains within these basins also results in changes to the extent of floodwater storage capacity in these basins and alters river-floodplain interactions. All these changes in turn result in changes to the patterns of delivery of water, sediment, suspended plant matter, and pollutants by the large rivers from their upstream catchments to the U.S. portion of the ecoregion.
- **Upstream Dams** regulate the delivery of water from the upstream catchments of the Rio Grande and Pecos River, thereby modifying their flow patterns within the U.S. portion of the ecoregion; trap sediment and suspended plant matter, preventing their delivery downstream; lose water to evaporation; modify river water chemistry; and block movement of river fauna. There are no dams on the Gila River within the ecoregion.
- **Upstream Air Pollution** across the upstream catchments of the three large rivers outside the U.S. portion of the ecoregion results in further introductions of pollutants into the waters of these rivers within the U.S. portion of the ecoregion, through atmospheric deposition and subsequent incorporation into runoff.

9.3 Large River-Floodplain System Stressor Model

Table 9-1 presents and defines the drivers, critical environmental elements, critical ecological processes, and ecological outcomes that characterize the large river floodplain system stressor model. The stressor model follows the methodology described in Chapter 4.

Table 9-1. Chihuahuan desert large river-floodplain system stressor model drivers, critical environmental elements, critical ecological processes, and ecological outcomes

Model Component	Definition
Drivers	
Air Temperature Regime	The pattern of variation in air temperature within the ecoregion, including daily, seasonal, annual, and longer-term variation; and the magnitude, frequency, timing, and duration of maxima and minima.
Atmospheric Deposition	The pattern of variation in the deposition of potential pollutants from the atmosphere onto the land and water surfaces of the ecoregion, including variation in pollutant types, and rates of wet, dry, and total deposition.
Domestic Grazing Management	The pattern of management of the spatial distribution, timing, duration, frequency, and density of domestic livestock grazing within the ecoregion.
Fire Management	The pattern of management of the spatial distribution, timing, and frequency of wildfire suppression and prescribed burns within the ecoregion.
Non-Native Species Introductions	The types, origins, and patterns of introduction (where, when, how) of non-native species into the ecoregion. This driver does not include domesticated livestock or species intentionally introduced by fish and game managers for recreational sport.
Precipitation & Snowmelt Regime	The form (rain, ice, snow) and pattern of variation in precipitation, including daily, seasonal, annual, and longer-term variation in magnitude, frequency, timing, and rate (intensity); the annual pattern of variation in the rate and timing of snowmelt; and the chemistry of the precipitation within the ecoregion.
Sport & Nuisance Species Management	The pattern of management of sport and nuisance species spatial distributions and densities within the ecoregion, including official management by governmental agencies and private management by individuals and non-governmental organizations; and including species rearing, releasing, monitoring, control, and removal; and also including removal of non-native species and habitat restoration to benefit native species.
Water Management & Use	The pattern of management of surface and groundwater storage, movement, and use (where, when, at what magnitudes) within the U.S. portion of the ecoregion by public and private institutions and private individuals, controlled by structures such as dams, diversions, well fields, conveyances, and levees, including “return flows” from agricultural, industrial, and municipal use. (In other ecoregions, this driver would also address management of water for navigation and/or hydropower generation, neither of which pertains within the U.S. portion of the ecoregion.)
Watershed & Riparian Land Development	The pattern of development of the land surface within the U.S. portion of the ecoregion to support human activities, involving intentional modification of vegetation, soils, or topography and/or construction and maintenance of structures and engineered surfaces; pollutants released by the associated human activities; and riparian habitat restoration.
Critical Environmental Elements	
Channel Reach-Scale Morphology	The overall shape and stability of river channels at the multi-kilometer scale within a watershed, including the types, abundance, relative confinement, and spatial and temporal distributions of (a) natural features such as confluences/deltas, linear reaches, braiding, bends and meanders, falls and rapids; and (b) artificial features such as dams and other barriers, channel control structures, and levees.
Fire Regime	The pattern of spatial distribution, extent, severity (intensity), timing, and frequency of fire on the landscape, as affected both by natural fuel and ignition dynamics and by management actions including prescribed burns.
Floodplain Levee & Drainage Systems	The location, spatial extent, and patterns of operation of artificial levees and drainage systems on floodplains constructed to prevent inundation and/or drain floodplain soils to increase their availability for human use.

Model Component	Definition
Fluvial Network Connectivity	The capacity of a river network to support the downstream transport of matter such as sediment, large woody debris and other plant litter, seeds and other propagules, plankton, and larger aquatic organisms; and the upstream movement of aquatic organisms such as migratory fishes, as determined by the spatial distribution of natural and artificial features that may prevent or inhibit such transport or movement.
Runoff Water Quality	The chemical properties of the water that runs off a watershed into a river, including temperature, pH, turbidity, and concentrations of dissolved and suspended constituents swept in off the watershed; and the patterns of variation in these properties, including daily, seasonal, annual, and longer-term variation in their magnitudes.
Runoff Regime	The pattern of variation in the amount of water flowing from a watershed into a river, including daily, seasonal, annual, and longer-term variation; and including the frequency, timing, and duration of particular flow rates or stages such as “floods.”
Upstream Discharge Regime	The discharge regimes of the Gila River, Pecos River, and Rio Grande at the points where they enter the U.S. portion of the ecoregion, and of the Rio Conchos and other tributaries in Mexico at the points where they flow into the Rio Grande, including their baseflow regimes and low-flow and high-flow pulse dynamics (see definitions of baseflow and low-flow and high-flow pulses below under Critical Ecological Processes).
Upstream Fluvial Network Connectivity	The capacities of the fluvial networks of the Gila River, Pecos River, and Rio Grande above the points where they enter the U.S. portion of the ecoregion, and of the Rio Conchos and other tributaries in Mexico above the points where they flow into the Rio Grande, to support the downstream transport of matter such as sediment, large woody debris and other plant litter, seeds and other propagules, plankton, and larger aquatic organisms; and the upstream movement of aquatic organisms such as migratory fishes, as determined by the spatial distribution of natural and artificial features that may prevent or inhibit such transport or movement.
Upstream Water Quality	The physical and chemical properties of the water in the Gila River, Pecos River, and Rio Grande at the points where they enter the U.S. portion of the ecoregion, and in the Rio Conchos and other tributaries in Mexico at the points where they flow into the Rio Grande, including temperature, pH, turbidity, and concentrations of different types of dissolved and suspended particulate matter, both inorganic and organic; and the patterns of variation in these properties, including daily, seasonal, annual, and longer-term variation in their magnitudes.
Water Impoundments, Diversions, Returns	The distribution and size of (1) river reaches that have been converted from natural fluvial conditions into either impounded (lacustrine) conditions by dams or other engineered structures, including those constructed by beavers; (2) diversions from rivers that function as distributary channels to water users; and (3) return-flow or drainage channels that collect and return river water back to the main channel from points of use.
Watershed Erosion	The pattern of daily, seasonal, annual, and longer-term variation in the amount (mass and volume) and particle size distribution of sediment eroded off the surface of a watershed and transported into its rivers suspended in the watershed runoff.
Watershed Ground Cover	The composition of the surface of a watershed in terms of the abundances and spatial distributions of classes of vegetated, disturbed, and artificial surfaces that differ in their permeability to water infiltration, hydraulic roughness to water runoff, ability to inhibit soil erosion, and provision of shade – the latter of which can affect runoff temperatures and snowmelt). Dissolved and particulate organic matter produced by watershed ground cover and swept by watershed runoff into rivers helps support the aquatic food web within the river.
Watershed-Scale Groundwater Dynamics	The locations and rates of recharge of precipitation to groundwater systems; the storage volumes, inter-connections, and flow path lengths and duration of the aquifers that comprise the groundwater system(s) of a watershed; the geochemical and hydrothermal dynamics of these groundwater systems; and the locations and rates of discharge to the ground surface from these groundwater systems.

Model Component	Definition
Critical Ecological Processes	
Alluvial Water-Table Dynamics	The pattern of daily, seasonal, annual, and longer-term variation in the elevation of the alluvial water table of riparian reach, including the frequency, timing, and duration of particular water table elevations such as the “X-year” maxima or minima.
Aquatic Primary (1 ^o) Productivity	The pattern of daily, seasonal, annual, and longer-term variation in the rate of primary (<i>aka</i> autochthonous) production of biomass through photosynthesis by organisms such as algae and aquatic and emergent plants. Autochthonous production differs from allochthonous production, the contributions of organic matter from the watershed and riparian corridor to the river. The stressor model addresses the latter as a component of runoff water quality and the impacts of riparian vegetation on river conditions.
Baseflow Regime	The pattern of seasonal, annual, and longer-term variation in the stage of the water in a river when it is fed only by groundwater discharge without any inputs from runoff. In regulated rivers, natural baseflow may be replaced by an artificial normal flow level.
Floodplain Inundation	The pattern of seasonal, annual, and longer-term variation in the frequency, magnitude (spatial extent), timing, and duration of floodplain inundation as a result of over-bank flooding during high-flow pulses. Inundation clearly is related to high-flow pulses, specifically to high flows that result in overbank flooding. The two processes are distinguished here to allow for the separate discussion of the effects of inundation <i>per se</i> , recognizing that many high-flow pulses do not result in overbank flooding.
Fluvial Biotic Connectivity	The downstream transport of large woody debris and other plant litter, seeds and other propagules, phytoplankton and zooplankton, and larger aquatic organisms including fish larvae; and the upstream and downstream movement of aquatic organisms such as fishes, as characterized by properties such as the timing, rates, and distances of transport or movement.
Fluvial-Alluvial Sediment Dynamics	The pattern of erosion, deposition, and storage of sediment along a river and between the river and its floodplain, as characterized by properties such as the annual and longer-term sediment mass balance of a given river reach; patterns of channel and floodplain aggradation and degradation; and changes in wetted area.
High-Flow Pulse Dynamics	The pattern of seasonal, annual, and longer-term variation in the stage and rate of discharge of a river when it is fed by large pulses of runoff or large releases from dams, as characterized by properties such as the magnitude, timing, duration, and hydrograph shape of the annual maximum flow; and the magnitude, timing, duration, and hydrograph shape of long-term effective, bankfull, and flood flows (approx. 1.5-year; 2-year; and less frequent flows such as the 10-year, 50-year, and 100-year “7-day” high flows).
Low-Flow Pulse Dynamics	The pattern of seasonal, annual, and longer-term variation in the stage and rate of discharge of a river during periods drought or during periods of reduced dam releases, as characterized by properties such as the magnitude, timing, and duration of the annual minimum flow; and the magnitude, timing, and duration of long-term 10-year, 50-year, and 100-year “7-day” low flows).
Riparian-Aquatic Native-Exotic Species Interactions	The ways, magnitudes, and spatial and temporal extent to which native and exotic riparian and aquatic species compete for habitat space, food, and other materials; prey on each other; infect or otherwise harm each other; or interact beneficially (mutualism).
River Meso-Habitat Dynamics	The frequencies of creation and destruction of meso-habitat features such as floodplain terraces, natural levees, channel bends, side channels, oxbow lakes, backwaters, islands, bars, pools, eddies, riffles, stranded snags (large woody debris), and bank overhangs in/along the channel; and the spatial extent and persistence of such features.
River Water Quality Dynamics	The physical and chemical properties of the water in a river, including temperature, pH, turbidity, and concentrations of different types of dissolved and suspended particulate matter, both inorganic and organic; and the patterns of variation in these properties, including daily, seasonal, annual, and longer-term variation in their magnitudes.

Model Component	Definition
Ecological Outcomes	
Amphibian Composition	The taxonomic, functional, and size composition; spatial and temporal distribution; abundance; health; and activity level of the amphibian assemblages of a river corridor.
Benthic Invertebrate Composition	The taxonomic, functional, and size composition; abundance; spatial and temporal distribution; and activity level of the aquatic invertebrate assemblage of a river, including biofilms and periphyton, phytoplankton, zooplankton, insect larvae, crayfish, and mollusks.
Bird Composition	The taxonomic, functional, and size composition; spatial and temporal distribution; abundance; health; and activity level of the avifaunal assemblage of a riparian corridor.
Emergent Vegetation Composition	The taxonomic composition; size range; spatial and temporal distribution; and abundance of emergent (aquatic) vegetation along a river.
Fish Composition	The taxonomic, functional, and size composition; spatial and temporal distribution; abundance; health; and activity level of the fish assemblages of a river.
Reptile & Mammal Composition	The taxonomic, functional, and size compositions; spatial and temporal distributions; abundances; health; and activity levels of the assemblages of reptiles and mammals that occupy or visit the riparian zone.
Riparian Insect Composition	The taxonomic, functional, and size composition; abundance; spatial and temporal distribution; and activity level of the insect assemblage of the riparian zone.
Riparian Vegetation Composition & Structure	The taxonomic composition; size range; spatial and temporal distribution; health; vertical above-ground (e.g., canopy) and below-ground (e.g., rooting) structure; and abundance of vegetation along a riparian corridor. This outcome includes the shade and organic matter, including large woody debris, provided by riparian vegetation to the river.

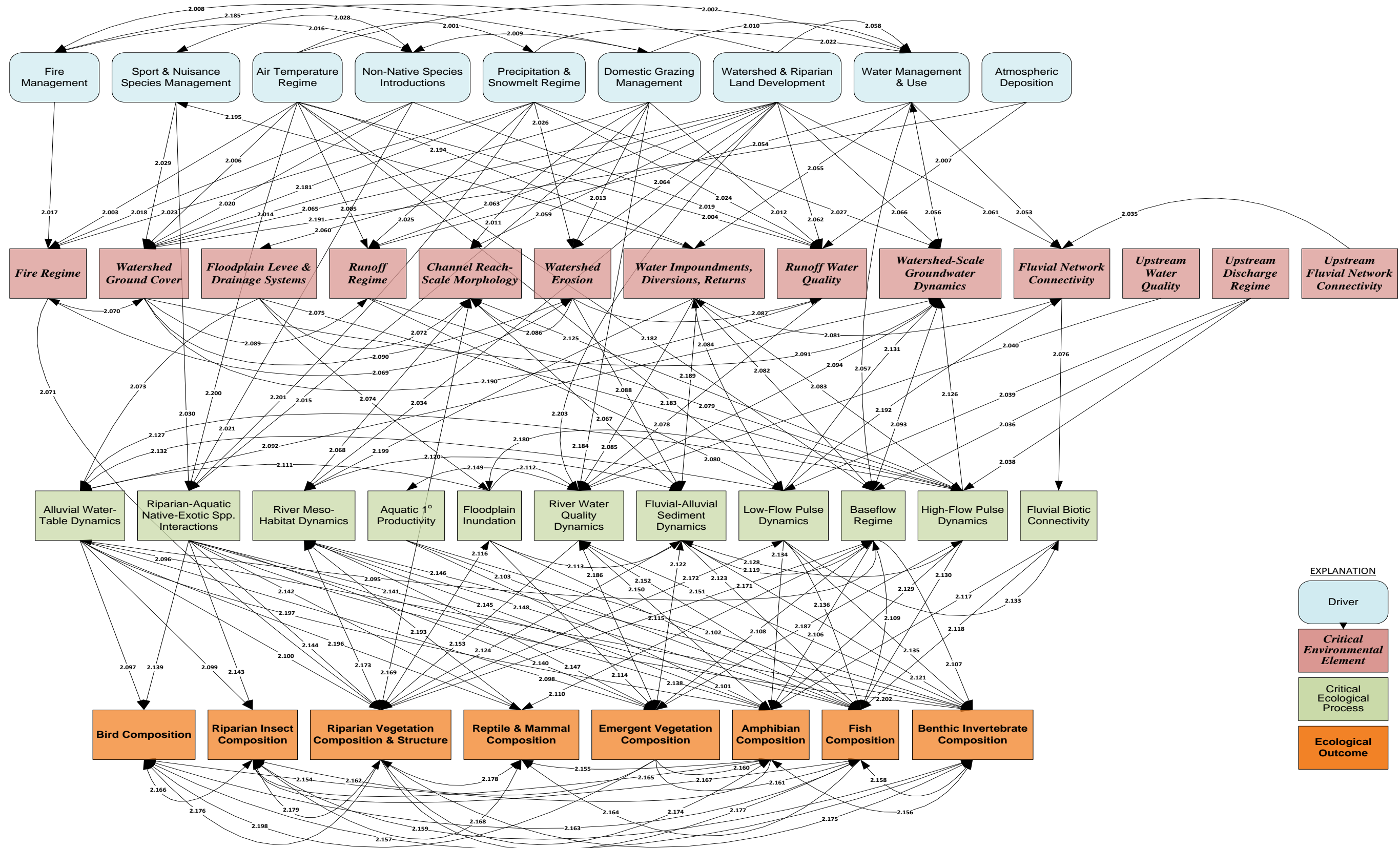
Figure 9-2 shows the stressor model for the large river-floodplain system in the Chihuahuan Desert ecoregion, built using the system model components shown in Table 9-1. It displays all the system model components listed in Table 9-1, along with their causal relationships. Appendix 1 describes and presents the rationale for including every causal relationship in the stressor model, with citations for each causal relationship, providing a comprehensive presentation of the stressor model. Figure 9-2 indicates the presence or absence of causal relationships between the system model components but does not indicate the potential magnitude or other characteristics of these relationships, as explained in Chapter 4.

As with the perennial stream systems stressor model (see Chapter 8), the large river-floodplain system stressor model (1) identifies the causal relationships that have affected how the condition of the system likely has changed in the past, in response to changes in its drivers; and (2) provides a means for articulating hypotheses about how the condition of the system will likely change in response to changes in its drivers. The second capability of stressor models is crucial for expressing individual management questions as hypotheses, as discussed in detail below. As discussed and illustrated below, the stressor model makes it clear: (a) which critical environmental elements would likely be affected by a change in one or more particular drivers, including change agents; (b) which critical ecological processes would likely be affected by the cascading effects of these changes in critical environmental elements; and (c) which system characteristics (ecological outcomes) would likely be affected by the cascading effects of these changes in critical environmental elements and ecological processes. By doing so, in turn, the stressor model also highlights those components of the model—drivers, environmental elements, ecological processes, and ecological outcomes—that demand indicator data.

As defined earlier, a causal relationship exists when a change in one component of the system results in a change in some other component. Change in the first component is said to “cause a change in the second component. Each chain of causation, from driver to outcome, describes how the condition of the system likely has changed in the past, or likely would change in the future, in response to changes in its drivers.

As with the perennial stream systems stressor model, the large river-floodplain system stressor model identifies known or likely causal relationships among the drivers, environmental elements, ecological processes, and ecological outcomes for the system. As defined earlier, a causal relationship exists when a change in one component of the system results in a change in some other component. Change in the first component is said to “cause a change in the second component. Each chain of causation, from driver to outcome, describes how the condition of the system likely has changed in the past, or likely would change in the future, in response to changes in its drivers.

Figure 9-2. Chihuahuan desert large river-floodplain system stressor model.



The details included in the large river-floodplain system stressor model also highlight those components of the model – drivers, environmental elements, ecological processes, and ecological outcomes – that demand indicator data. The stressor model for the large river-floodplain system takes this one step further, distinguishing three critical environmental elements located outside the U.S. portion of the ecoregion that the Gila River, Pecos River, and the Rio Grande within the U.S. portion of the ecoregion. These three “upstream” environmental elements integrate the effects of climate, land use, water use, and water infrastructure along and across the watersheds of the Gila River, Pecos River, and Rio Grande upstream from the points where they enter the U.S. portion of the ecoregion, and along and across the watersheds of the Rio Conchos and other tributaries in Mexico at the points where they flow into the Rio Grande.

Figure 9-2 omits the three gray, background fields included in the perennial stream systems stressor model (see Figure 8-2, Chapter 8). Those fields are included in the earlier diagram to illustrate two points. First, the critical environmental elements of the system within the ecoregion derive from and are shaped by long-term dynamics of watershed geology and topography. Second, the ecological outcomes of the system stressor model depend in part on the compositions of the regional reservoirs of native animal and plant species. These relationships are implicit in Figure 9-2 but are omitted to avoid further crowding an already crowded diagram.

The following paragraphs discuss how each Change Agent affects Chihuahuan Desert Large River-Floodplain Systems, as articulated in the stressor model for this CE. A “sub-model” diagram for each Change Agent presents a simplified version of the master stressor model diagram, showing only the direct and indirect effects of the Change Agent of interest. The text accompanying each sub-model diagram summarizes the information from Appendix 1 concerning the impacts of each Change Agent, and includes selected citations from the more detailed list of citations for each causal link presented in Appendix 1. For each Change Agent, the paragraphs summarize present understanding of: (a) which critical environmental elements would likely be affected—directly or indirectly—by each Change Agent and (b) which critical ecological processes would likely be affected—directly or indirectly—by the cascading effects of these changes in critical environmental elements.

However, the ecological characteristics of the system (ecological outcomes) affect each other and are affected by critical environmental elements and ecological processes in the same way regardless of which Change Agent is involved in altering these elements and ecological processes. For this reason, the presentation below begins with a discussion of the interactions between critical ecological processes and ecological outcomes, and among ecological outcomes. This latter discussion closely resembles that presented for the perennial stream systems CE in Chapter 8: The outcome types and their interactions in the three large river-floodplain systems and the perennial streams of the U.S. portion of the ecoregion are essentially the same. Nevertheless, the large river-floodplain systems differ from the perennial stream systems in the U.S. portion of the ecoregion in several ways: (1) in the influence of climate, land use, water use, and water infrastructure *outside* the U.S. portion of the ecoregion on river conditions *within* the U.S. portion of the ecoregion; (2) in the current or former (pre-regulation) magnitude of many riverine characteristics, including discharge, depths, wetted areas, and sediment loads; (3) in the current or former (pre-regulation) intensity of interactions between the rivers and their floodplains, including

through floodplain inundation; and (4) the current or former (pre-regulation) presence or greater abundance of aquatic species adapted to the unique hydrologic and water quality conditions of the larger alluvial rivers.

9.3.1 Critical Ecological Processes and Ecological Outcomes

The large river-floodplain system stressor model includes eight ecological outcomes: amphibian composition, benthic invertebrate composition, bird composition, emergent vegetation composition, fish composition, reptile & mammal composition, riparian insect composition, and riparian vegetation composition & structure. Table 9-1, above, defines these eight model components. These eight ecological outcomes directly affect each other in numerous ways – and therefore also affect each other indirectly in even more numerous ways – as shown in Figure 9-2 and documented in Appendix 1. For example (see Appendix 1 for full presentation):

- The composition and density of riparian and emergent vegetation affect native benthic invertebrate, amphibian, and fish assemblage composition in the large rivers by providing habitat such as substrates and cover and/or by providing food to various life stages (e.g., Allan 1995, Propst 1999, Rosen and Caldwell 2004, Bateman et al. 2008a, Wallace and Anderson 2008, USFWS 2009, Kinzli and Myrick 2010, Boeing et al. 2014, Forstner et al. 2014).
- Riparian vegetation composition and structure affect terrestrial riparian insects and vertebrate fauna by providing food and habitat options, which differ depending on the composition and structure of the riparian vegetation, and by shaping the physical environment including shade, humidity, and soil moisture (e.g., Bateman et al. 2008a; 2008b; 2008c; 2009, Malcom and Radke 2008, Andersen and Shafroth 2010, Ruth et al. 2010, Wild 2011, Merritt and Bateman 2012, Brand et al. 2013, Minckley et al. 2013, Refsnider et al. 2013, Forstner et al. 2014, Gibson and Olden 2014, Smith and Finch 2014).
- Emergent vegetation provides nesting and feeding habitat for waterfowl. The types and quality of this habitat varies with the composition, spatial and temporal distribution, and abundance of emergent vegetation along the three rivers and across their floodplains, thereby affecting the composition of the bird assemblage along the rivers (e.g., NMDGF 2006, Lougheed and Rodriguez 2008, Malcom and Radke 2008, Ruth et al. 2010, Merritt and Bateman 2012, Brand et al. 2013, Minckley et al. 2013, Smith and Finch 2014).
- Riparian insects provide food options for birds, reptile and mammals, and amphibians along the three river corridors – options that differ depending on what insects are available, at what times, and in what abundances. The composition and abundance of the riparian insect assemblage therefore can affect the composition of the bird, reptile and mammal, and amphibian assemblages along the river corridors (Hunter et al. 1985, Johnson and Haight 1985, Kozma and Mathews 1997, Skagen et al. 1998, Krueper et al. 2003, Hinojosa-Huerta et al. 2004, Rosen and Caldwell 2004, Makings 2005, Skagen et al. 2005, Price et al. 2005, Rosen 2005, Rosen et al. 2005, Stromberg et al. 2005, Brand et al. 2006, NMDGF 2006, Stromberg et al. 2006, Bateman et al. 2008a; 2008b, Levick et al. 2008, Cerasale and Guglielmo 2010, Ruth et al. 2010, Nagler et al. 2011, Hagen and Sabo 2012; 2014, Oring et al. 2013, Flesch 2014, Forstner et al. 2014). Freshwater fish may sometimes also consume riparian insects that fall into the water (e.g., Tyus and Minckley 1988), although this has not been documented in the present ecoregion.
- Amphibians along the three river corridors provide food options for birds and reptiles and mammals – options that differ depending on what amphibians are available, at what times, and

in what abundances. The composition and abundance of the amphibian assemblage therefore can affect the composition of the bird and reptile and mammal assemblages along these river corridors (e.g., Mora et al. 2002, Schmitt et al. 2005, White et al. 2006, Bateman et al. 2009; 2013).

- Fish in the large rivers also provide food options for birds and reptiles and mammals – options that differ depending on what fish are available, at what times, and in what abundances. The composition and abundance of the fish assemblage therefore can affect the composition of the bird and reptile and mammal assemblages along the three river corridors (e.g., Schmidly and Ditton 1978, MacRae et al. 2001, Mora et al. 2002, Schmitt et al. 2005, Bateman et al. 2009, Forstner et al. 2014).
- Benthic invertebrates in the three large rivers, at least in shallows, provide food options for some birds – options that again vary depending on what invertebrates are available, at what times, and in what abundances. The composition and abundance of the benthic invertebrate assemblage in these rivers therefore can affect the composition of the bird assemblages along their riparian corridors. Additionally, birds that consume benthic invertebrates can bioaccumulate contaminants (e.g., organochlorines, mercury) that these invertebrates have bioaccumulated through their own diets and exposures. This can result in high body loads of such contaminants in the birds, to such high levels that this impairs health and reproduction (e.g., MacRae et al. 2001, White et al. 2006).
- Benthic invertebrates in the three large rivers also provide food options for fish and amphibians – options that again vary depending on what invertebrates are available, at what times, and in what abundances. In turn, feeding pressure by fish and amphibians can affect the composition and abundance of the benthic invertebrate assemblage, with these pressures varying depending on what fish and amphibians are present, at what times, and in what abundances. Consequently, the composition and abundance of the benthic invertebrate assemblage in the three large rivers can both affect and be affected by the composition of the bird assemblages along their riparian corridors (e.g., Allan 1995, Karr and Chu 1999, Stoddard et al. 2005, Witte 2005, Bergeron et al. 2011, Luce et al. 2012, Rolls et al. 2013).
- Riparian insect larvae are important constituents of the benthic invertebrate assemblage. The composition and abundance of the benthic invertebrate assemblage therefore both affects and is affected by the composition of the riparian insect assemblages (e.g., Allan 1995, Karr and Chu 1999, Stoddard et al. 2005, Wallace and Anderson 2008).

The large river-floodplain system stressor model includes eleven critical ecological processes that directly affect the ecological outcomes discussed above, shown in the following order in Figure 9-2: alluvial water-table dynamics, riparian-aquatic native-exotic species interactions, river meso-habitat dynamics, aquatic ¹ productivity, floodplain inundation, river water quality dynamics, fluvial-alluvial sediment dynamics, low-flow pulse dynamics, baseflow regime, high-flow pulse dynamics, and fluvial biotic connectivity. Table 3, above, defines these eleven model components. These eleven critical ecological processes directly affect—and in some cases are also affected by—the eight ecological outcomes in numerous ways, as shown in Figure 9-2 and documented in Appendix 1. Alterations to these critical ecological processes as a result of changes in drivers and critical environmental elements necessarily lead to altered ecological outcomes. The following paragraphs provide examples of the interactions of the eleven critical ecological processes with ecological outcomes in relatively unaltered systems (see Appendix 1 for full presentation):

- Alluvial water-table dynamics directly affect riparian and emergent vegetation dynamics by affecting the depth to the water table across the floodplain, and the vegetation reciprocally affects alluvial water table dynamics through evapotranspiration (Scott et al. 2004, Lite and Stromberg 2005, Price et al. 2005, Stromberg et al. 2005, Leenhouts et al., eds. 2006, Stromberg et al. 2006, Baillie et al. 2007, Scott et al. 2008, Katz et al. 2009, Doody et al. 2011, Nagler et al. 2011). Alluvial water-table dynamics directly affect the composition of the bird, riparian insect, reptile and mammal, and amphibian assemblages by affecting soil moisture and humidity levels across the floodplain, including the presence and extent of surface water – both lotic and lentic – and wetlands, all of which may be crucial factors affecting habitat quality for these species (Bateman et al. 2008b; 2009, Levick et al. 2008, Stromberg et al. 2005, Brand et al. 2006, Stromberg et al. 2006, Loughheed and Rodriguez 2008, Cerasale and Guglielmo 2010, Ruth et al. 2010, McCluney and Sabo 2012; 2014, Oring et al. 2013, Flesch 2014, Mosher and Bateman 2016). Finally, alluvial water-table dynamics may affect the availability of hyporheic habitat for some benthic invertebrates (Hancock et al. 2005, Boulton et al. 2010, Tockner et al. 2010).
- Riparian-aquatic native-exotic species interactions have pervasive effects across all ecological characteristics of three large river-floodplain systems, as discussed later in this chapter (see Invasive Species, below).
- River meso-habitat dynamics establish the physical habitat template and the dynamics of that template for all fishes and benthic invertebrates and for all emergent plant species, amphibians, and semi-aquatic mammals that use riverine shallows and shorelines, and their interactions with each other (e.g., Pease et al. 2006, Stromberg et al. 2006, Dudley and Platania 2007; 2011, Higgins and Strauss 2008, Loughheed and Rodriguez 2008, Propst et al. 2008, Magaña 2009, Hoagstrom et al. 2010, Kinzli and Myrick 2010, Theobald et al. 2010, USFWS 2010, Dean and Schmidt 2011, Nagler et al. 2011, Karatayev et al. 2012, Connally, ed., 2012a, Haase et al. 2012, Heard et al. 2012, Booth et al. 2013, Gido et al. 2013, Jones and Woods, eds., 2013, Garrett and Edwards 2014, Sandoval-Solis and McKinney 2014, Worthington et al. 2014).
- Aquatic 1° (primary or autochthonous) productivity strongly affects the abundance of benthic invertebrates, amphibians, and fish that feed on the resulting biomass (e.g., Kupferberg 1997, USFWS 2009, Luce et al. 2012, Turner and Edwards 2012, Magaña 2013, Rolls et al. 2013, Wellard Kelly et al. 2013, Boersma et al. 2014, East 2015, Wood et al. 2016, Propst 2016).
- Floodplain inundation is a crucial natural disturbance process affecting riparian vegetation in relatively unregulated river systems. It maintains a complex vegetation mosaic by removing riparian vegetation patches, disturbing succession in others, and establishing new patches. Inundation pulses create conditions that actively promote seed germination and seedling establishment of some riparian plants such as cottonwood and willow. Floodplain inundation in relatively unregulated river systems also transports and deposits seeds and other plant propagules, and deposits sediment, nutrients, and organic matter across the floodplain. Reciprocally, riparian vegetation affects floodplain hydraulic roughness, which affects flow velocities and depths in the inundating water and the duration of overbank flooding. Alterations to the timing and magnitude of inundation (along with changes in the sediment load and water quality of the inundating water) can alter the composition, abundance, and spatial distribution of riparian vegetation across the floodplain. The spatial extent of these alterations varies with the spatial extent of the inundation, which in turn varies with the magnitude and duration of the high-flow pulse responsible for the flooding and the presence or absence of artificial structures that may limit inundation (Shafroth and Beauchamp 2006, Hultine et al. 2007, Stromberg et al. 2006; 2007; 2012, Loughheed and Rodriguez 2008, Merritt et al. 2010, Theobald et al. 2010, Dean and Schmidt 2011, Nagler et al. 2011, Poff et al. 2011, Sandoval-Solis and McKinney 2014).

- As with riparian vegetation, floodplain inundation can remove emergent vegetation patches, disturb succession in others, or help establish new patches by transporting and depositing seeds and other plant propagules. As noted above, floodplain inundation in relatively unregulated river systems also transports and deposits seeds and other plant propagules, and deposits sediment, nutrients, and organic matter across the floodplain. Alterations to the timing and magnitude of inundation (along with changes in the sediment load and water quality of the inundating water) can alter the composition, abundance, and spatial distribution of emergent vegetation in floodplain and shoreline wetlands (Opperman 2008, Katz et al. 2009, Andersen and Shafroth 2010, USBR 2012, Minckley et al. 2013, Cole and Cole 2015).
- Floodplain inundation affects aquatic and semi-aquatic fauna along the three large rivers and their floodplains, as well. These effects are mostly indirect, through the effects of floodplain inundation on the vegetation, alluvial water table dynamics, fluvial-alluvial water table dynamics, and river water quality. However, floodplain inundation can also affect aquatic fauna along the three rivers, among fish species that use floodplains as refuge, feeding, or reproductive habitat during inundation. Alterations to the timing and magnitude of inundation (along with changes in the sediment load and water quality of the inundating water) can alter the abundance and diversity of aquatic and semi-aquatic fauna along the three large rivers and their floodplains (King et al. 2003, Magoulick and Kobza 2003, Pease et al. 2006, Magaña 2009; 2013, Propst et al. 2009, Kinzli and Myrick 2010, Turner et al. 2010, Opperman 2012, Gido et al. 2013, Medley and Shirey 2013, Miyazono 2014, Worthington et al. 2014).
- River water quality dynamics directly affect the composition and abundance of both riparian and emergent vegetation by affecting the availability of dissolved nutrients, and salts and potentially harmful chemicals; and the vegetation reciprocally can affect the concentrations of salts and potentially harmful chemicals as well (e.g., Deloach et al. 2000, Mainston and Parr 2002, Seiler et al. 2003, Lite and Stromberg 2005, Shafroth et al. 2005, USEPA 2005, Bhattacharjee et al. 2006, Chipps et al. 2006, Mitsch and Gosselink 2007, Shafroth et al. 2008, Johnston et al. 2009, Rooney and Bayley 2010).
- River water quality dynamics directly affect the composition and abundance of the benthic invertebrate, amphibians, and fish assemblages, because all aquatic fauna are sensitive to variation in water temperature, pH, turbidity, salinity, and concentrations of specific chemical constituents such as metals and organochlorines. These properties of the water can affect organism health, development, reproduction, feeding activities, and vulnerabilities to predation; can cause them to depart from or avoid affected river reaches with intolerable water quality; or, if they are in fact adapted to extreme conditions of water quality, can allow them to safely occupy affected river reaches at the expense of other species (Cowley and Sublette 1987, Allan 1995, Edwards 1997, Karr and Chu 1999, Propst 1999, MacRae et al. 2001, Edwards et al. 2002, Cowley et al. 2003, Hoagstrom 2003, Calamusso 2005, Stoddard et al. 2005, NMDGF 2006, White et al. 2006, Zymonas and Propst 2007, Gregory and Hatler 2008, Hoagstrom 2009, Theobald et al. 2010, Witte 2005, Connally, ed., 2012a, Heard et al. 2012, Karatayev et al. 2012, Jones and Woods, eds., 2013, Garrett and Edwards 2014, Hubbs 2014, Miyazono 2014). Additionally, benthic invertebrates may bio-accumulate contaminants and pass them up the food chain, as discussed above.
- River water quality dynamics also affect the benthic invertebrate, amphibians, and fish assemblages indirectly by affecting (a) aquatic primary productivity and (b) the concentrations of allochthonous organic matter carried into the stream by runoff (Allan 1995, e.g., White et al. 2006, Zymonas and Propst 2007, Hoagstrom 2009, Theobald et al. 2010, Gregory and Hatler 2008, Heard et al. 2012, Karatayev et al. 2012, Jones and Woods, eds., 2013, Garrett and Edwards 2014, Hubbs 2014, Miyazono 2014).

- Fluvial-alluvial sediment dynamics affect the ecological characteristics of the three large rivers directly by affecting the overall stability and particle size distributions of habitat substrates, for which different benthic invertebrate and fish species have different preferences and tolerances (Magaña 2009, Herbst and Cooper 2010, Hoagstrom et al. 2010, Kinzli and Myrick 2010, Theobald et al. 2010, Heard et al. 2012, Karatayev et al. 2012, Jones and Woods, eds., 2013). The erosion, transport, and deposition of sediment along a river corridor also affect habitat quality for riparian and emergent vegetation of the river: Erosive disturbances of submerged and exposed alluvial soils reset succession in the disturbed areas, while rooted vegetation in turn can stabilize alluvial soils so that they resist erosive disturbance (Schmidly and Ditton 1978, Stromberg et al. 2006, Loughheed and Rodriguez 2008, Theobald et al. 2010, Dean and Schmidt 2011, Nagler et al. 2011, Connally, ed., 2012a, Sandoval-Solis and McKinney 2014). Fluvial-alluvial sediment dynamics also affect the ecological characteristics of the three large rivers indirectly, by affecting two other critical ecological processes, baseflow and river meso-habitat dynamics.
- High-flow and low-flow pulse dynamics affect the ecological characteristics of the three large rivers directly through their effects on fish, amphibian, and benthic invertebrate species. These species differ from each other in their needs for high-flow pulses as triggers for reproduction or movement, in their abilities to cope with the extremes of velocity and turbulence associated with extreme-high-flow pulses, and in their abilities to tolerate and recover from extended low-flow pulses (Propst 1999, Schmidt et al. 2003, Stromberg et al. 2005, Dudley and Platania 2007, Zymonas and Propst 2007, Bateman et al. 2008a; 2008b; 2008c, Levick et al. 2008, Propst et al. 2008, Small et al. 2009, Perkin et al. 2010, Turner et al. 2010, Stefferud et al. 2011, Bogan and Boersma 2012, Gido and Propst 2012, Heard et al. 2012, Karatayev et al. 2012, Gido et al. 2013, Jones and Woods, eds., 2013, Rolls et al. 2013, Bogan et al. 2014a; 2014b, Boersma et al. 2014, Hubbs 2014, Jaeger et al. 2014, Miyazono 2014, Sandoval-Solis and McKinney 2014, Propst 2016). High-flow and low-flow pulse dynamics affect the ecological characteristics of the three large rivers by affecting three other critical ecological processes, floodplain inundation, alluvial water table dynamics, and fluvial-alluvial sediment dynamics.
- Outside of high-flow and low-flow events, the seasonal magnitude of baseflow and its interaction with river morphology determine the extent of the wetted perimeter and patterns of water depths and velocities in an unregulated river. Baseflow also affects water temperatures, and can also affect water quality when the natural source of the baseflow lies within deeper aquifers. All these factors affect the availability and suitability of river habitat for emergent vegetation, benthic invertebrates, amphibians, and fishes along unregulated rivers; and also affect the attractiveness of a river reach for visitation or use by reptiles and mammals outside of high-flow episodes (Deason 1998, Propst 1999, Schmidt et al. 2003, McFarland et al. 2004, Makings 2005, Stromberg et al. 2005, Witte 2005, Dudley and Platania 2007, Zymonas and Propst 2007, Bateman et al. 2008a; 2008b, Loughheed and Rodriguez 2008, Propst et al. 2008, Katz et al. 2009, Small et al. 2009, Perkin et al. 2010, Turner et al. 2010, Heard et al. 2012, Karatayev et al. 2012, Rogalski and Skelly 2012, Jones and Woods, eds., 2013, Rolls et al. 2013, Forstner et al. 2014, Hubbs 2014, Miyazono 2014, Sandoval-Solis and McKinney 2014).
- Fluvial biotic connectivity affects the taxonomic and genetic composition of the amphibian and, most particularly, the fish assemblage, by affecting the ability of these fauna to escape river reaches at risk of becoming isolated by interruptions in fluvial network connectivity, and/or their ability to subsequently return to reaches that were emptied or isolated by such interruptions (Cowley et al. 2003, Benda et al. 2004a; 2004b, Meyer et al. 2007, Fullerton et al. 2010, Propst 1999, Pringle 2003, Alò and Turner 2005, Dudley and Platania 2007, Zymonas and Propst 2007,

Levick et al. 2008, Hoagstrom et al. 2010, Perkin et al. 2010, Theobald et al. 2010, Turner and List 2007, Heard et al. 2012, Miyazono 2014, Pilger et al. 2015).

One critical environmental element, the fire regime, also directly affects one of the ecological outcomes discussed above: Fire through a large-river riparian corridor can significantly affect the vegetation and heat the surface water within the floodplain sufficiently to cause significant mortality among the biota in these floodplain water bodies. Uncharacteristic wildfire frequency and intensity in riparian corridors therefore can alter the frequency, intensity, and spatial extent of such mortality events (e.g., Gresswell 1999, Brown et al. 2001, Brunelle and Minckley 2002, Nagler et al. 2011, Luce et al. 2012, Whitney et al. 2015; 2016).

Another critical environmental element, channel reach-scale morphology, directly both affects and is affected by another ecological outcome, riparian vegetation, through effects on bank and floodplain soil stability. Riparian vegetation, including invasive species such as salt cedar, can stabilize floodplain soils so that the channel narrows or changes from multi-stranded to single-stranded, as has been observed along the Big Bend of the Rio Grande (Schmidt et al. 2003, Dean and Schmidt 2011) and elsewhere, often in conjunction with changes in river flow and sediment load. Conversely, changes in channel reach-scale morphology greatly disturb riparian vegetation, removing some (e.g., to clear land for agriculture or urban development) or resetting succession through disturbance; and the overall potential width of the floodplain – which determines the potential availability of land for riparian vegetation – depends on channel reach-scale morphology (see also Stromberg 1998, Shafroth et al. 2000, Price et al. 2005, Cornell et al. 2008, Levick et al. 2008, Doody et al. 2011, Cooper and Andersen 2012, Luce et al. 2012).

Finally, several critical ecological processes directly affect each other, as described in Appendix 1: Alluvial water table dynamics directly both affect and are affected by the baseflow regime; floodplain inundation directly affects alluvial water table dynamics, fluvial-alluvial water table dynamics, river water quality, and high-flow pulse dynamics; fluvial-alluvial sediment dynamics directly affect both the baseflow regime and stream meso-habitat dynamics; high-flow pulses directly affect alluvial water-table dynamics, floodplain inundation, and fluvial-alluvial sediment dynamics; low-flow pulse dynamics also affect alluvial water-table dynamics; and stream water quality dynamics both affect and are affected by aquatic primary productivity.

9.3.2 Climate Change

The final report for the Chihuahuan Desert REA will include a discussion of forecasts of the ways in which climate change will affect the ecoregion and its ecological resources. The present chapter presents only the conceptual model of the causal relationships for large river-floodplain systems that potentially will be affected.

Figure 9-3 presents the stressor model for the large river-floodplain systems in the U.S. portion of the ecoregion, simplified to show only those causal relationships that potentially will be affected by changes in the air temperature and precipitation and snowmelt regimes. Appendix 1 presents the rationale and citations for every causal link shown in the diagram.

Climate change will affect the three large river-floodplain systems in the U.S. portion of the ecoregion through its effects on two drivers in the large river-floodplain systems stressor model: the air temperature and the precipitation and snowmelt regimes within the U.S. portion of the ecoregion. Climate change will also affect the air temperature and the precipitation and snowmelt regimes across the watersheds of the Gila and Pecos Rivers and the Rio Grande that lie outside the U.S. portion of the ecoregion. Climate change across these upstream watersheds will affect the Gila and Pecos Rivers and the Rio Grande within the U.S. portion of the ecoregion indirectly: Changes in air temperature and precipitation will affect land use, water use, and the operations of upstream dams, diversions, and return flows across these upstream watersheds. In turn, these changes in upstream land use, water use, and dam, diversion, and return flow operations will affect three critical environmental variables in the large river-floodplain stressor model: upstream discharge regime, upstream fluvial network connectivity, and upstream water quality.

Changes in the air temperature and the precipitation and snowmelt regimes within the U.S. portion of the ecoregion may be termed “first-order” impacts of climate change. These first-order impacts may include changes in annual and seasonal averages, in the timing and magnitude of annual and seasonal extreme temperatures, and in the timing and magnitude of precipitation. The present discussion of these potential first-order impacts of climate change largely follows the discussion of first-order impacts of climate change to perennial streams, in Chapter 8. The discussions differ only in the inclusion here of potential first-order effects outside the U.S. portion of the ecoregion, recognized implicitly in the large river-floodplain stressor model.

Changes in the air temperature and precipitation and snowmelt regimes within the U.S. portion of the ecoregion will directly affect seven critical environmental elements within the U.S. portion of the ecoregion: the fire regime; watershed ground cover; runoff regime; watershed erosion; water impoundments, diversions, returns; runoff water quality; and watershed-scale groundwater dynamics. The air temperature regime also affects the precipitation and snowmelt regime within the U.S. portion of the ecoregion. Specifically, air temperature affects whether precipitation falls as rain or snow, whether precipitation even reaches the ground or evaporates as it falls (termed “virga” precipitation), and how much water runs off or infiltrates following precipitation versus simply evaporating.

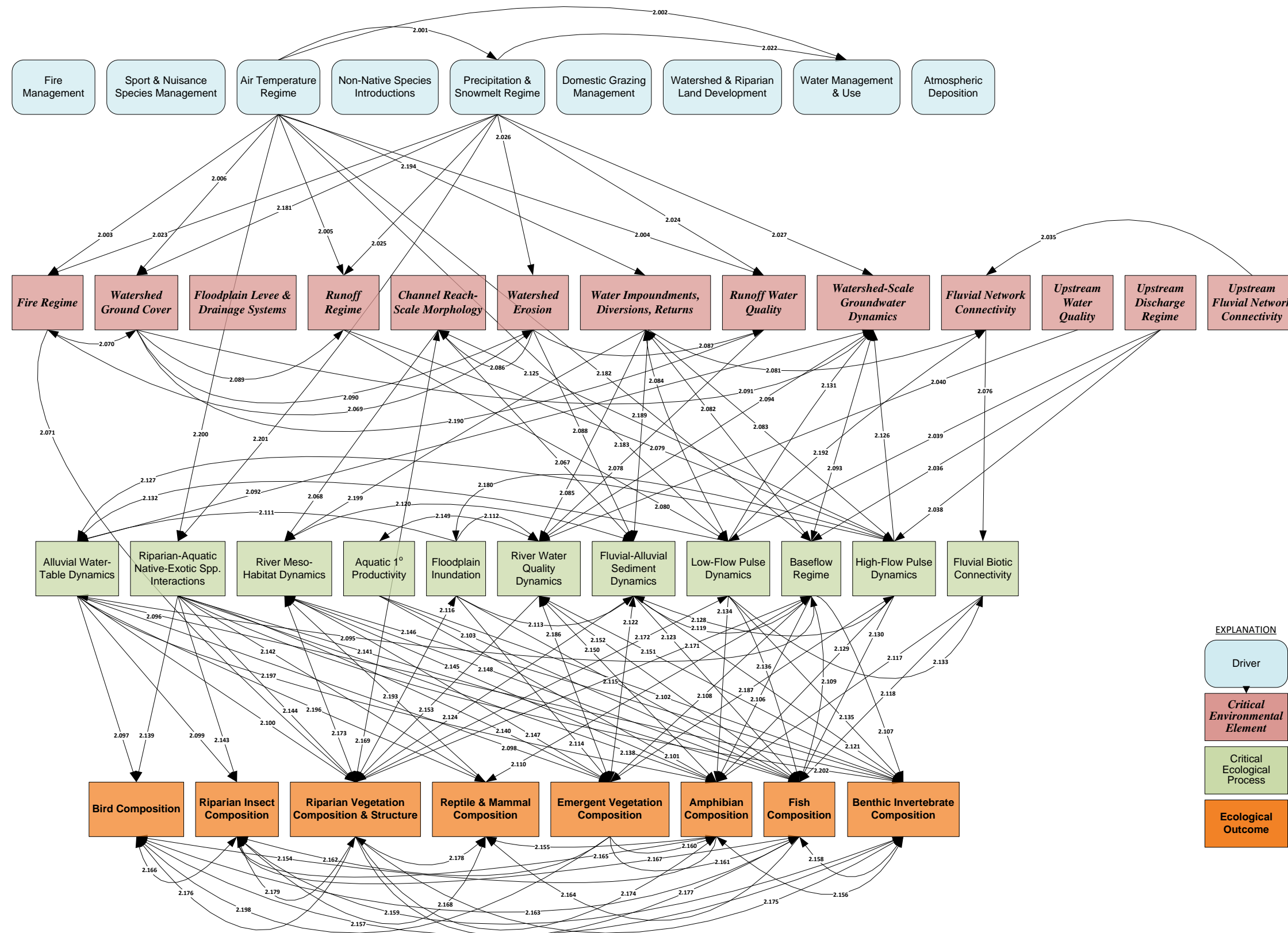
Changes in the air temperature and precipitation and snowmelt regimes within the U.S. portion of the ecoregion also will directly affect another driver within the U.S. portion of the ecoregion, water management and use. Specifically, changes in air temperatures and precipitation will affect annual and seasonal water supply and demand within the U.S. portion of the ecoregion. The resulting changes in water management and use within the U.S. portion of the ecoregion will have their own, further effects on the runoff regime, watershed erosion, and watershed-scale groundwater dynamics; on the operation of dams and diversions within the U.S. portion of the ecoregion; and so forth. These impacts would add to the impacts of climate change on the perennial streams within the U.S. portion of the ecoregion. The potential impacts of changes in water management and use are examined further, later in this chapter, in the discussion of the impacts of development.

The impacts of changes in the air temperature and precipitation regimes within the U.S. portion of the

ecoregion on the seven critical environmental elements noted above—fire regime; watershed ground cover; runoff regime; watershed erosion; water impoundments, diversions, returns; runoff water quality; and watershed-scale groundwater dynamics—may be termed “second-order” impacts of climate change. Examples of these second-order effects within the U.S. portion of the ecoregion include the following:

- Changes in air temperature and precipitation patterns will affect the probability of wildfires within the U.S. portion of the ecoregion directly along the river riparian corridors (see uncharacteristic wildfire, below) (D'Antonio and Vitousek 1992, Pyne et al. 1996, see Chapters 2-3 and 5-7).
- Changes in air temperature and precipitation patterns will affect watershed ground cover by affecting the types, density, and rates of mortality of upland vegetation across watersheds within the U.S. portion of the ecoregion (see Chapters 2-3 and 5-7).
- Changes in precipitation will affect the runoff regime within the U.S. portion of the ecoregion by altering the timing, amounts, forms, and rates of accumulation of the precipitation on watershed surfaces (Ward and Elliot, eds. 1995, Rango 2006, see Chapters 2-3).
- Changes in the intensity of rainfall events (e.g., the maximum rainfall rate within a storm) will affect the rate and spatial extent of soil erosion within the U.S. portion of the ecoregion caused by individual storm events.
- Changes in air temperature will affect the rate of evaporative losses from impoundment surfaces, which are substantial (Hogan 2013) and affect dam operations and releases.
- Changes in air temperatures and precipitation patterns will affect runoff water quality within the U.S. portion of the ecoregion by affecting water temperature—which affects other aspects of water quality – and the relative concentrations of soluble matter transported in the runoff. Changes in air temperature and precipitation patterns will also affect the rate at which salts accumulate across soil surfaces as a consequence of natural evaporative processes, and therefore the rates at which such salts are available for dissolution and transport in runoff, further affecting runoff water quality (Manahan 1991, see Chapters 2-3).

Figure 9-3. Chihuahuan desert large river-floodplain systems stressor model: Potential impacts of climate change.



- Recharge to regional groundwater systems both within and outside the U.S. portion of the ecoregion mostly takes place at higher elevations across the mountains and mountain fronts (foothills) of the ecoregion, and varies both with the amount of precipitation received and whether the precipitation occurs as rain or snow. Melting snow recharges more effectively than does rainfall. Changes in precipitation therefore will affect the spatial distribution and rates of recharge, which will affect watershed-scale groundwater dynamics within the U.S. portion of the ecoregion. However, flow-path lengths may be long, and changes in recharge may not affect groundwater discharge at springs or along rivers for decades to centuries (e.g., Scanlon et al. 2005, Serrat-Capdevila et al. 2007, Stonestrom et al., eds. 2007, Wolaver et al. 2008, Magruder et al. 2009, USBR 2011, Szykiewicz et al. 2012; 2015a; 2015b, Friggens et al. 2013a, Sheng 2013, Friggens and Woodlief 2014, Jaeger et al. 2014, Eng et al. 2016, Meixner et al. 2016, Sigstedt et al. 2016).

The second-order effects of changes in the air temperature and precipitation regimes within the U.S. portion of the ecoregion will also include direct impacts to three critical ecological processes:

- Changes in air temperature and precipitation will directly affect riparian-aquatic native-exotic species interactions within the U.S. portion of the ecoregion. Air temperature affects water demand in plants and thermal regulation in land animals, and native species may differ in their abilities to adjust to changes in air temperature patterns compared to non-native species. Similarly, precipitation directly along riparian corridors affects water availability for both plants and land animals along the corridors. Native species may differ in their abilities to adjust to changes in precipitation patterns compared to non-native species (e.g., Price et al. 2005, Enquist et al. 2008, Jones et al. 2010, Nagler et al. 2011, Friggens et al. 2013a; 2013b, Friggens and Woodlief 2014).
- Changes in air temperature will affect baseflow and low-flow pulse dynamics within the U.S. portion of the ecoregion by affecting the rates of evaporation of surface water and evapotranspiration by phreatophytes along the riparian corridors of the Gila and Pecos Rivers and the Rio Grande. Long-term changes to watershed-scale groundwater dynamics, as a result of changes in recharge (see above), also potentially could affect baseflow and low flows along some reaches (see below) (e.g., Scott et al. 2004; 2008, Price et al. 2005, Stromberg et al. 2006, Serrat-Capdevila et al. 2007, Hatler et al 2009, Kennedy and Gungl 2010, Friggens and Woodlief 2014). However, flow patterns along Pecos River and Rio Grande within the U.S. portion of the ecoregion depend almost entirely on the operations of dams, diversions, and return flows within and upstream from the U.S portion of the ecoregion, as discussed below (see Development, this chapter).

Changes in the air temperature and precipitation regimes outside the U.S. portion of the ecoregion similarly will directly affect the fire regime, watershed ground cover, runoff regime, watershed erosion, water impoundments, diversions, and returns, runoff water quality, and watershed-scale groundwater dynamics outside the ecoregion. The large river-floodplain systems stressor model does not include these distant causal relationships. Instead, the stressor model focuses on the consequences of these distant second-order effects on the upstream discharge regime, upstream fluvial network connectivity, and upstream water quality.

The second-order effects of climate change within the U.S. portion of the ecoregion, and on upstream discharge regime, upstream fluvial network connectivity, and upstream water quality, will have third-

order effects on each other, on critical ecological processes, and directly on some ecological outcomes; and these in turn will have effects on other critical ecological processes and ecological outcomes, through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above). For example (see Appendix 1 for full presentation):

- Changes in air temperature and precipitation patterns will affect the amount of runoff that flows into the three large rivers within and upstream from the U.S. portion of the ecoregion, both directly and from tributaries, and the amount of water recharged in the mountains and along mountain fronts to regional aquifers (USBR 2011). However, the changes in runoff and recharge will not directly affect large-river flows within U.S. portion of the ecoregion (see Chapters 2-3), for two reasons. First, changes in montane and mountain-front recharge will take decades to centuries to affect groundwater discharges along the rivers (see above). Second, the baseflow regimes and high- and low-flow pulse dynamics along the Pecos River and Rio Grande depend on the patterns of operations of dams, diversions, and return flows within their watersheds, as discussed below (see Development, this chapter, below). (Neither the Gila River nor any of its tributaries are dammed or diverted within the U.S. portion of the ecoregion). The impacts of climate change on runoff volumes and timing will affect the amount of water available for management along the Pecos River and Rio Grande. At the same time, changes in air temperature and precipitation patterns will affect water consumption (see Development, this chapter, below) and evaporative losses from impoundments (see above, this chapter). These several factors together will affect dam operations, diversions, and return flows. The close relationships among water availability, water demand, and the operations of dams, diversions, and return-flow systems – and the strong effects of dam, diversion, and return-flow operations (both within and upstream from the U.S. portion of the ecoregion) on discharge along the Pecos River and Rio Grande within the ecoregion—make it difficult to predict how climate change will affect the Pecos River and Rio Grande flow regimes within the ecoregion.
- Changes in watershed ground cover driven by climate change—both within and outside the U.S. portion of the ecoregion – will affect tributary sediment loads (see Chapter 8). However, the many dams along the Pecos River and Rio Grande within and outside the U.S. portion of the ecoregion trap essentially all the sediment delivered to their impoundments from upstream (e.g., Collins and Ferrari 2000a; 2000b, Ferrari 2013, Hogan 2013, IBWC 2013, Varyu and Fotherby 2015). As a result, changes in watershed ground cover driven by climate change will likely not affect the amount of sediment transported into the Pecos River and Rio Grande within the ecoregion. On the other hand, the changes in watershed ground cover driven by climate change both within and outside the U.S. portion of the ecoregion could affect other aspects of river water quality dynamics such as water temperatures and concentrations of dissolved and particulate organic matter (Allan 2004).

9.3.3 Uncharacteristic Wildfire

The fire regime within the U.S. portion of the Chihuahuan desert has changed as a result of the interaction of several drivers. Chapters 1 and 2 discuss the causes and consequences of uncharacteristic wildfire across the ecoregion in general, Chapters 4-6 discuss the causes and consequences of altered fire regimes for watershed ground cover, and Chapter 7 discusses the consequences of these changes for perennial streams within the U.S. portion of the ecoregion. As shown in Figure 9-4 four drivers directly affect the fire regime across the U.S. portion of the ecoregion in ways that, in turn, affect the three large river-floodplain systems: fire management, the air temperature regime, non-native species

introductions, and the precipitation regime. Three other drivers indirectly affect the fire regime across the U.S. portion of the ecoregion: domestic grazing management and watershed and riparian land development both directly fire management; and domestic grazing management also affects non-native species introductions, which in turn also affect fire management.

Figure 9-4 shows that watershed ground cover, a critical environmental element, also strongly affects the fire regime across the U.S. portion of the ecoregion (see Chapters 2-3 and 5-7). Watershed ground cover in turn is affected by numerous drivers. Chapters 5-7 provide detailed discussions of the drivers that shape watershed ground cover across the U.S. portion of the ecoregion and the ways in which watershed ground cover in turn affects wildfire and vice versa.

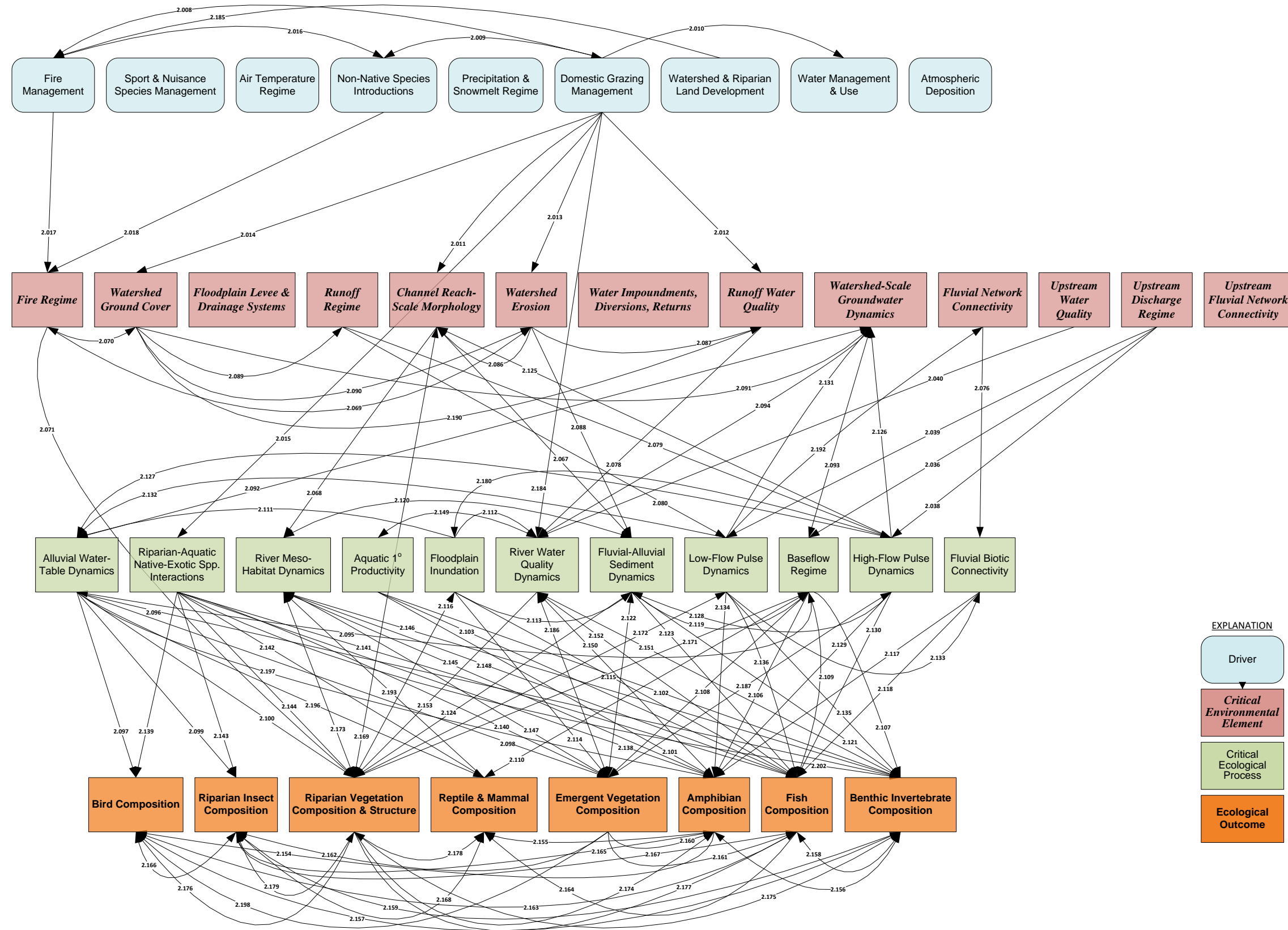
The stressor model assumes that the same four drivers also affect the fire regime across the watersheds of the Gila River, Pecos River, and Rio Grande outside the U.S. portion of the ecoregion. However, the stressor model does not display these additional interactions. Instead, the stressor model assumes that these latter interactions affect conditions within the U.S. portion of the ecoregion through their indirect effects on upstream water quality and upstream discharge regimes as a result of direct effects on the runoff regime and runoff water quality.

The fire regime, in turn, affects the three large rivers in the U.S. portion of the ecoregion at two scales: (1) indirectly through its effect on larger watershed dynamics that affect the rivers and their riparian corridors; and (2) directly through its impacts along the riparian corridors of the three rivers, as shown in the sub-model diagram, Figure 9-4. Appendix 1 presents the rationale and citations for each causal link shown in the diagram.

Uncharacteristic wildfire – i.e., ecologically significant alteration to the fire regime – at the watershed scale can alter (a) watershed ground cover, (b) the vulnerability of upland soils to erosion, and (c) the availability of burned and unburned particulate organic matter and soluble nutrients from ash for downhill transport in runoff (see Chapters 5-8). These changes in watershed characteristics in turn affect other critical environmental elements in the large river-floodplain stressor model at the watershed scale, including the runoff regime (which also affects infiltration and recharge), runoff water quality, and the transport of eroded sediment and organic matter downhill toward the large rivers or their tributaries, as noted above in the discussion of the potential impacts of climate change.

However, as also noted in the discussion of the potential impacts of climate change, the dams and diversions along the Pecos River and Rio Grande will significantly transform most watershed effects of uncharacteristic wildfire on runoff water quality and sediment transport. For example (see Appendix 1 for full presentation):

Figure 9-4. Chihuahuan desert large river-floodplain systems stressor model: Potential impacts of uncharacteristic wildfire and excessive domestic grazing.



- Alterations to the runoff regime as a result of uncharacteristic wildfire across watersheds can have little effect on the baseflow regime or the low-flow or high-flow pulse regimes along the Pecos River and Rio Grande. Instead, these components of the river flow regime will depend almost entirely on the patterns of operations of dams, diversions, and return flows within their watersheds, as discussed below (see Development, this chapter) (e.g., Theobald et al. 2010; Poff et al. 2011).
- Alterations to runoff water quality as a result of uncharacteristic wildfire across watersheds also can have little effect on river water quality or sediment transport (affecting fluvial-alluvial sediment dynamics) along the Pecos River and Rio Grande. Instead, these dynamics again will depend almost entirely on the patterns of operations of dams, diversions, and return flows within their watersheds, as discussed below (see Development, this chapter).

On the other hand, uncharacteristic wildfire at the watershed scale will also affect the frequency and intensity of wildfire directly along riparian corridors, where uncharacteristic wildfire can have significant ecological consequences on riparian vegetation dynamics (e.g., Gresswell 1999, BLM 2000, Brown et al. 2001, Brunelle and Minckley 2002, Stromberg et al. 2009a; 2009b, Theobald et al. 2010, Luce et al. 2012, Whitney et al. 2015; 2016).

9.3.4 Invasive Species

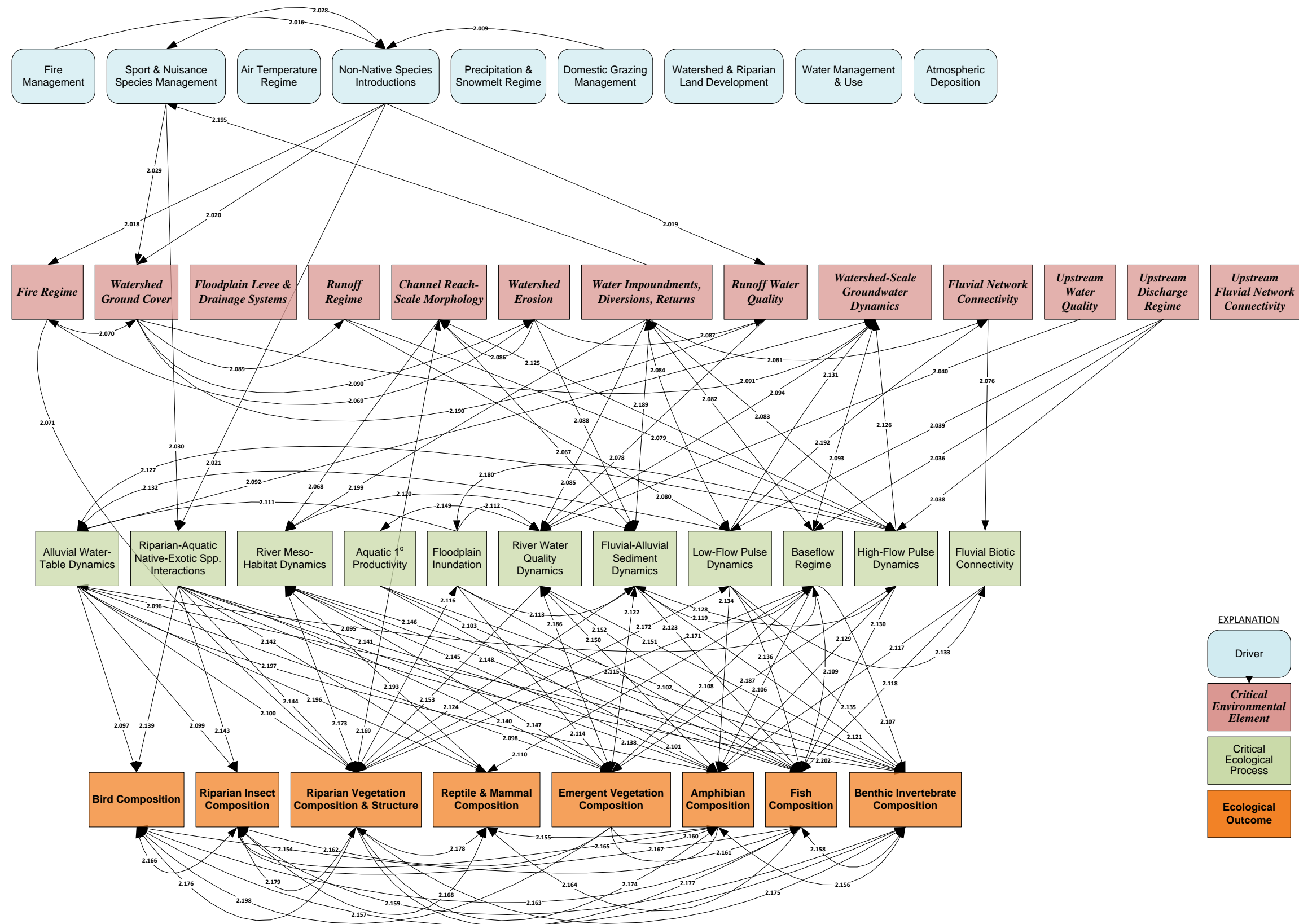
Chapters 2-3 discuss the ecological consequences of non-native species across the U.S. portion of the ecoregion in general. The large river-floodplain systems stressor model, Figure 9-5 includes two drivers that address the ways in which non-native species affect the three large river-floodplain systems in the U.S. portion of the ecoregion: non-native species introductions; and sport and nuisance species management. Appendix 1 presents the rationale and citations for each causal link shown in the diagram. The presentation here is essentially the same as the presentation on this Change Agent in Chapter 8 for the perennial stream systems.

Together, native species introductions and sport and nuisance species management significantly shape the ecological status of the large rivers across the ecoregion. The two affect each other and in turn are directly or indirectly shaped in part by two other drivers, fire management, and domestic grazing management, the effects of which are discussed separately above and below, respectively. Water management and use also indirectly affects sport and nuisance species management, as also discussed below (see Development): Decisions on impoundment management take into consideration needs for sport and nuisance species management.

Native species introductions and sport and nuisance species management affect the native aquatic and riparian fauna and flora of the ecoregion in part by indirectly affecting several critical environmental elements. The impacts to these critical environmental elements in turn affect several critical ecological processes. For example (see Appendix 1 for full presentation):

- Impoundments along the Pecos River and Rio Grande create large areas of lentic, warm-water aquatic habitat favorable to the persistence, growth, and reproduction of a large number of non-native aquatic and riparian species, including sport fish. As a result, the impoundments become reservoirs not only of water but of non-native biota, affecting riparian-aquatic native-exotic species interactions in the impoundments, along their shorelines, and in the rivers up- and down-stream (Shafroth et al. 2005, USBR 2009, Heard et al. 2012, IBWC 2013, Israël et al. 2014, NMDGF 2016).

Figure 9-5. Chihuahuan desert larger river-floodplain systems stressor model: Potential impacts of invasive species and landscape restoration.



- The introduction and spread of non-native dry-land and riparian vegetation across a watershed tributary to one of the large rivers can alter the fire regime of the watershed and the riparian corridors the river, as noted above and discussed in detail in Chapters 5-8. Non-native vegetation across a watershed can alter runoff rates, thereby potentially affecting flow patterns in a receiving stream or river. Non-native vegetation across a watershed can alter runoff water quality by altering soil salt buildup and the types of organic matter swept into the runoff, thereby potentially affecting stream water quality. Native and non-native vegetation across the watersheds also can have different abilities to anchor soils (see Chapters 5-7), altering watershed soil erosion dynamics, sediment inputs to perennial streams, and runoff water quality. However, as discussed above (see Uncharacteristic Wildfire, this chapter), alterations to runoff rates or water quality as a result of uncharacteristic wildfire across watersheds may have little effect on river flows, water quality, or sediment transport (affecting fluvial-alluvial sediment dynamics) along the Pecos River and Rio Grande. Instead, these dynamics depend almost entirely on the patterns of operations of dams, diversions, and return flows within the watersheds of these two rivers, as discussed below (see Development, this chapter).
- Vegetation across watersheds contributes dissolved and particulate organic matter to runoff. Alterations to watershed vegetation therefore may alter the amounts, decomposability, and chemistry of plant litter available to contribute to the allochthonous inputs to streams carried in watershed runoff (e.g., Vannote et al. 1980, Allan 1995; 2004, Hauer et al. 2002, Babler et al. 2011, Kominoski and Rosemond 2012). However, alterations to runoff transport of organic matter may have little effect on the aquatic food webs of the Pecos River and Rio Grande. Instead, the organic matter becomes trapped in impoundments along these two rivers and supports the aquatic food webs of these lakes instead (see Development, this chapter).

Native species introductions and sport and nuisance species management also affect the native aquatic and riparian fauna and flora of the three large river-floodplain systems in the U.S. portion of the ecoregion directly. The stressor model represents this direct connection in the form of a critical ecological process, riparian-aquatic native-exotic species interactions. For example (see Appendix 1 for full presentation):

- Non-native diseases of aquatic fauna, such as introduced fish parasites and the now-widespread chytridiomycosis, harm native fish and amphibians along the three large rivers in the ecoregion; and blooms of golden alga in their main channels and off-channel ponds (e.g., seasonal ponds on floodplains) have poisoned native amphibians and native fish (Propst 1999, Rosen and Caldwell 2004, Witte 2005, Zymonas and Propst 2007, Propst et al. 2009, USFWS 2009; 2010, Bean et al. 2010, Israël et al. 2014).
- Non-native aquatic fauna, including non-native sport and nuisance fishes, the non-native American bullfrog, non-native crayfish, and Asian clam (*Corbicula fluminea*) may compete with native aquatic fauna within the three large rivers and in surface waters across their floodplains for habitat space and materials, including food; prey on native aquatic fauna; or interbreed with them, destroying their genetic integrity. Stocking of sport fishes, including native species, also alters food web dynamics (Hubbs et al. 1977, Cowley and Sublette 1987, Rinne and Minckley 1991, Rosen et al. 1994; 2005, Fuller et al. 1999, Gido and Propst 1999; 2012, Propst 1999, Edwards et al. 2002; 2003, Propst and Gido 2004, Rosen and Caldwell 2004, Kapuscinski and Patronski 2005, Clarkson et al. 2005, Turner and List 2007, Gregory and Hatler 2008, Propst et al. 2008, Paroz et al. 2009, Hoagstrom et al. 2010, Pilger et al. 2010, Turner et al. 2010, Stefferud et al. 2011, Franssen et al. 2015, Heard et al. 2012, Martinez 2012, Gido et al. 2013, Moody and

Sabo 2013, Rolls et al. 2013, USFWS 2011; 2013, Hershler et al. 2014, McCluney and Sabo 2014, Miyazono 2014, Whitney et al. 2014, Hedden et al 2016).

- Non-native aquatic and riparian plant species may interact directly with native aquatic and riparian species, for example by outcompeting native species for space and water, as do Russian olive (*Elaeagnus angustifolia*), tamarisk (aka salt cedar) (*Tamarisk* spp.), giant reed (*Arundo donax*), and hydrilla (*Hydrilla verticillata*) (DeBano et al., eds. 1995, Bell et al. 1999, Krueper et al. 2003, Scott et al. 2004, Lite and Stromberg 2005, Makings 2005, Price et al. 2005, Rosen 2005, Bateman et al. 2008a; 2008b, Cornell et al. 2008, Levick et al. 2008, Stromberg et al. 2009a; 2009b, Dean and Schmidt 2011, Doody et al. 2011, Nagler et al. 2011; 2012, Fain et al. 2014).
- The non-native nutria (*Myocastor coypus*), a semi-aquatic rodent, alters riverine aquatic and shoreline habitat and vegetation, affecting habitat and food resources for native fauna and affecting both native and non-native emergent vegetation (Milholland et al. 2010).

These changes to critical ecological processes and ecological outcomes in turn have cascading effects on other critical ecological processes and ecological outcomes in the large river-floodplain systems of the ecoregion, as explained in the section on Critical Ecological Processes and Ecological Outcomes, above (this chapter, above).

9.3.5 Development

Chapters 2-3 discuss the ecological consequences of land and water development across the U.S. portion of the ecoregion in general. The large river-floodplain systems stressor model includes two drivers that address the impacts of development within the U.S. portion of the ecoregion: (1) watershed and riparian land development, and (2) water management and use. Figure 9-6 shows the causal relationships through which these two drivers directly or indirectly affect every critical environmental element represented in the large river-floodplain systems stressor model, changes in which will affect almost every critical ecological process and ecological outcome. Figure 9-6 also shows that two other drivers, the air temperature regime and the precipitation and snowmelt regime, directly affect water management and use, as discussed above (see climate change, this chapter). Appendix 1 presents the rationale and citations for each causal link shown in Figure 9-6.

Development across the watersheds of the Pecos River and Rio Grande and their tributaries – and the absence of comparable development across the watershed of the Gila River – outside the U.S. portion of the ecoregion also affect conditions within the U.S. portion of the ecoregion. Water management and use across the upstream watersheds includes the operations of dams, diversions, and return flows in the upstream watersheds of the Pecos River, Rio Grande, and Rio Conchos (Kelly 2001; Hogan 2013; IBWC 2013). The large river-floodplain systems stressor model does not include these neighboring causal relationships. Instead, the stressor model focuses on the consequences of these relationships for upstream discharge, upstream fluvial network connectivity, and upstream water quality. These consequences arise through the same interactions, through which watershed and riparian land development and water management and use within the U.S. portion of the ecoregion affect river hydrology, river water quality, and fluvial network connectivity within the U.S. portion of the ecoregion. Figure 9-6 shows the causal relationships through which the three upstream critical environmental elements affect other critical environmental elements as well as critical ecological processes and

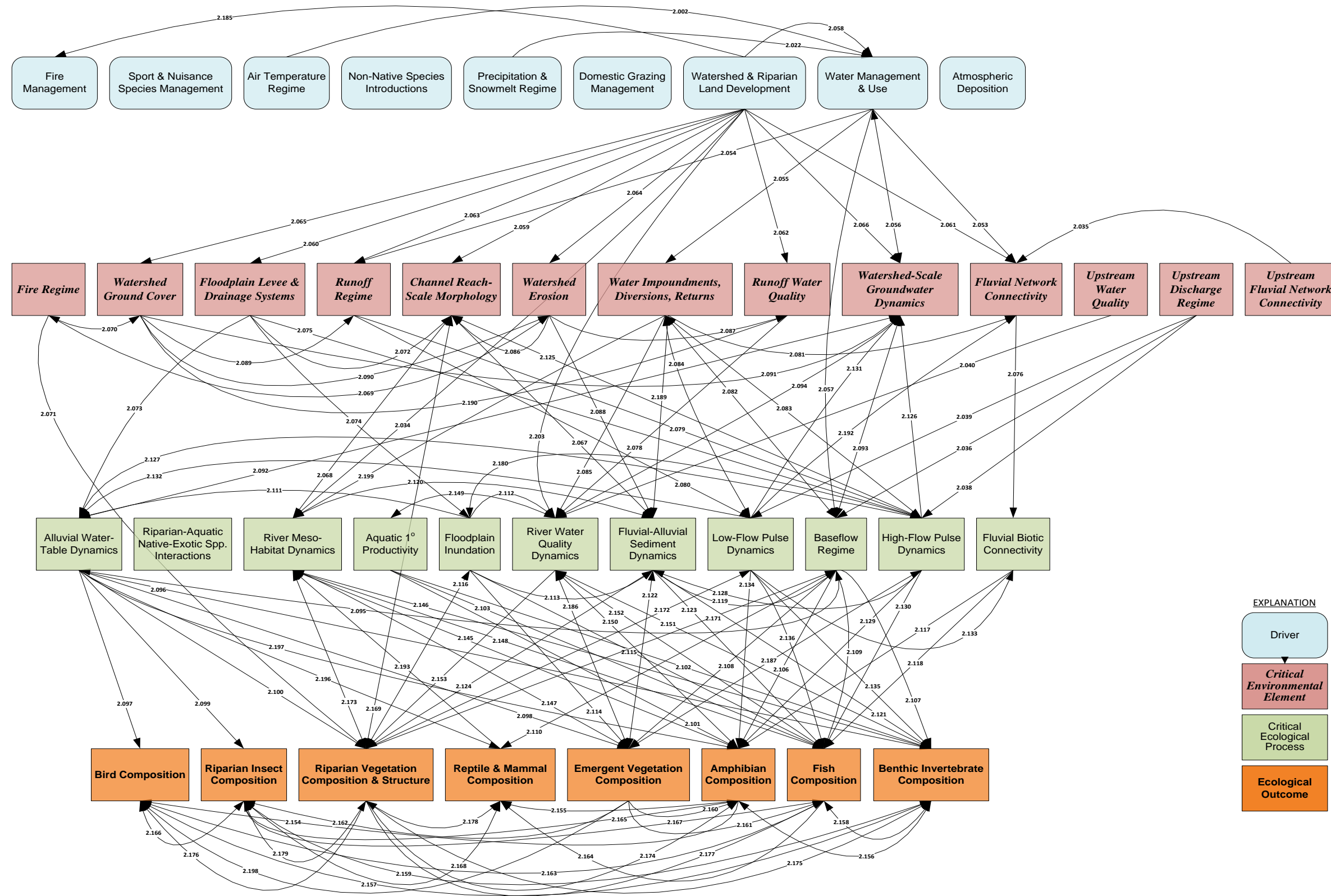
ecological outcomes along the three large river-floodplain systems in the U.S. portion of the ecoregion. Appendix 1 presents the rationale and citations for these causal relationships as well.

Watershed and riparian land development within the U.S. portion of the ecoregion directly affects water management and use: Many if not most forms of land development in the ecoregion require a water supply to sustain the activities associated with the development. Watershed and riparian land development also affects another driver within the U.S. portion of the ecoregion, fire management. Fire management decisions also must take into account the types and locations of developed land, as these evolve across the landscape (see Uncharacteristic Wildfire, above). Water management and use, in turn, indirectly affects another driver, sport and nuisance species management within the U.S. portion of the ecoregion, by affecting the availability of water in stream reaches and in impoundments suitable for sport and nuisance species. Chapter 8 provides detailed presentations on all these causal relationships.

Watershed and riparian land development within the U.S. portion of the ecoregion directly affects eight critical environmental elements in the large river-floodplain systems stressor model: watershed ground cover; floodplain levee and drainage systems; runoff regime; channel reach-scale morphology; watershed erosion; runoff water quality; watershed-scale groundwater dynamics; and fluvial network connectivity (see Appendix 1 and Chapters 2-3 and 5-8). In turn, water management and use within the U.S. portion of the ecoregion directly affects four critical environmental elements in the large river-floodplain systems stressor model: runoff regime; water impoundments, diversions, returns; watershed groundwater dynamics; and fluvial network connectivity (see Appendix 1 and Chapters 2, 3, and 8). The impacts of water management and use thus compound the effects of watershed and riparian land development on three critical environmental elements within the U.S. portion of the ecoregion: runoff regime; watershed groundwater dynamics; and fluvial network connectivity.

The effects of land development across watersheds on the runoff regime and watershed erosion do not translate into effects on river hydrology and sediment dynamics. As noted above, this chapter, in the discussion concerning the potential impacts of climate change and uncharacteristic wildfire, the baseflow regime and high- and low-flow pulse dynamics of the Pecos River and Rio Grande within the U.S. portion of the ecoregion instead depend almost entirely on the patterns of operations of dams, diversions (including from groundwater wells), and return flow systems along these two rivers and upstream within their watersheds, including the Rio Conchos watershed in Mexico. The storage and release of water from dams in the Rio Grande basin overall are highly regulated, including by interstate agreements between New Mexico and Texas and bi-national agreements between the U.S. and Mexico (Kelly 2001, Hogan 2013, IBWC 2013, NMOSE 2013, TWDB 2016). Further, the dams trap essentially all sediment that may wash off their watersheds (e.g., Collins and Ferrari 2000a; 2000b, Ferrari 2013, Hogan 2013, IBWC 2013), preventing this sediment from contributing to fluvial-alluvial sediment dynamics downstream. The section below on the impacts of water impoundments, diversions, and returns, and the impacts of watershed-scale groundwater dynamics, discuss these effects on river hydrology and sediment dynamics in greater detail.

Figure 9-6. Chihuahuan desert large river-floodplain systems stressor model: Impacts of land and water development.



Watershed and riparian land development and water management and use nevertheless substantially affect the large river-floodplain systems in the U.S. portion of the ecoregion, at least along the Pecos River and Rio Grande. (The Gila River and its watershed are largely undeveloped, both within the ecoregion and upstream). Examples of the first-order effects of watershed and riparian land development and water management and use on critical environmental elements include the following (see Appendix 1 for full presentation):

- Land development on floodplains converts meso-habitat features to human use, altering the distribution of these features and/or eliminating or severely reducing habitat function. Additionally, as has occurred throughout the world, land development alongside the Rio Grande and Pecos Rivers and efforts to prevent or reduce flooding to the developed lands have confined the channels of these two rivers within artificial channel structures, or artificially “hardened” channel banks to prevent flows from reshaping their morphology, even in reaches not affected by levees (see below) (Tockner and Stanford 2002, NMDGF 2006, Lougheed and Rodriguez 2008, Parcher et al. 2010, Theobald et al. 2010, Poff et al. 2011, Hogan 2013, see Chapters 2-3).
- Land development across the watersheds and floodplains of the Pecos River and Rio Grande within and upstream from the U.S. portion of the ecoregion affects runoff water quality within the U.S. portion of the ecoregion – another pattern associated with land development worldwide. Developed lands produce point-source and diffuse surface water pollution in watershed runoff, unless the development includes effective runoff pollution control systems (Bexfield and Anderholm 1997, Levings et al. 1998, Kelly 2001, Allan 2004, USEPA 2005, Cornell et al. 2008, Gregory and Hatler 2008, Levick et al. 2008, Theobald et al. 2010, Poff et al. 2011, Hogan 2013, IBWC 2013, NMOSE 2013, Gutiérrez and Johnson 2014, Borderplex Alliance 2016).
- Land development across the watersheds and floodplains of the Pecos River and Rio Grande within the U.S. portion of the ecoregion sometimes involves the construction of road and railroad crossings over streams or the burying of stream courses in underground culverts that can reduce fluvial network connectivity (Diebel et al. 2015, Fuller et al. 2015).
- Land development on the floodplains of all three large rivers within the U.S. portion of the ecoregion – as is often the case with large alluvial river floodplains – has led to the construction of levees to control or prevent flooding. Land development on the floodplains of the Rio Grande and Pecos River within the U.S. portion of the ecoregion has also involved the construction of drainage systems to shrink or eliminate wetlands and lower water tables to allow greater human use of the floodplains (Cowley et al. 2003, Schmidt et al. 2003, Wittler et al. 2004, Lougheed and Rodriguez 2008, Pease et al. 2006, Cody and Carter 2008, Dean and Schmidt 2011, USBR 2012, Hogan 2013, IBWC 2013, Magaña 2013, NMOSE 2013; 2016a; 2016b, Sandoval-Solis and McKinney 2014, Borderplex Alliance 2016, TWDB 2016).
- Water management and use in the ecoregion have long involved the construction and operation of dams, diversions, and return-flow systems, as discussed in Chapter 2. The dams eliminate riverine and floodplain habitat within their area covered by their impoundments. In turn, floodplain development has both stimulated and been stimulated by the diversions and return flows (Kelly 2001, Hogan 2013, NMOSE 2013).
- Water management and use in the ecoregion also have long involved the construction and operation of groundwater wells. Local and state water management policies and practices and both interstate and bi-national agreements determine how much water is consumed from the aquifers of the ecoregion and where and when it is withdrawn (e.g., George et al. 2011, Hogan 2013, NMOSE 2013, TWDB 2016, see also Chapters 2-3).

The first-order effects of development, both within and upstream from the U.S. portions of the ecoregion, have numerous significant second-order effects on critical ecological processes within the U.S. portion of the ecoregion. For example (see Appendix 1 for full presentation):

- Dams fragment the fluvial network. The impacts of individual dams on fluvial network connectivity depend on the location, design, and operation of each dam. Design factors that affect the impacts of individual dams on fluvial network connectivity include dam height, overflow rates and frequencies, the presence/absence and design of hydropower turbines or diversion structures, and the presence/absence and design of accommodations for fish passage (Pringle 2003, Fullerton et al. 2010, Hogan 2013, Fuller et al. 2015, see also Chapters 2-3).
- The operations of the dams, diversions, and return-flow systems together significantly alter all aspects of river discharge along the Pecos River and Rio Grande within the U.S. portion of the ecoregion. The dams on these rivers and some of their tributaries store and release water to accomplish specific water management objectives, which may include flood control, hydropower generation, and/or water supply for crop irrigation, livestock watering, municipal and industrial consumption. The natural precipitation regime of the ecoregion, as also discussed in Chapter 2, produces a large pulse of runoff in the winter and early spring, and smaller, more variable pulses in the late summer and early fall. The dams in the ecoregion store this water for flood control and for use on a different time-table determined by the demands for both water and electricity and by interstate and bi-national agreements. Further, impoundments, crop irrigation, livestock watering, and municipal and industrial users consume (i.e., lose to evaporation) large fractions of the water they receive, retaining or returning only modest fractions back to the surface water or groundwater systems. Dam releases, diversions, and return flows along the Pecos River and the Rio Grande within the U.S. portion of the ecoregion and upstream therefore create baseflow regimes and high- and low-flow pulse dynamics along these two rivers that differ greatly from what these rivers would experience in the absence of regulation—as has happened to numerous highly regulated rivers in arid climates worldwide (Williams and Wolman 1984, Collier et al. 1996, Graf 1999; 2006, Smakhtin 2001, Poff and Hart 2002, Richter and Thomas 2007, Fitzhugh and Vogel 2011, Hogan 2013).
- As mentioned in Chapter 2, above, the Gila River presently has no dams along its mainstem, and diversions along the mainstem within the U.S. portion of the ecoregion deliver water only to local users. However, efforts are ongoing to permit construction of a large diversion facility somewhere along the mainstem immediately upstream from the present analysis extent, under the terms of the New Mexico-Arizona Water Settlement Act of 2004 (New Mexico Interstate Stream Commission 2017). Such a diversion clearly would reduce annual discharge along the Gila River within the U.S. portion of the ecoregion. However, the ways in which such a diversion would be constructed and operated remain unresolved, making it difficult to estimate the specific ways in which the diversion would affect river discharge or connectivity.
- Alterations to the low-flow pulse regimes along the Rio Grande include the disappearance of flow entirely along some reaches below Elephant Butte dam and downstream from El Paso/Ciudad Juarez. These reaches are naturally losing reaches, for geologic reasons, but the frequency, spatial extent, and duration of disappearance have all increased with flow regulation, decreasing fluvial network connectivity (Hogan 2013, Sabo 2014, Murphy et al. 2015).
- Water impoundments, diversions, and return flows directly affect river water quality in numerous ways. (1) Impoundments on large rivers are unique aquatic environments with their own hydrologic, chemical, and ecological dynamics. They trap incoming sediment and organic matter, and create a lacustrine environment with its own food web. Evaporation from

impoundment surfaces can raise concentrations of dissolved constituents in the remaining water. They also become thermally stratified, with the colder bottom layer (hypolimnion) becoming anoxic and able to dissolve some metal ions that are insoluble in oxygenated water. Dams may release impounded water from the warmer epilimnion (upper layer), the hypolimnion, or some combination of the two. The released water therefore typically is thermally highly altered from natural conditions, carries very little or no sediment, contains elevated salt concentrations, may contain little dissolved oxygen, and may contain dissolved chemicals, including pollutants, shaped by both the inflows to the impoundment and the chemical and ecological processes of the impoundment itself. (2) The water quality of return flows to a river is shaped by the ways in which the water was used. Irrigation drainage contains farm chemicals (nutrients, biocides) and salts leached from the irrigated soils. Municipal wastewaters contain residual concentrations of numerous contaminants from nutrients to pharmaceuticals that remain in the water following treatment, and in this region also contain elevated salt concentrations. Municipal storm-water outflows may contain untreated wastewater in addition to non-point-source pollution. Industrial wastewaters also contain residual concentrations of industrial wastes that remain following treatment, and may have elevated salt concentrations; accidental spills also can occur (Baxter 1977, Bexfield and Anderholm 1997, Kelly 2001, Cooke et al. 2005, Miyamoto et al. 2007, Cornell et al. 2008, Porter et al. 2009, Basnet et al. 2013, Hogan 2013, IBWC 2013, Gutiérrez and Johnson 2014, Sandoval-Solis and McKinney 2014, Borderplex Alliance 2016).

- The dams on the Pecos River and the Rio Grande within the U.S. portion of the ecoregion and upstream, including on the Rio Conchos, trap essentially all sediment that washes off their watersheds and floodplains or erodes from the river channel downstream from the next-upstream dam (e.g., Collins and Ferrari 2000a; 2000b, Ferrari 2013, Hogan 2013, IBWC 2013). This system of dams prevents upstream sediment from contributing to fluvial-alluvial sediment dynamics downstream and alters the capacity of the river to erode its own channel and banks – another effect common to dams worldwide (Graf 2005; 2006, Kondolf 1997, Brandt 2000, Collins and Ferrari 2000a; 2000b, Kelly 2001, Grant et al. 2003, Schmidt et al. 2003, Cornell et al. 2008, Schmidt and Wilcock 2008, Arthington et al. 2010, Kinzli and Myrick 2010, Theobald et al. 2010, USFWS 2010, Poff et al. 2011, Sabo et al. 2012, Ferrari 2013, Hogan 2013, IBWC 2013, Varyu and Fotherby 2015, Wohl et al. 2015).
- Reservoirs and their shorelines may provide homes for non-native aquatic and riparian species, as discussed above (see Invasive Species, above, this chapter).

These changes in critical ecological processes for the large river-floodplain systems in turn affect numerous ecological outcomes through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above).

9.3.6 Excessive Domestic Grazing

Chapters 2-3 discuss the ecological consequences of excessive domestic grazing across the ecoregion in general, Chapters 5-7 discuss the causes and consequences of excessive domestic grazing for watershed ground cover, and Chapter 8 discusses the effects of excessive domestic grazing on both watershed and riparian ground cover, and the consequences of these impacts on perennial streams within the U.S. portion of the ecoregion. As shown in Figure 9-4, domestic grazing management in the U.S. portion of the ecoregion affects water and fire management practices. Ranchers across the ecoregion provide water for their livestock by damming intermittent streams to form small ponds, diverting perennial

streams to watering structures, and drilling small wells to supply watering tanks. As also shown in Figure 9-4, domestic grazing management in the U.S. portion of the ecoregion both affects and is affected by the spread of non-native vegetation, as discussed in detail in Chapter 5-7. For example, grazed livestock act as vectors for spreading non-native grasses.

The stressor model assumes that excessive domestic grazing affects both watershed and riparian ground cover across the watersheds of the Gila River, Pecos River, and Rio Grande outside the U.S. portion of the ecoregion. However, the stressor model does not display these additional interactions. Instead, the stressor model assumes that these latter interactions affect conditions within the U.S. portion of the ecoregion through their indirect effects on upstream water quality and upstream discharge regimes as a result of direct effects on the runoff regime and runoff water quality.

The stressor model for large river-floodplain systems in turn proposes that excessive domestic grazing affects the three large rivers in the U.S. portion of the ecoregion at two scales: (1) indirectly through its effect on watershed dynamics that affect the rivers and their riparian corridors; and (2) directly through its impacts along the riparian corridors of the three rivers, as shown in the sub-model diagram, Figure 9-4. Appendix 1 presents the rationale and citations for each causal link shown in the diagram.

At the watershed scale, excessive domestic grazing alters (a) watershed ground cover, (b) the vulnerability of upland soils to erosion, and (c) runoff water quality – the latter effect arising from animal liquid and solid wastes. These changes in watershed characteristics in turn affect other critical environmental elements at the watershed scale, including the runoff regime (which also affects infiltration and recharge), runoff water quality, and the transport of eroded sediment downhill toward stream channels. In combination, these changes in watershed dynamics affect numerous critical ecological processes and their ecological outcomes in the perennial stream systems, as noted above in the discussion of the potential impacts of climate change.

However, as also noted in the discussion of the potential impacts of climate change and development, the dams, diversions, and return flows along the Pecos River and Rio Grande within and upstream from the U.S. portion of the ecoregion significantly transform the effects of most changes in watershed characteristics. Specifically, the dams, diversions, and return flows completely reshape the river flow regime, patterns of sediment storage, erosion, and transport, and water quality. These transformations effectively mask all watershed-scale effects of excessive domestic grazing on the two rivers.

Excessive domestic grazing along riparian corridors, on the other hand, can affect the large river-floodplain systems of the ecoregion. As discussed also for perennial streams (Chapter 8), livestock may feed on native and non-native riparian vegetation; rest under riparian canopy cover; trample shorelines, native biota, and animal burrows while moving in/out of river and floodplain waters; and compete with native herbivores. Excessive trampling by livestock can destabilize river banks and channel substrates, thereby directly altering channel reach morphology and consequently river meso-habitat dynamics. And livestock that graze, rest, and move in and along rivers and on their floodplains deposit their wastes directly alongside and into river and floodplain surface waters, directly altering water quality (Schmidly and Ditton 1978, Medina and Martin 1988, El-Hage and Moulton 1998, Belsky et al. 1999, Propst 1999,

BLM 2000, Krueper et al. 2003, Gordon and Meentemeyer 2006, Wohl 2006, Levick et al. 2008, Lucas et al. 2009, Herbst and Cooper 2010, Theobald et al. 2010, Poff et al. 2011, USFWS 2012, Garrett and Edwards 2014, Cole and Cole 2015, see Chapters 2-3).

These changes in critical environmental elements and critical ecological processes for the large river-floodplain systems in turn affect numerous ecological outcomes through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above).

9.3.7 Landscape Restoration

Chapter 3 discusses the types of landscape restoration projects taking place in the U.S. portion of the ecoregion, some of which currently or in the future could affect the large river-floodplain systems. Ongoing and potential landscape (i.e., habitat) restoration efforts along the three large rivers include efforts to remove non-native aquatic and riparian species, and to ensure the replacement of removed non-native vegetation with native vegetation. Specifically such efforts include or potentially could include the following (see Figure 9-5 and Appendix 1):

- Restoration of river habitat through the control or removal of non-native fishes, removal or improvement of floodplain road-crossing barriers, exclusion of livestock, and re-introduction of beaver and locally extirpated fish (e.g., Kapuscinski and Patronski 2005, NMDGF 2006, Dudley and Platania 2007, Hoagstrom et al. 2008, USFWS 2009; 2010; 2012, Wild 2011, Januchowski-Hartley et al. 2013; 2014, Gibson and Olden 2014, Hershler et al. 2014, Pilger et al. 2015). (Captive breeding and re-introduction of native species is not considered a type of landscape or habitat restoration, and so is not included here).
- Restoration of riparian and wetland habitat on floodplains through the restoration of critical hydrologic dynamics such as flooding, exclusion of livestock, and/or removal of non-native vegetation, with or without active restoration of native vegetation in its place (Farley et al. 1994, Stromberg 1998; 2001, Belsky et al. 1999, BLM 2000, Krueper et al. 2003, Bateman et al. 2008a; 2008b; 2008c; 2009, Malcom and Radke 2008, Shafroth et al. 2008, Abelho and Molles 2009, Katz et al. 2009, Smith et al. 2009, Stromberg et al. 2009a; 2009b, Theobald et al. 2010, Nagler et al. 2011, Poff et al. 2011, Watts and Moore 2011, Brand et al. 2013, Reynolds et al. 2014, Cole and Cole 2015, Mosher and Bateman 2016). Non-native species targeted for control or removal include salt cedar (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*), *Arundo donax*, an invasive reed, and Phragmites (*Phragmites* spp.), another invasive reed. Removal methods may include biological control, prescribed fire, mechanical removal, and chemical control, and these methods can have their own effects on riparian biota (e.g., Nagler et al. 2012, Goolsby et al. 2016).

9.4 Large River-Floodplain System Key Ecological Attributes

As noted earlier, all ecological outcomes and critical ecological processes in a system stressor model constitute key ecological attributes for the system. The list below identifies 19 key ecological attributes for the Chihuahuan Desert large river-floodplain system based on these criteria. *Characterizing the present condition of a system requires data on indicators for its key ecological attributes.* The definitions for the key ecological attributes are the same as the definitions for these components in the stressor model presented above.

- **Ecological Outcomes**
 - Amphibian Composition
 - Benthic Invertebrate Composition
 - Bird Composition
 - Emergent Vegetation Composition
 - Fish Composition
 - Reptile & Mammal Composition
 - Riparian Insect Composition
 - Riparian Vegetation Composition & Structure
- **Critical Ecological Processes**
 - Alluvial Water-Table Dynamics
 - Aquatic Primary (1°) Productivity
 - Baseflow Regime
 - Floodplain Inundation
 - Fluvial Biotic Connectivity
 - Fluvial-Alluvial Sediment Dynamics
 - High-Flow Pulse Dynamics
 - Low-Flow Pulse Dynamics
 - Riparian-Aquatic Native-Exotic Spp. Interactions
 - River Meso-Habitat Dynamics
 - River Water Chemistry Dynamics

10 Springs-Emergent Wetlands Conceptual Model

This chapter presents the conceptual ecological model for the Springs-Emergent Wetlands CE. As discussed in Chapter 3, springs and emergent wetlands in the U.S. portion of the ecoregion are known by a variety of names, including cenote, ciénega, marsh, spring, and spring complex, and occur across a wide range of elevations and topographic settings, for river floodplains to mountain canyons. The water that emerges – seasonally or perennially – at a spring-emergent wetland site may consist of groundwater from alluvial or other shallow aquifers, montane bedrock, basin-fill, and larger-scale or regional aquifers. The water therefore may originate as recharge nearby or elsewhere and may spend years, decades, or longer moving through the groundwater system before re-emerging. As a result, each spring-emergent wetland site has a distinct pattern of variation in discharge or water level, and a distinct pattern of temperature controlled by the temperatures in the aquifers through which the water has passed, some of which may be affected by geothermal activity. Each spring-emergent wetland site is also distinct in its concentrations of dissolved matter, controlled by the chemistry of the groundwater pathways along which its water has traveled.

The groundwater emerging at a spring may constitute the dominant source(s) of a lowland-headwater perennial stream, which connects its source springs to the larger surface water network of the ecoregion. However, other spring-emergent wetland sites in the ecoregion may only infrequently become connected to the larger surface water network of the region. Such connections (over both historic and geologic time scales) allow aquatic species to move among spring-emergent wetland sites or between springs and streams: the more isolated a site, the more likely the site will come to harbor unique, endemic species.

Spring-emergent wetland sites in the ecoregion rarely experience short-term hydrologic disturbance from runoff. However, springs with limited, local groundwater catchments and short groundwater flow paths have fluctuating discharges driven by variation in precipitation (e.g., Porter et al. 2009). River flooding downstream from a spring outlet also can cause water to back up into a spring. Other natural disturbances include fire in the emergent or surrounding vegetation, which may originate in surrounding uplands. Data on potential criteria (e.g., hydrogeology, morphology, discharge rates, chemistry, fauna, and flora) for distinguishing spring-emergent wetland sub-types are available for only a few sites across the ecoregion. Consequently, the present assessment does not distinguish any sub-types.

The model for the conceptual ecological model for the Springs-Emergent Wetlands CE shares several features in common with the perennial streams conceptual model (Chapter 8). Many lowland-headwater perennial streams in the ecoregion originate at discrete springs or spring complexes, and the fluvial and biotic connections between the two system types can significantly affect ecological conditions at the originating springs. Several fish species occur in both system types, as discussed in Chapter 8 in the section on “Fishes of the U.S. Portion of the Chihuahuan Desert Ecoregion.” Nevertheless, it is important ecologically to distinguish springs-emergent wetlands from lowland-headwater perennial streams in the ecoregion for purposes of this REA.

There are three main ecological reasons to distinguish the springs-emergent wetlands CE from the

lowland-headwater perennial streams CE. First, some lowland-headwater perennial streams do not originate at springs, and some springs-emergent wetlands do not feed lowland-headwater perennial streams. Second, the hydrologic, chemical, and thermal conditions immediately within the orifice and pool of a spring differ from those that develop after the water becomes part of a flowing surface stream. These conditions result in the formation of distinctive spring communities of aquatic organisms, including unique endemic fishes and invertebrates (see below). Third, the hydrologic, chemical, thermal, and topographic conditions immediately around a spring pool may support shoreline and emergent wetland vegetation that differs from what may exist downstream along the banks of the flowing stream.

Despite the need to distinguish springs-emergent wetlands from lowland-headwater perennial streams in the ecoregion, it sometimes may be difficult to separate the two in practice. In some cases, springs discharge directly into the sides or bottoms of streams, such as along the Black River, New Mexico (Cowley and Sublette 1987). In these and other cases, further, the outflow of one or more springs may be so large that it is difficult to draw a hard line between “spring outflow” and “perennial stream,” such as with Chandler and Caroline Springs, the discharges of which form Independence Creek, Texas. Even in such cases, however, it is still possible to distinguish the geochemical environment of the spring orifice and its immediate vicinity from the surface flows created or supplemented by the spring discharge.

Examples of springs and spring complexes in the ecoregion include:

- Numerous geothermal springs along the East, Middle, and West Forks of the Gila River in New Mexico (e.g., NMDGF 2006, Paroz et al. 2009).
- Mimbres Spring in the closed Mimbres River valley/Guzmán Basin (NMDGF 2006).
- Bitter Lake National Wildlife Refuge, a complex of springs and sinkholes across the Pecos River floodplain near Roswell, New Mexico.
- Geyser, Washington, Rattlesnake, Blue, and Castle Springs, discrete sources of Black River, New Mexico, lowland-headwater perennial stream (see above).
- Chandler and Caroline Springs, the sources of Independence Creek, Texas, a lowland-headwater perennial stream (see above).
- Diamond-Y Spring, the source of Diamond-Y Draw, Texas, lowland-headwater perennial stream (see above).
- Comanche Springs, in Stockton, Texas.
- The San Solomon Springs complex, near Balmorhea, Texas.
- Cattail Spring in Big Bend National Park (e.g., Walsh et al. 2014).

The presentation of the conceptual ecological model for the Springs-Emergent Wetlands CE follows the structure described in Chapter 4.

10.1 Sources of Information

The Springs-Emergent Wetlands system control and stressor models integrate information from several sources:

- (1) The Springs ecosystem control model presented in Miller et al. (2010).

- (2) The conceptual model for ciénega, marsh, and pond conservation elements developed for the Madrean Archipelago rapid ecoregional assessment immediately to the west of the Chihuahuan desert ecoregion (Crist et al. 2014).
- (3) Type descriptions (Comer et al. 2003, NatureServe 2014) for wetland system types associated with springs in the ecoregion (Dinerstein et al. 2001, NMDGF 2006, Connally, ed. 2012).
- (4) Literature on the history, conservation, geohydrology, and biota of springs across the U.S. portions of the ecoregion (e.g., White et al. 1941, Brune 1975, Mace et al., eds. 2001, Echelle et al. 2003, Heitmuller and Reece 2003, Hubbs 2003; 2014, Lang et al. 2003, Sharp et al. 2003, Chowdhury et al. 2004, Pronatura Noroeste 2004, Heitmuller 2006, Heitmuller and Williams 2006, NMDGF 2006, Grunstra and Van Auken 2007, Karges 2007, Stonestrom et al., eds. 2007, George et al. 2011, Hershler et al. 2011; 2014, Hanna et al. 2013, Misztal et al. 2013, USFWS 2011; 2012; 2013, USBR 2012 - (see additional citations below and in Appendix 1).
- (5) The assessments presented in Chapters 3 and 8 concerning the amphibian and fish species of the U.S. portion of the ecoregion and their associations with springs, perennial streams, and rivers (see Chapters 3 and 8).
- (6) Review comments provided by Dr. Michael T. Bogan on a draft of this chapter (Bogan 2016).

Several publications identify the wetland ecological system types associated with springs in the Chihuahuan desert ecoregion (Dinerstein et al. 2001, NMDGF 2006, Connally, ed. 2012, NatureServe 2014). As noted for the other wet system conservation elements, the term, “ecological systems” here refers to “... recurring groups of biological communities that are found in similar physical environments and are influenced by similar dynamic ecological processes, such as fire or flooding” (Comer et al. 2003). The wetland ecological system types associated with spring-emergent wetland sites in the Chihuahuan desert ecoregion include North American Arid West Emergent Marsh (International Ecological Classification Code CES300.729) and North American Warm Desert Ciénega (CES302.747). However, these two system types also are associated with the perennial streams and rivers of the ecoregion (see Chapters 8 and 9) and therefore are not uniquely characteristic of springs in the ecoregion.

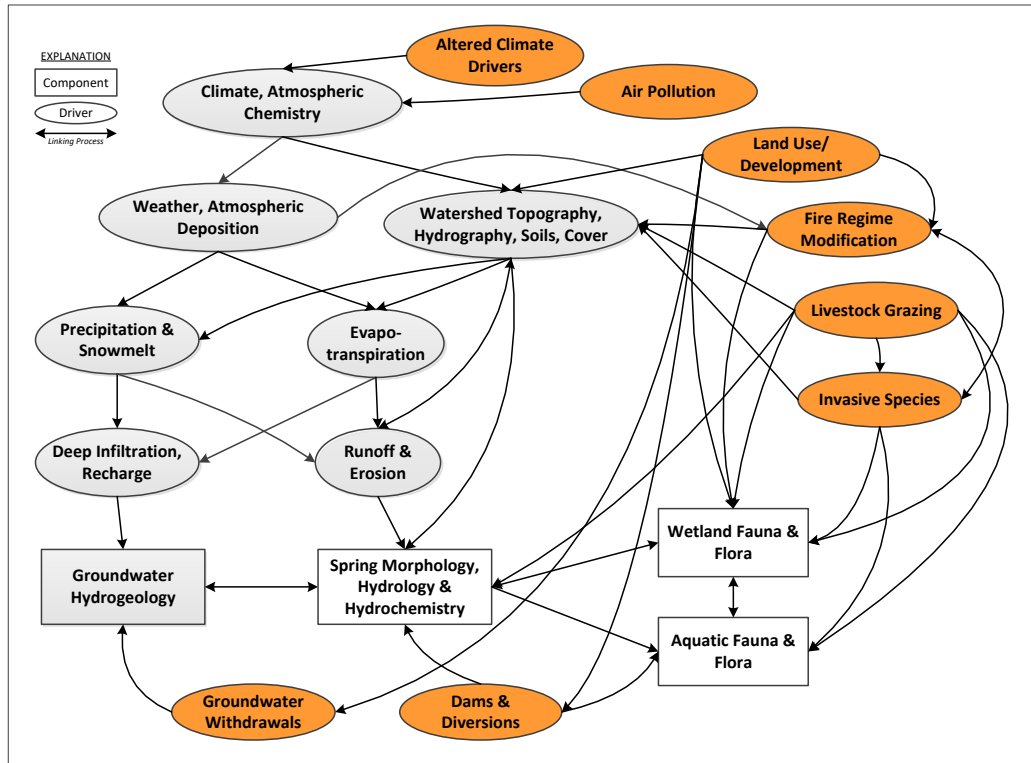
10.2 Springs-Emergent Wetlands System Control Model

Figure 10-1 shows the control model for the Chihuahuan desert springs-emergent wetlands system. The control model shows drivers and system components in greater detail than the overarching Chihuahuan Desert wet system conceptual model. System components consist of pivotal physical, biological, and ecological characteristics of a resource, its abundance, and its distribution. Anthropogenic drivers are colored orange, to distinguish them from natural drivers (grey). As in the overarching wet system model, arrows simply represent relationships in which one model component affects or influences another. The stressor model presented later in this chapter addresses the details of these relationships.

Spring-emergent wetland sites in the ecoregion, as noted above, occur across a wide range of elevations and topographic settings within the U.S. portion of the ecoregion; exhibit a wide range of variation in geophysical features; exhibit a wide range of hydrologic characteristics (e.g., differing average rates and consistencies of discharge); and exhibit a wide range of water quality (including thermal) properties. This diversity is constrained only by the geology (including geochemistry), hydrology, and topography of the region. Spring-emergent wetland sites in the ecoregion also vary in the frequency with which they become or remain connected to the larger surface water network of the region. The frequency of

connection (over both historic and geologic time scales) allows obligate aquatic species to move among spring-emergent wetland sites: the more hydrologically isolated a site, the more likely the site will come to harbor unique, endemic obligate aquatic species.

Figure 10-1. Chihuahuan desert springs-emergent wetlands system control model.



Subterranean networks of caves and fissures may also harbor endemic stygofauna; and springs connected to such subterranean networks will share some of these fauna, as noted above. Such connectivity exists and is known to shape the ecology of springs in Mexican portions of the Chihuahuan desert ecoregion (e.g., Dinerstein et al. 2001), and in the Edwards Plateau ecoregion immediately to the east of the Chihuahuan desert ecoregion in the U.S. However, subterranean biotic connectivity does not appear to shape the ecology of springs in the U.S. portion of the Chihuahuan desert ecoregion.

Spring-emergent wetland sites in the ecoregion may be classified ecologically based on their vegetation, topographic position, morphology, chemistry, size/flow rate, and other characteristics (e.g., Muldavin et al. 2000, Dinerstein et al. 2001, Heitmuller and Williams 2006, Springer and Stevens 2008, Porter et al. 2009, Miller et al. 2010, Misztal et al. 2013). Two other factors also significantly affect ecological variation among the springs and their associated emergent wetlands in the ecoregion: the frequency (perennial to rare or null) with which a spring-emergent wetland site is connected to (1) the larger surface water network of a region; and (2) any subterranean aquatic network of caves and fissures. However, the present assessment does not offer a classification, for two reasons. First, data on potential classification criteria (e.g., hydrogeology, morphology, discharge rates, chemistry) are available for only a few springs across the ecoregion. Second, across their great diversity, the springs and associated emergent wetlands of the ecoregion nevertheless are shaped by a single set of general physical

environmental and ecological processes in common. These processes result in (a) broad biological and ecological similarities and differences across space, and (b) similar broad patterns of variation over time in response to disturbance. The springs-emergent wetlands system control and stressor models attempt to capture these shared processes.

The springs-emergent wetlands system control model specifically identifies the following system components:

- **Spring Morphology, Hydrology & Hydrochemistry** refers to the geomorphology, hydrology, and hydrochemistry of the spring orifice and pool. Morphology includes depth and width, other aspects of geometry (e.g., overhangs or shelves), and connection to subterranean aquatic habitats. An individual spring may have multiple orifices (points of groundwater emergence) feeding a single pool. Several springs sometimes may occur in close proximity, with their outflows and associated wetlands merging into a single “spring complex.” Hydrology includes the daily, seasonal, annual, and longer-term variability in spring discharge. The conceptual model specifically addresses spring-emergent wetland sites with naturally seasonal or perennial discharge. Springs fed entirely by large and/or deep aquifers flow perennially with great consistency; those fed only by small and/or shallow aquifers may flow only seasonally and/or may exhibit varying discharge over time, day-to-day up to year-to-year. Springs fed by combinations of aquifers may show intermediate variability. Hydrochemistry includes the daily, seasonal, annual, and longer-term variability in spring water quality, in terms of dissolved and suspended matter constituents, and water temperature and pH. Hydrochemical conditions immediately at the spring orifice(s) in a spring-emergent wetland site typically differ from conditions downstream along the outflow path. The groundwater at its point of emergence reflects the chemistry, temperatures, and pressures of the geologic formations through which it has passed. The chemistry of the water changes as the temperature of the water and its dissolved gases and solids interact with the air and the biota of the spring pool(s) (Springer and Stevens 2008).
- **Aquatic Fauna & Flora** refers to the density, composition, and structure of the biotic community in the water. The unique geologic history and chemistry of each spring uniquely shapes the spectrum of aquatic fauna and flora that occur in or float on its waters, or live or root in its benthic substrates. Specifically, each spring supports only aquatic species that can tolerate or take advantage of the unique hydrochemistry of the site. Community composition may vary along the gradient from immediately at and around the spring orifice(s) to further downstream along the outflow path, shaped by changes in habitat morphology, water temperature, and chemistry along this distance.
- **Wetland Fauna & Flora** refers to the density, composition, and structure of the palustrine biotic community that surrounds the open waters of each spring or spring complex. The unique chemistry of each spring uniquely shapes the spectrum of its surrounding palustrine biota. Specifically, each spring supports only flora that can tolerate or take advantage of the unique hydrochemistry of the site; and fauna that tolerate or prefer these unique waters and plant assemblages. Native near-spring vegetation also affects discharge quantity and quality by: (1) consuming water (transpiration); (2) creating shade during warmer months, moderating water temperature; and (3) dropping litter into the pool, thereby altering the availability of organic matter. The wetlands associated with springs typically support rich assemblages of insects, reptiles, birds, and mammals attracted to the oases created by the presence of water in a desert environment

Environmental components and natural drivers that shape these system components in turn include the following:

- **Groundwater Hydrogeology** determines the natural chemistry and hydrology of springs and their emergent wetlands. The water that emerges at each spring has passed through a unique combination of groundwater flow systems, with distinctive geochemical compositions, temperatures, pressures, and flow path durations. Each unique combination imparts a distinctive chemical “signature” to the water at the spring where it emerges (e.g., Hem 1992, Alley, ed., 1993). The water itself mostly (but not always) entered these flow paths as a result of recharge from the ground surface (*aka* “meteoric” water). The time from recharge to discharge may span years to millennia. In the absence of human interventions, the amount of water discharging at a spring therefore reflects long-term averages for precipitation, ground surface conditions, and recharge.
- **Runoff & Erosion** across watershed surfaces may deliver surface water to spring pools and outflows, along with sediment, particulate organic matter, and dissolved inorganic and organic matter. Contributions of runoff may cause fluctuations in spring outflow beyond any caused by fluctuations in groundwater discharge.
- **Watershed Topography, Hydrography, Soils, and Cover** affect springs indirectly through their effects on watershed processes that shape groundwater recharge, surface water movement, chemistry, temperature; watershed soil erosion and deposition; and the transport of sediment and organic matter in runoff. However, this environmental element also can affect springs directly by determining the topographic locations where springs can emerge and their outflows depart; and by shaping the potential for upland wildfires to spread into the palustrine zone.

The following anthropogenic drivers shape these system components, environmental elements, and natural drivers:

- **Groundwater Withdrawals** alter aquifer system storage and flow gradients in ways that can reduce spring discharge and alter its water quality. Since the aquifer systems from which springs emerge may extend beneath large areas (hundreds to thousands of square km), groundwater withdrawals can affect springs hundreds of km distant from the site(s) of the withdrawals.
- **Dams & Diversions** at individual springs or spring complexes change the morphology of spring pools and the hydrology of spring pools and outflows. Dams at springs and dry reaches created by diversions from springs alter the connectivity of springs with downstream aquatic (stream and river) ecosystems, preventing the up/downstream exchange of aquatic biota, nutrients, and organic matter.
- **Invasive Species** alter the composition of the aquatic and palustrine biotic communities. Invasive species can also alter ecological processes such as herbivory and predation on or among native species, competition for food and habitat among native aquatic fauna, the structure of the aquatic food web, evapotranspiration, and water chemistry. Invasive species also can affect springs indirectly by altering the ground cover, soils, and wildfire regimes of surrounding watersheds.
- **Livestock Grazing** can alter palustrine vegetation through herbivory; and alter spring/spring complex morphology through trampling, which can also directly cause mortality among both terrestrial and aquatic flora and fauna (e.g., Kodric-Brown and Brown 2007, Kodric-Brown et al. 2007). Use of spring sites historically by large native herbivores may have had similar impacts. Livestock grazing also can affect springs indirectly through its impacts on upland soils and

ground cover, thereby affecting watershed processes; and by serving as a vector for the introduction of non-native species into a locality.

- **Fire Regime Modification**, both through wildfire management and through the effects of altered watershed vegetation and climate, alter the frequency, timing, and severity of wildfires across a landscape. Such changes can affect palustrine vegetation at springs, both directly through changes in their wildfire regime and indirectly through the effects of upland wildfire on the spread of invasive species across watersheds. Fire regime modifications also affect springs indirectly by altering land surface permeability and soil vulnerability to erosion, which in turn affect watershed processes such as recharge and runoff.
- **Land-Use/Development** can directly affect springs, spring complexes, and their wetlands when the development occurs directly at and alongside springs and their outflows, including recreational development. At a greater remove, land-use/development drives the intensity and geography of groundwater withdrawals, and the damming and diversion of springs to supply human activities. Land-use/development also alters watershed cover, land surface permeability, soil vulnerability to erosion, and releases of chemical pollutants into both watershed soils and water courses, with effects that cascade through the entire eco-hydrologic system; and shapes wildfire management policies and actions.

The springs-emergent wetlands system control model also recognizes the impacts of climate change and air pollution on this system. These drivers affect conditions at spring-emergent wetland sites indirectly, through their effects on weather and atmospheric deposition, which in turn have cascading effects on upland soils, ground cover, and watershed processes affecting both groundwater recharge and surface runoff quantity and chemistry.

10.3 Springs-Emergent Wetlands System Stressor Model

Table 10-1 presents and defines the drivers, critical environmental elements, critical ecological processes, and ecological outcomes included in the springs-emergent wetlands system stressor model. The stressor model subdivides and categories the system components in even greater detail than does the system control model. The stressor model follows the methodology described in Chapter 4.

Table 10-1. Chihuahuan desert springs-emergent wetlands system stressor model drivers, critical environmental elements, critical ecological processes, and ecological outcomes.

Model Component	Definition
Drivers	
Air Temperature Regime	The pattern of variation in air temperature, including daily, seasonal, annual, and longer-term variation in central tendencies, maxima, and minima.
Atmospheric Deposition	The pattern of variation in the deposition of potential pollutants from the atmosphere onto the land and water surfaces of the Chihuahuan desert ecoregion, including variation in pollutant types and in their rates of wet, dry, and total deposition.
Domestic Grazing Management	The pattern of management of the spatial distribution and intensity of domestic livestock grazing immediately at and across watersheds surrounding springs.
Fire Management	The pattern of management of wildfire suppression and prescribed burns immediately at and across watersheds surrounding springs.
Near-Spring & Watershed Land Development	The pattern of development of the land surface immediately at and across watersheds surrounding springs to support human activities, involving intentional modification of vegetation, soils, or topography and/or construction and maintenance of structures and engineered surfaces; and pollutants released by the associated human activities

Model Component	Definition
Non-Native Species Introductions	The types, origins, and patterns of introduction (where, when, how) of non-native species into the ecoregion. This driver does not include domesticated livestock or species intentionally introduced by fish and game managers for recreational sport.
Precipitation & Snowmelt Regime	The form (rain, ice, snow) and pattern of variation in precipitation, including daily, seasonal, annual, and longer-term variation in magnitude, frequency, timing, and rate (intensity); and the annual pattern of variation in the rate and timing of snowmelt.
Sport & Nuisance Species Management	The pattern of management of sport and nuisance species spatial distributions and densities, including management by governmental agencies, private individuals, and non-governmental organizations; and including species rearing, releasing, monitoring, control, and removal.
Water Management & Use	The pattern of management of surface- and groundwater storage, transport, and use (where, when, at what magnitudes) by public agencies, private organizations, and private individuals, controlled by structures such as dams, diversions, pipelines, and well fields.
Critical Environmental Elements	
Fluvial Network Connectivity	The capacity of the surface water flow network to which a spring's outflow naturally connects (or would connect in the absence of artificial barriers) to support up- and downstream exchanges of aquatic biota, as determined by the spatial distribution of natural and artificial features that may prevent or inhibit such exchanges.
Runoff Water Quality	The chemical and physical properties of the water that runs off a watershed into a spring or its outflow, including temperature, pH, turbidity, and concentrations of dissolved and suspended constituents; and the patterns of variation in these properties, including daily, seasonal, annual, and longer-term variation in their magnitudes.
Runoff Regime	The pattern of variation in the amount of water flowing off the surface of a watershed into a spring or its outflow, including daily, seasonal, annual, and longer-term variation; and including the frequency, timing, and duration of particular high and low flow rates.
Spring Pool and Outflow Morphology	The overall shape and stability of spring pools and outflow zones including pool and outflow geometry and edge stability; and artificial features such as dams and other barriers, and confining structures.
Watershed Erosion	The pattern of daily, seasonal, annual, and longer-term variation in the amount (mass and volume) and particle size distribution of sediment eroded off the surface of a watershed and transported as runoff into a spring or its outflow.
Watershed Ground Cover	The abundances and spatial distributions of classes of vegetated, disturbed, and artificial surfaces across a watershed that differ in their permeability to water infiltration, hydraulic roughness to water runoff, ability to inhibit soil erosion, and provision of shade (the latter of which can affect runoff temperatures and snowmelt).
Watershed-Scale Groundwater dynamics	The locations and rates of recharge of precipitation to groundwater systems; the storage volumes, inter-connections, and flow path lengths and duration of the aquifers that comprise these groundwater systems; the geochemical and hydrothermal dynamics of these groundwater systems; and the locations and rates of discharge to springs from these groundwater systems.
Critical Ecological Processes	
Aquatic Primary (1°) Productivity	The pattern of daily, seasonal, annual, and longer-term variation in the rate of production of biomass through photosynthesis by aquatic flora, including diatoms and algae.
Fluvial Biotic Connectivity	The ability of spring-adapted aquatic and palustrine species to move or be carried by currents from or to other spring-emergent wetland sites within a drainage network.
Spring Discharge Baseflow Regime	The pattern of seasonal, annual, and longer-term variation in groundwater-fed spring discharge without any inputs from runoff, but still affected by evapotranspiration.
Spring Discharge High-Flow Regime	The pattern of seasonal, annual, and longer-term variation in the rate of discharge of a spring-emergent wetland site caused by pulses of runoff.
Spring Discharge Low-Flow Regime	The pattern of variation in the seasonal, annual, and longer-term minimum rate of discharge of a spring-emergent wetland site (e.g., during drought).

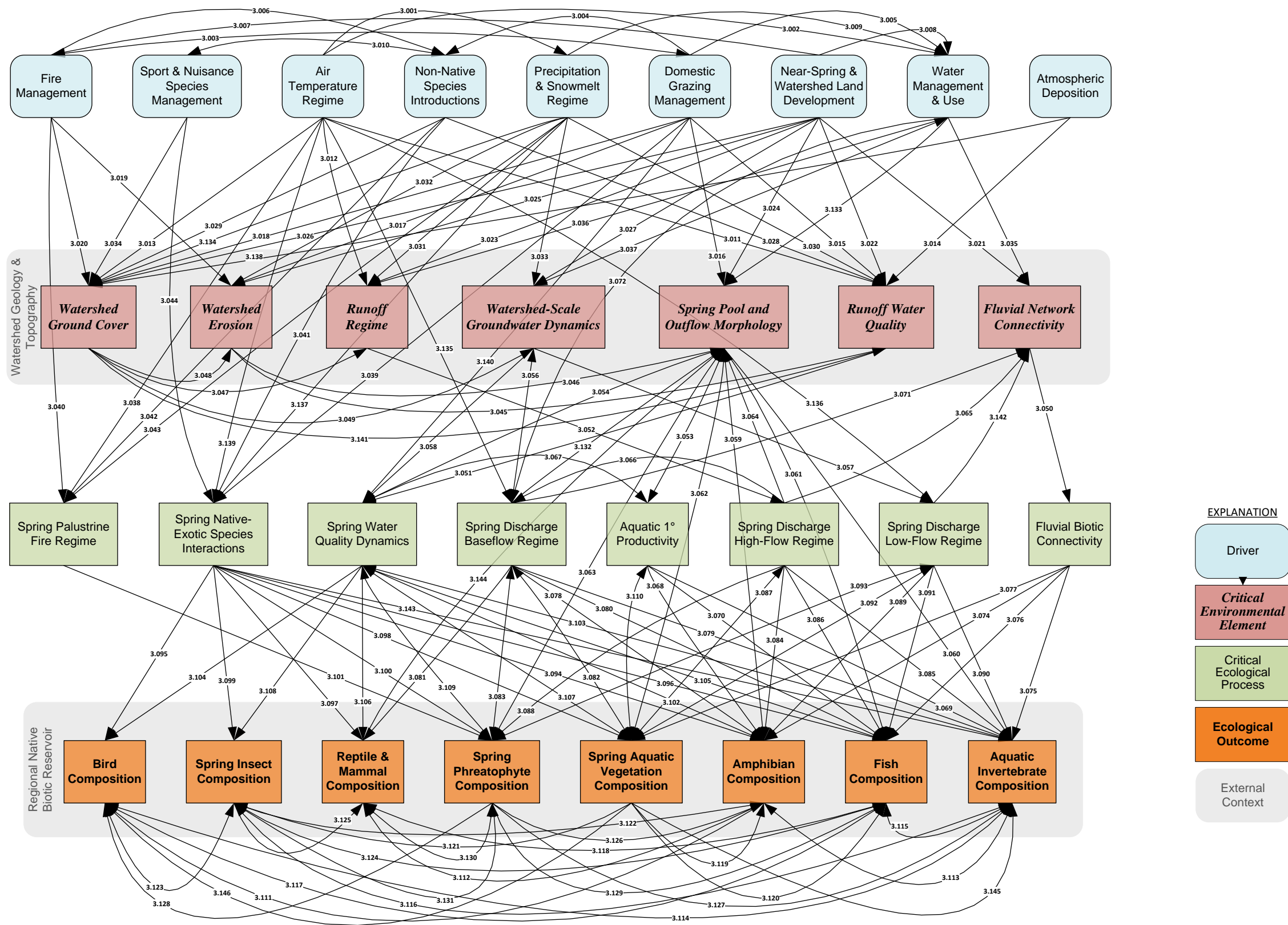
Model Component	Definition
Spring Native-Exotic Species Interactions	The ways, magnitudes, and spatial and temporal extent to which native and exotic species at a spring-emergent wetland site compete for habitat, food, and other materials; prey on, infect, or otherwise harm each other; or interact in mutually beneficial ways.
Spring Palustrine Fire Regime	The pattern of spatial distribution, extent, severity, timing, and frequency of wildfire and prescribed burns immediately surrounding springs.
Spring Water Quality Dynamics	The chemical properties of the water at a spring-emergent wetland site, including temperature, pH, and concentrations of dissolved and suspended constituents; and the patterns of daily, seasonal, annual, and longer-term variation in these properties, including variation over distance.
Ecological Outcomes	
Amphibian Composition	The taxonomic, functional, and size composition; spatial and temporal distribution; abundance; health; and activity level of the amphibian assemblages of a spring-emergent wetland site.
Aquatic Invertebrate Composition	The taxonomic, functional, and size composition; abundance; spatial and temporal distribution; and activity level of the aquatic invertebrate assemblage of a spring-emergent wetland site, including plankton, insect larvae, crayfish, and mollusks.
Bird Composition	The taxonomic and functional composition, spatial and temporal distribution, abundance, health, and activity level of the bird assemblage of a spring-emergent wetland site.
Fish Composition	The taxonomic and functional composition, spatial and temporal distribution, abundance, health, and activity level of the fish assemblages of a spring-emergent wetland site.
Reptile & Mammal Composition	The taxonomic, functional, and size compositions, spatial and temporal distributions, abundances, health, and activity levels of the assemblages of reptiles and mammals that occupy or visit a spring-emergent wetland site.
Spring Aquatic Vegetation Composition	The taxonomic composition, size range, spatial and temporal distribution, and abundance of emergent (aquatic) vegetation at a spring-emergent wetland site.
Spring Insect Composition	The taxonomic, functional, and size composition; abundance; spatial and temporal distribution; and activity level of the insect assemblage of a spring-emergent wetland site. (Note: aquatic larvae of insects are addressed as aquatic macroinvertebrates.)
Spring Phreatophyte Composition	The taxonomic composition, size range, spatial and temporal distribution, and abundance of phreatophytes surrounding a spring-emergent wetland site.

Figure 10-2 shows the stressor model for the springs-emergent wetlands system in the Chihuahuan desert ecoregion, built using the system model components shown in Table 10-1. Specifically, it displays all the system model components listed in Table 10-1 along with their causal relationships. Appendix 1 describes and presents the rationale for including every causal relationship in the stressor model, with citations for each causal relationship, providing a comprehensive presentation of the stressor model. Figure 10-2 indicates the presence or absence of causal relationships between the system model components but does not indicate the potential magnitude or other characteristics of these relationships, as explained in Chapter 4.

The stressor model (1) identifies the causal relationships that have affected how the condition of the system likely has changed in the past, in response to changes in its drivers; and (2) provides a means for articulating how the condition of the system will likely change in response to changes in its drivers. As discussed and illustrated below, the stressor model makes it clear: (a) which critical environmental elements would likely be affected by a change in one or more particular drivers, including change agents; (b) which critical ecological processes would likely be affected by the cascading effects of these changes in critical environmental elements; and (c) which system characteristics (ecological outcomes)

would likely be affected by the cascading effects of these changes in critical environmental elements and ecological processes. By doing so, in turn, the stressor model also highlights those components of the model—drivers, environmental elements, ecological processes, and ecological outcomes—that demand indicator data.

Figure 10-2. Chihuahuan desert springs-emergent wetlands system stressor model.



As defined earlier, a causal relationship exists when a change in one component of the system results in a change in some other component. Change in the first component is said to “cause a change in the second component. Each chain of causation, from driver to outcome, describes how the condition of the system likely has changed in the past, or likely would change in the future, in response to changes in its drivers.

Figure 10-2 includes two gray, background fields. One, labeled “Watershed Geology & Topography,” surrounds the critical environmental elements of the model. This first background field indicates that the critical environmental elements of the system derive from and are shaped by long-term dynamics of watershed geology and topography. However, the stressor model does not attempt to capture these longer-term dynamics. The second background field, labeled “Regional Native Biotic Reservoir,” surrounds the ecological outcomes of the system. This second background field indicates that the ecological outcomes of the system stressor model depend in part on the compositions of the regional reservoirs of native animal and plant species. Again, the stressor model does not attempt to capture these larger-scale dynamics.

The following paragraphs discuss how each Change Agent affects Chihuahuan Desert Springs-Emergent Wetlands, as articulated in the stressor model for this CE. A “sub-model” diagram for each Change Agent presents a simplified version of the master stressor model diagram, showing only the direct and indirect effects of the Change Agent of interest. The text accompanying each sub-model diagram summarizes the information from Appendix 1 concerning the impacts of each Change Agent, and includes selected citations from the more detailed list of citations for each causal link presented in Appendix 1. For each Change Agent, the paragraphs summarize present understanding of: (a) which critical environmental elements would likely be affected—directly or indirectly—by each Change Agent and (b) which critical ecological processes would likely be affected—directly or indirectly—by the cascading effects of these changes in critical environmental elements. However, the ecological characteristics of the system (ecological outcomes) affect each other and are affected by critical environmental elements and ecological processes in the same way regardless of which Change Agent is involved in altering these elements and ecological processes. For this reason, the presentation below begins with a discussion of the interactions between critical ecological processes and ecological outcomes, and among ecological outcomes.

10.3.1 Critical Ecological Processes and Ecological Outcomes

The springs-emergent wetlands stressor model includes eight ecological outcomes: amphibian composition, aquatic invertebrate composition, bird composition, spring aquatic vegetation composition, fish composition, reptile & mammal composition, spring insect composition, and spring phreatophyte composition. Table 10-1 above, defines these eight model components. These eight ecological outcomes directly affect each other in numerous ways—and therefore also affect each other indirectly in even more numerous ways—as shown in Figure 10-2 and documented in Appendix 1. For example (see Appendix 1 for full presentation):

- Phreatic and aquatic vegetation at springs affect native aquatic invertebrate, amphibian, and fish assemblage composition by providing habitat such as substrates and cover and/or by

providing food to various life stages (e.g., Kupferberg 1997, Propst 1999, Edwards et al. 2002, 2004, Echelle et al. 2003, Hubbs 2003; 2014, Kennedy and Hobbie 2004, Pease et al. 2006, Karges 2007, Zymonas and Propst 2007, Bateman et al. 2008a, Wallace and Anderson 2008, Propst et al. 2009, Porter et al. 2009, USFWS 2009; 2010; 2011; 2012; 2013a; 2013b, Bogan et al. 2012, Hanna et al. 2013, Minckley et al. 2013, Moody and Sabo 2013, Boeing et al. 2014, Miyazono 2014, Wood et al. 2016).

- Phreatic vegetation at springs affects spring insects, birds, and reptiles and mammals by providing food, water, and habitat options, which may differ depending on the composition and structure of the vegetation (e.g., Muldavin et al. 2000, Rosen and Caldwell 2004, Rosen 2005, Rosen et al. 2005, Bateman et al. 2008a; 2008b; 2008c; 2009, Malcom and Radke 2008, Sogge et al. 2008, Cerasale and Guglielmo 2010, Hagen and Sabo 2012; 2014, McCluney and Sabo 2012, 2014, Refsnider et al. 2013, Flesch 2014, Forstner et al. 2014, Gibson and Olden 2014, Gibson et al. 2014, Smith and Finch 2014). Non-native salt cedar at spring sites may also attract the Tamarisk beetle introduced as a biocontrol (see Chapters 8-9).
- Aquatic vegetation at springs provides nesting and feeding habitat for waterfowl. The types and quality of this habitat varies with the composition, spatial and temporal distribution, and abundance of the vegetation (e.g., Muldavin et al. 2000, NMDGF 2006, Malcom and Radke 2008, Merritt and Bateman 2012, Brand et al. 2013, Minckley et al. 2013, Smith and Finch 2014, Cole and Cole 2015). Non-native giant reed at spring sites may also attract insects introduced as biological control agents (Goolsby et al. 2016).
- Spring insects provide food options for birds, reptiles and mammals, and amphibians, depending on what insects are available, at what times, and in what abundances. The composition and abundance of the spring insect assemblage therefore can affect the composition of the bird, reptile and mammal, and amphibian assemblages using these sites (e.g., Kupferberg 1997, Skagen et al. 1998, Rosen and Caldwell 2004, Rosen 2005, Rosen et al. 2005, Bateman et al. 2008a; 2008b, Sogge et al. 2008, Cerasale and Guglielmo 2010, Hagen and Sabo 2012; 2014, Flesch 2014, Flesch 2014, Forstner et al. 2014, Smith and Finch 2014).
- Amphibians at spring-emergent wetland sites may provide food options for birds and reptiles and mammals, depending on what amphibians are available, at what times, and in what abundances. The composition and abundance of the amphibian assemblage therefore can affect the composition of the bird and reptile and mammal assemblages at these sites (e.g., Mora et al. 2002, Schmitt et al. 2005, White et al. 2006, Bateman et al. 2009; 2013).
- Fish at spring-emergent wetland sites also may provide food options for birds and reptiles and mammals, depending on what fish are available, at what times, and in what abundances. The composition and abundance of the fish assemblage therefore can affect the composition of the bird and reptile and mammal assemblages at these sites (e.g., Schmidly and Ditton 1978, MacRae et al. 2001, Mora et al. 2002, Schmitt et al. 2005, Bateman et al. 2009, Oring et al. 2013, Forstner et al. 2014).
- Aquatic invertebrates at spring-emergent wetland sites may provide food options for some birds, again depending on what invertebrates are available, at what times, and in what abundances. The composition and abundance of the aquatic invertebrate assemblage therefore can affect the composition of the bird assemblages at these sites. Additionally, birds that consume aquatic invertebrates can bio-accumulate contaminants (e.g., organochlorines, mercury) that these invertebrates have bio-accumulated through their own diets and exposures. This can result in high body loads of such contaminants in the birds, to such high levels that this impairs health and reproduction (e.g., MacRae et al. 2001, Mora et al. 2002, White et al. 2006, Oring et al. 2013).

- Aquatic invertebrates at spring-emergent wetland sites may provide food options for some fish and amphibians, depending on what invertebrates are available, at what times, and in what abundances. In turn, feeding pressure by fish and amphibians can affect the composition and abundance of the benthic invertebrate assemblage, with these pressures varying depending on what fish and amphibians are present, at what times, and in what abundances. Consequently, the composition and abundance of the benthic invertebrate assemblage can both affect and be affected by the composition of the bird assemblages along the stream corridors (e.g., Edwards 1997, MacRae et al. 2001, Witte 2005, Bergeron et al. 2011, USFWS 2011; 2012; 2013, Martinez 2012, Moody and Sabo 2013).
- Spring insect larvae are important constituents of the aquatic invertebrate assemblage. The composition and abundance of the aquatic invertebrate assemblage therefore both affects and is affected by the composition of the spring insect assemblages (e.g., Wallace and Anderson 2008).

The springs-emergent wetlands stressor model includes eight critical ecological processes that directly affect the ecological outcomes discussed above, shown in the following order in Figure 10-2: spring palustrine fire regime, spring native-exotic species interactions, spring water quality dynamics, spring discharge baseflow regime, aquatic 1^o productivity, spring discharge high-flow regime, spring discharge low-flow regime, and fluvial biotic connectivity. Table 10-1, above, defines these eight model components. These eight critical ecological processes directly affect – and in some cases are also affected by – the eight ecological outcomes in numerous ways, as shown in Figure 10-2 and documented in Appendix 1. Alterations to these critical ecological processes as a result of changes in drivers and critical environmental elements necessarily lead to altered ecological outcomes. The following paragraphs provide examples of the interactions of the eight critical ecological processes with ecological outcomes in relatively unaltered systems (see Appendix 1 for full presentation):

- The spring palustrine fire regime directly affects the spring phreatic vegetation. Uncharacteristic wildfire frequency and intensity in riparian corridors therefore can alter the frequency, intensity, and spatial extent of such mortality events (e.g., Muldavin et al. 2000). Chapter 8 provides further information on how wildfire affects phreatophyte assemblage composition along perennial streams.
- Spring native-exotic species interactions have pervasive effects across all ecological characteristics of perennial streams, as discussed later in this chapter (see Invasive Species, below).
- Spring water quality directly affect the composition and abundance of phreatic and aquatic vegetation through the effects of water temperature and the nutrients, salts, and potentially harmful chemicals dissolved in the emerging spring water or carried in with runoff; and the vegetation reciprocally can affect the concentrations of salts and potentially harmful chemicals as well (e.g., El-Hage and Moulton 1998, Muldavin et al. 2000, Deloach et al. 2000, Mills 2005, Shafroth et al. 2005; 2008, USEPA 2005, Grunstra and Van Auken 2007, Karges 2007, Mitsch and Gosselink 2007, Faber-Langendoen et al. 2008, Minckley et al. 2013, Cole and Cole 2015).
- Spring water quality directly affects the composition and abundance of the aquatic invertebrate, amphibians, and fish assemblages. Water temperature, pH, turbidity, salinity, and concentrations of specific metals and ions can affect organism health, development, reproduction, feeding activities, and vulnerabilities to predation; can cause them to avoid springs with extremes of water quality or, if they are in fact adapted to these extreme

conditions, can allow them to occupy springs unsuitable for other species (e.g., Taylor 1987, Rinne and Minckley 1991, Edwards 1997, Propst 1999, MacRae et al. 2001, Edwards et al. 2002, 2004, Cowley et al. 2003, Hoagstrom 2003; 2009, Lang et al. 2003, Rosen and Caldwell 2004, Mills 2005, Witte 2005, Heitmuller and Williams 2006, White et al. 2006, Karges 2007, Zymonas and Propst 2007, Propst et al. 2008; 2009, Wallace and Anderson 2008, Porter et al. 2009, USFWS 2009; 2011; 2012; 2013a; 2013b, Hershler et al. 2011; 2014, Hanna et al. 2013, Johnson et al. 2013, Misztal et al. 2013, Bogan et al. 2014, Garrett and Edwards 2014, Hubbs 2014, Miyazono 2014, Walsh et al. 2014). Additionally, aquatic invertebrates may bio-accumulate contaminants and pass them up the food chain, as discussed above.

- Spring water quality also affect the aquatic invertebrate, amphibians, and fish assemblages at springs indirectly by affecting aquatic 1° (primary) productivity and the concentrations of allochthonous organic matter carried in by runoff (e.g., MacRae et al. 2001, White et al. 2006, Meyer et al. 2007, Porter et al. 2009, USFWS 2011; 2013a; 2013b). Aquatic 1° productivity and allochthonous inputs of organic matter in turn strongly affect the abundance of aquatic invertebrates, amphibians, and fish that feed on the resulting biomass (e.g., Edwards 1997, Kupferberg 1997, Propst 1999, Edwards et al. 2002; 2004, Hubbs 2003; 2014, Bonner et al. 2005, Hulbert et al. 2007, Karges 2007, Zymonas and Propst 2007, Hubbs et al. 2008, Wallace and Anderson 2008, Porter et al. 2009, USFWS 2009; 2010; 2011; 2012; 2013a; 2013b, Miller et al. 2010, Tank et al. 2010, Hershler et al. 2011; 2014, Hanna et al. 2013, Bell et al. 2014, Garrett and Edwards 2014, Whitney et al. 2014, East 2015, Wood et al. 2016).
- Spring baseflow, in combination with spring pool and outflow morphology, determines the extent of the wetted perimeter at a spring, the patterns of water depths and velocities in its outflow, and the depth to the water table in the immediate vicinity. In combination with spring water quality (see above), these factors affect the availability and suitability of habitat for aquatic and phreatic vegetation (e.g., Muldavin et al. 2000; Minckley et al. 2013; Cole and Cole 2015). See also the Chapter 8 discussion of riparian vegetation, most species of which occur at springs in the ecoregion as well.
- Similarly, spring baseflow, in combination with spring pool and outflow morphology, determines the extent of the wetted perimeter at a spring, the patterns of water depths and velocities in its outflow, and the depth to the water table in the immediate vicinity. In combination with spring water quality (see above), these factors also affect the availability and suitability of habitat for aquatic invertebrates, amphibians, fishes, reptiles, and mammals (e.g., Edwards 1997, Taylor 1987, Propst 1999, Muldavin et al. 2000, MacRae et al. 2001, Edwards et al. 2002; 2004, Hubbs 2003; 2014, Lang et al. 2003, Bonner et al. 2005, Witte 2005, Hulbert et al. 2007, Karges 2007, Zymonas and Propst 2007, Bateman et al. 2008a; 2008b, Hubbs et al. 2008, Propst et al. 2008, Springer and Stevens 2008, Hoagstrom 2009, Porter et al. 2009, USFWS 2009; 2010; 2011; 2012; 2013a; 2013b, Milholland et al. 2010, Turner et al. 2010, Hershler et al. 2011; 2014, Heard et al. 2012, Hanna et al. 2013, Minckley et al. 2013, Forstner et al. 2014, Hubbs 2014, Walsh et al. 2014).
- Baseflow can maintain the surface connectivity of perennial springs to the larger fluvial network, if the baseflow is sufficiently large and/or the downstream distance from spring to a perennial stream is sufficiently short. Some springs receive groundwater discharge only seasonally or intermittently after periods of intense recharge to their parent aquifer(s). These situations result in an intermittent connection to the larger fluvial network, again if the baseflow is sufficiently large and/or the distance from spring to a perennial stream is sufficiently short. Similarly, high-flow pulses through spring-emergent wetland sites, which occur when runoff flows into or through the site, also can connect otherwise isolated springs to the fluvial network of a watershed or basin, allowing movement of obligate aquatic biota and suspended matter in/out

of springs (e.g., Edwards et al. 2004, Meyer et al. 2007, Levick et al. 2008, Porter et al. 2009, Bogan et al. 2013; 2015, Boersma et al. 2014, Jaeger et al. 2014, Miyazono 2014, Sabo 2014, Murphy et al. 2015, Turner et al. 2015). Both baseflow and high-flow pulses at springs shape the morphology of the spring outflow zone through erosion and deposition along the zone, an effect similar to but less intense and more localized than the effect of runoff on stream channel morphology (Springer and Stevens 2008).

- Periods of reduced (low) outflow from springs – as with extreme low-flows in streams in the ecoregion – can result in the disappearance of flow along reaches downstream, if those reaches receive no further groundwater inputs or are naturally losing reaches. The disappearance of surface flow along such reaches creates breaks in connectivity between the spring and the larger fluvial network, preventing movement of obligate aquatic biota and suspended matter in/out of the affected springs (e.g., Edwards et al. 2004, Meyer et al. 2007, Levick et al. 2008, Porter et al. 2009, Bogan et al. 2013; 2015, Boersma et al. 2014, Jaeger et al. 2014, Miyazono 2014, Sabo 2014, Murphy et al. 2015, Turner et al. 2015). Reduced outflow from springs also can result in a smaller, shallower pool, which in turn can result in reduced habitat for obligate aquatic plants and animals, and reduced soil moisture for phreatophytes and ground-dwelling vertebrates and invertebrates (e.g., Rosen et al. 1994, Muldavin et al. 2000, Rosen and Caldwell 2004, Witte 2005, Bateman et al. 2008a; 2008b, Levick et al. 2008, Minckley et al. 2013, Bogan et al. 2013; 2014; 2015, Boersma et al. 2014, Jaeger et al. 2014, Hubbs 2014, Propst 2016).
- Fluvial biotic connectivity allows obligate aquatic vertebrates and invertebrates to move or disseminate their offspring within a drainage network, to occupy or find mates in undisturbed spring-emergent wetland sites, or recolonize disturbed sites. Such movements maintain genetic diversity across meta-populations. Fluvial biotic connectivity also allows a stream network to transport eggs, larvae, seeds, rhizomes, and other propagules among sites (e.g., Propst 1999, Muldavin et al. 2000, Edwards et al. 2002; 2004, Hubbs 2003; 2014, Lang et al. 2003, Karges 2007, Meyer et al. 2007, Zymonas and Propst 2007, Hubbs et al. 2008, Levick et al. 2008, Porter et al. 2009, USFWS 2009; 2010; 2011; 2012; 2013a; 2013b, Bogan et al. 2013; 2014, 2015, Boersma et al. 2014, Jaeger et al. 2014, Miyazono 2014, Sabo 2014, Murphy et al. 2015, Turner et al. 2015).

A few of the critical ecological processes also affect each other, as described in Appendix 1. Specifically, spring water quality dynamics both affect and are affected by aquatic 1° productivity; and high-flow pulses from runoff may recharge local aquifers around a spring site, supporting subsequent baseflow.

Finally, one critical environmental element, spring pool and outflow morphology, directly affects several of the ecological outcomes discussed above, as shown in the following order in Figure 10-2 and described in Appendix 1. Aquatic plant species, and aquatic invertebrate, amphibian, and fish species and life-stages at spring-emergent wetland sites often have specific habitat requirements for water depths, bank heights, and substrate textures that affect the likelihood that they will occupy parts (or any) of any given spring and associated emergent wetland. Any change in these physical characteristics of a spring, whether from natural processes, trampling by livestock, or artificial modifications for development of a spring site, therefore can affect the aquatic vegetation, aquatic invertebrate, amphibian, and fish assemblages at a site, as discussed below in the sections on the impacts of development and excessive domestic grazing.

10.3.2 Climate Change

The final report for the Chihuahuan Desert REA will include a discussion of current forecasts of the ways in which climate change will affect the ecoregion and its ecological resources. The present chapter presents only the conceptual model of the causal relationships and outcomes that potentially will be affected for spring-emergent wetland sites.

Figure 10-3 presents the stressor model for spring-emergent wetland sites in the U.S. portion of the ecoregion, simplified to show only those causal relationships that potentially will be affected by changes in the air temperature and precipitation and snowmelt regimes. Appendix 1 presents the rationale and citations for every causal link shown in the diagram.

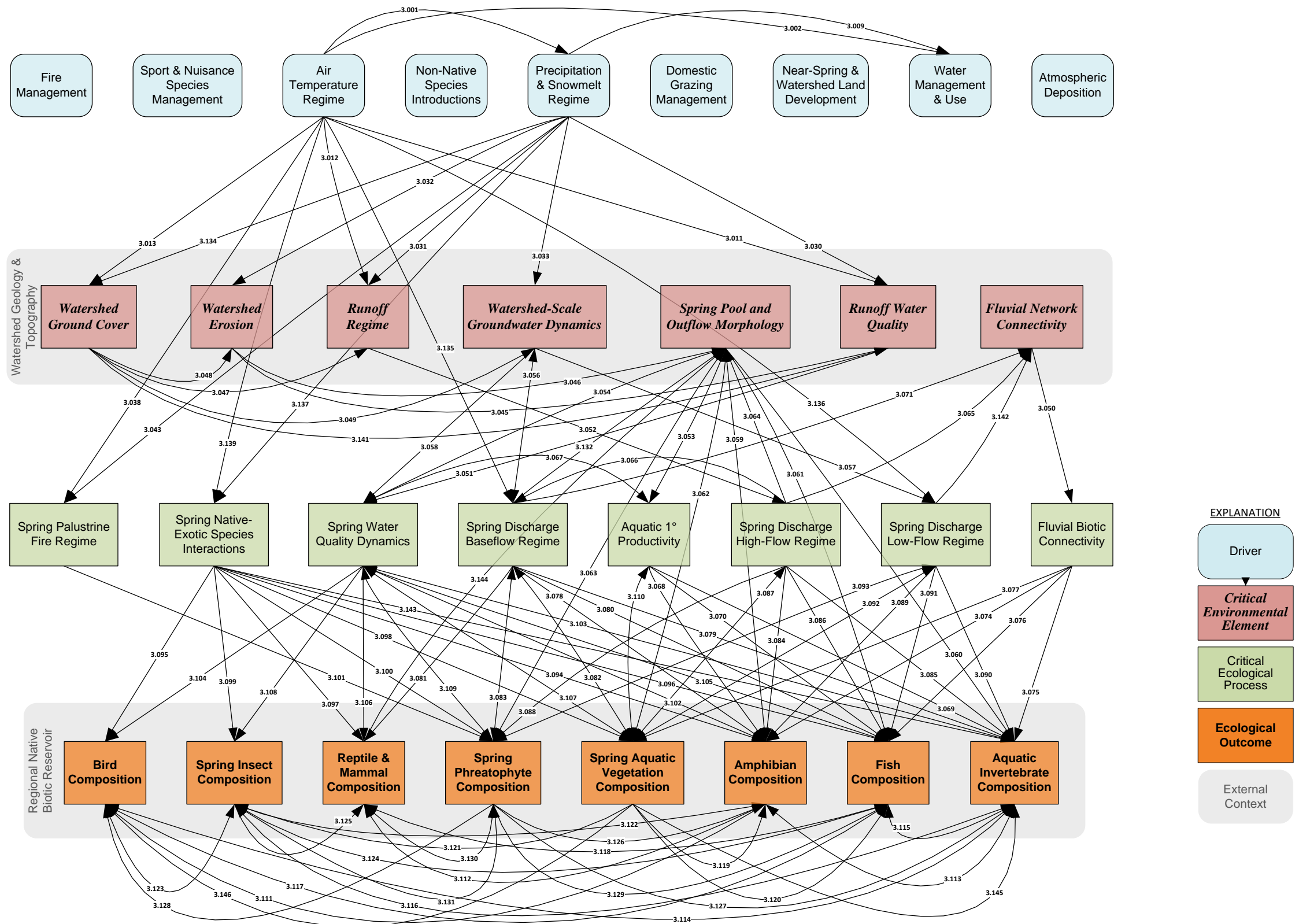
Climate change will affect spring-emergent wetland sites in the U.S. portion of the ecoregion through its effects on two drivers in the spring-emergent wetland systems stressor model: the air temperature regime; and the precipitation and snowmelt regime. Changes in these regimes may include changes in annual and seasonal averages, in the timing and magnitude of annual and seasonal extreme temperatures, and in the timing and magnitude of precipitation, as discussed in Chapter 2.

Changes in the air temperature and precipitation and snowmelt regimes in turn will directly affect five critical environmental elements in the spring-emergent wetland systems stressor model: watershed ground cover, watershed erosion, runoff regime, watershed-scale groundwater dynamics, and runoff water quality. The air temperature regime also affects the precipitation and snowmelt regime. Specifically, air temperature affects whether precipitation falls as rain or snow, whether precipitation even reaches the ground or evaporates as it falls (termed “virga” precipitation), and how much water runs off or infiltrates following precipitation versus simply evaporating.

Changes in the air temperature and precipitation and snowmelt regimes also will directly affect another driver, water management and use. Specifically, changes in air temperatures and precipitation will affect annual and seasonal water supply and demand. The resulting changes in water management and use will have their own, further effects on the runoff regime, watershed erosion, and watershed-scale groundwater dynamics, as discussed in Chapters 8-9 and in the section on the impacts of development, below, this chapter.

The impacts of changes in the air temperature and precipitation regimes on the five critical environmental elements in the spring-emergent wetland systems stressor model noted above are the same as in the perennial stream systems stressor model presented in Chapter 8.

Figure 10-3. Chihuahuan desert perennial stream systems stressor model: Potential impacts of climate change.



- Changes in air temperature and precipitation patterns will affect: (1) watershed ground cover by affecting the types, density, and rates of mortality of upland vegetation across a watershed (see Chapters 2-3 and 5-8); (2) runoff water quality by affecting water temperature – which affects other aspects of water quality – and the relative concentrations of soluble matter transported in the runoff; and (3) the rate at which salts accumulate across soil surfaces as a consequence of natural evaporative processes, and therefore the rate at which such salts are available for dissolution and transport in runoff, further affecting runoff water quality (Manahan 1991, see Chapters 2-3).
- Changes in precipitation, including storm intensity, will affect: (1) the runoff regime by altering the timing, amounts, forms, and rates of accumulation of the precipitation on the watershed surface (Chapters 2, 3, 8); and (2) the rate and spatial extent of soil erosion caused by individual storm events.
- Recharge to non-alluvial aquifers in the ecoregion mostly takes place at higher elevations across the mountains and foothills, and varies both with the amount of precipitation received and whether the precipitation occurs as rain or snow. Melting snow recharges more effectively than does rainfall. Changes in precipitation therefore will affect the spatial distribution and rates of recharge, which will affect the groundwater dynamics crucial to spring baseflow (e.g., Stonestrom et al., eds. 2007, Wolaver et al. 2008, Porter et al. 2009, USBR 2011, Szykiewicz et al. 2012; 2015a; 2015b, Friggens et al. 2013a, Sheng 2013, Friggens and Woodlief 2014, Jaeger et al. 2014, Eng et al. 2016, Meixner et al. 2016).

Changes in the air temperature and precipitation regimes will also directly affect three critical ecological processes in the spring-emergent wetland systems stressor model (see also Chapter 8):

- Changes in air temperature and precipitation patterns will affect the probability of wildfires initiated in or spreading into spring-emergent wetland sites (see uncharacteristic wildfire, below) (Pyne et al. 1996, Luce et al. 2012; see Chapters 2-3 and 5-8).
- Changes in air temperature and precipitation will directly affect spring native-exotic species interactions. Air temperature affects water demand in plants and thermal regulation in land animals, and native species may differ in their abilities to adjust to changes in air temperature patterns compared to non-native species. Similarly, precipitation directly at spring-emergent wetland sites also may affect water availability for both plants and land animals. Native species may differ in their abilities to adjust to changes in precipitation patterns compared to non-native species (e.g., Price et al. 2005, Enquist et al. 2008, Jones et al. 2010, Nagler et al. 2011, Friggens et al. 2013a; 2013b, Friggens and Woodlief 2014).
- Air temperatures determine the rates of evaporation of open water and the rates of transpiration by emergent and phreatic vegetation, which together affect baseflow after the groundwater emerges at a spring site. Changes in air temperature will affect spring baseflow by affecting these rates (e.g., Scott et al. 2000, Scott et al. 2004; 2008, Price et al. 2005, Stromberg et al. 2006, Serrat-Capdevila et al. 2007, Hatler and Hart 2009, Kennedy and Gungl 2010, Tillman et al. 2011, USBR 2011, Friggens and Woodlief 2014). Long-term changes to watershed-scale groundwater dynamics, as a result of changes in recharge (see above), also will affect baseflow, as discussed below.

These direct impacts of changes in the air temperature and precipitation and snowmelt regimes in turn will have further cascading impacts on other critical environmental elements, other critical ecological processes, and all ecological outcomes in the spring-emergent wetland systems stressor model. These cascading impacts will occur through the causal relationships described earlier (see Critical Ecological

Processes and Ecological Outcomes, above). These cascading effects are essentially the same as those described in Chapter 8 for perennial stream systems in the ecoregion (see Appendix 1 for full presentation).

Most crucially, changes in recharge to non-alluvial aquifers will result in changes to spring baseflow. However, the geologic flow paths that deliver groundwater to springs in the ecoregion vary greatly in length, with flow path durations of days to millennia. As a result, the effects on baseflow resulting from altered recharge will emerge over time spans of years to centuries or even millennia, with each spring uniquely affected (e.g., Heitmuller and Williams 2006, Webb and Leake 2006, Magruder et al. 2009, Porter et al. 2009, Kennedy and Gungle 2010, Tillman et al. 2011, USBR 2011, Friggens and Woodlief 2014, Jaeger et al. 2014, Eng et al. 2016, Meixner et al. 2016).

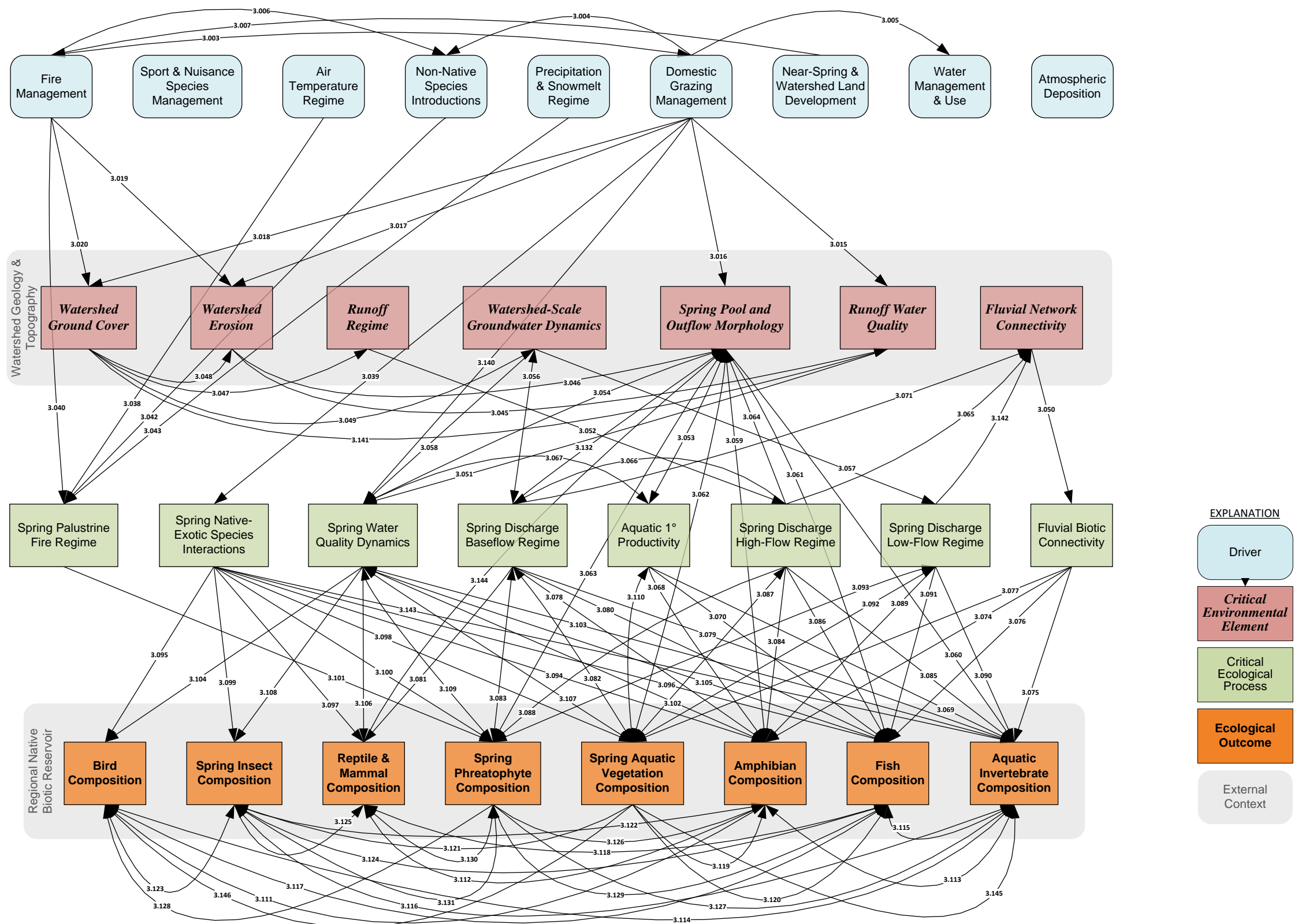
Changes in high- and low-flow pulse patterns at spring-emergent wetland sites due to changes in the air temperature and precipitation and snowmelt regimes may also affect fluvial network connectivity, which will in turn affect fluvial biotic connectivity, as discussed above. Such alterations to biotic connectivity could result in increasing biological isolation and species loss among spring-emergent wetland sites—see discussion of fluvial biotic connectivity above.

10.3.3 Uncharacteristic Wildfire

The fire regime of the U.S. portion of the Chihuahuan desert has changed as a result of the interaction of several drivers. Chapters 2 and 3 discuss the causes and consequences of uncharacteristic wildfire across the ecoregion in general, Chapters 5-7 discuss the causes and consequences of altered fire regimes specifically across the terrestrial ecological systems of the region, and Chapter 8 discusses the consequences of uncharacteristic wildfire for perennial streams. Figure 10-4 presents the stressor model for spring-emergent wetland sites in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by uncharacteristic wildfire and excessive domestic grazing. As shown in Figure 10-4, four drivers in the spring-emergent wetland systems stressor model directly affect the spring palustrine fire regime in the U.S. portion of the ecoregion: fire management, the air temperature regime, non-native species introductions, and the precipitation regime. For purposes of the spring-emergent wetland systems stressor model, “fire management” includes management of fires both immediately at spring-emergent wetland sites and across the larger landscape.

Fire management practices directly determine whether and how fires at spring-emergent wetland sites are managed, including the use of prescribed fire (NMDGF 2006, Katz et al. 2009, Stromberg et al. 2009a; 2009b; 2012, BLM 2000, Theobald et al. 2010, Nagler et al. 2011, Poff et al. 2011, Connally, ed. 2012, see Chapters 2, 3, 5-8). Air temperature and precipitation patterns affect fire probabilities and intensities (Pyne et al. 1996, Luce et al. 2012, see Chapters 2-3). And introduced plant species at spring-emergent wetland sites may have different susceptibilities or adaptations to fire and/or may contribute at different rates to fuel loads, compared to native plants (D'Antonio and Vitousek 1992, see Chapters 2, 3, and 5-8).

Figure 10-4. Chihuahuan desert perennial stream stressor model: Potential impacts of uncharacteristic wildfire and excessive domestic grazing.



Fire management practices at the landscape scale also may indirectly affect spring-emergent wetland sites across the U.S. portion of the ecoregion, by affecting watershed ground cover and watershed erosion patterns. These effects in turn affect the runoff regime and runoff water quality, which affect several critical ecological processes in spring-emergent wetland systems.

Three other drivers also indirectly affect the spring palustrine fire regime across the U.S. portion of the ecoregion. Domestic grazing management affects both fire management and non-native species introductions; near-spring and watershed development affects fire management; and non-native species introductions also affect fire management (see sections on invasive species, development, and excessive domestic grazing, below, this chapter).

The resulting changes in the spring palustrine fire regime will in turn affect spring-emergent wetland sites across the U.S. portion of the ecoregion through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above).

10.3.4 Invasive Species

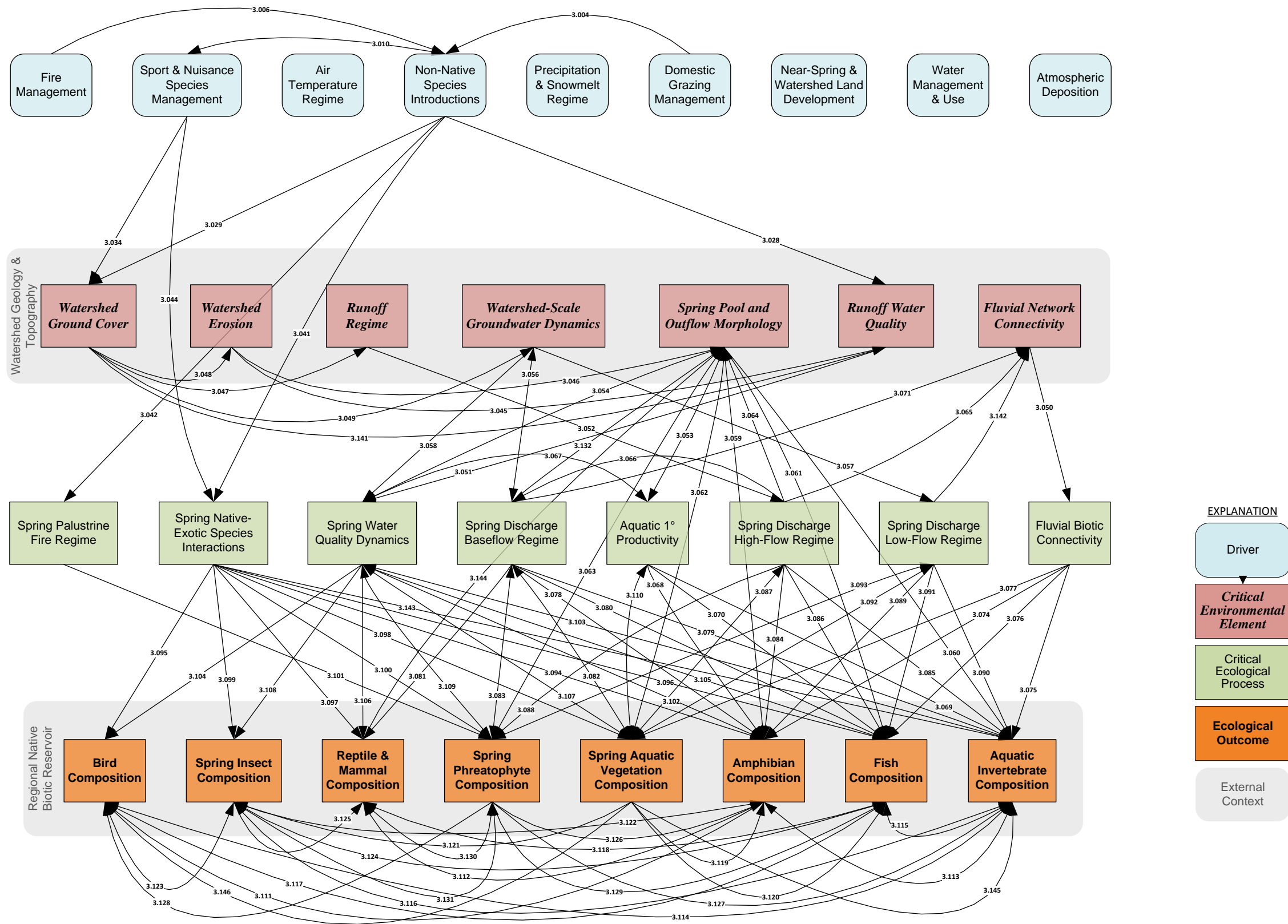
Chapters 2-3 discuss the ecological consequences of non-native species across the U.S. portion of the ecoregion in general, and Chapter 8 discusses these consequences for perennial streams. Figure 10-5 presents the stressor model for spring-emergent wetland sites in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by invasive species and their management. Figure 10-5 include two drivers that address the ways in which non-native species affect springs and emergent wetlands in the U.S. portion of the ecoregion: non-native species introductions; and sport and nuisance species management. Appendix 1 presents the rationale and citations for each causal link shown in the diagram.

Together, native species introductions and sport and nuisance species management significantly shape the ecological status of spring-emergent wetland sites in the U.S. portion of the ecoregion. The two affect each other and in turn are directly or indirectly shaped in part by two other drivers, fire management, and domestic grazing management, the effects of which are discussed separately above and below, respectively, this chapter.

Water management and use also indirectly affects sport and nuisance species management, as also discussed below (see development, this chapter). Decisions on the management of developed springs take into consideration needs for sport and nuisance species management, especially at springs that also harbor endangered native species.

The native spring and emergent wetland fauna and flora of the U.S. portion of the ecoregion in turn are affected by their interactions with non-native species and, potentially, by management actions taken to control some non-native species. These causal relationships are essentially the same as those discussed for perennial streams in Chapter 8, above: Introduced species may compete with native species for habitat or foods, prey on, infect, poison, or hybridize with native aquatic, wetland, and phreatic species. Non-native phreatic and emergent vegetation may compete with native plants for water and, at least in the case of salt cedar, alter soil salinity (see Chapter 8 and Appendix 1 for full presentation).

Figure 10-5. Chihuahuan desert perennial stream stressor model: Potential impacts of invasive species and landscape restoration.



10.3.5 Development

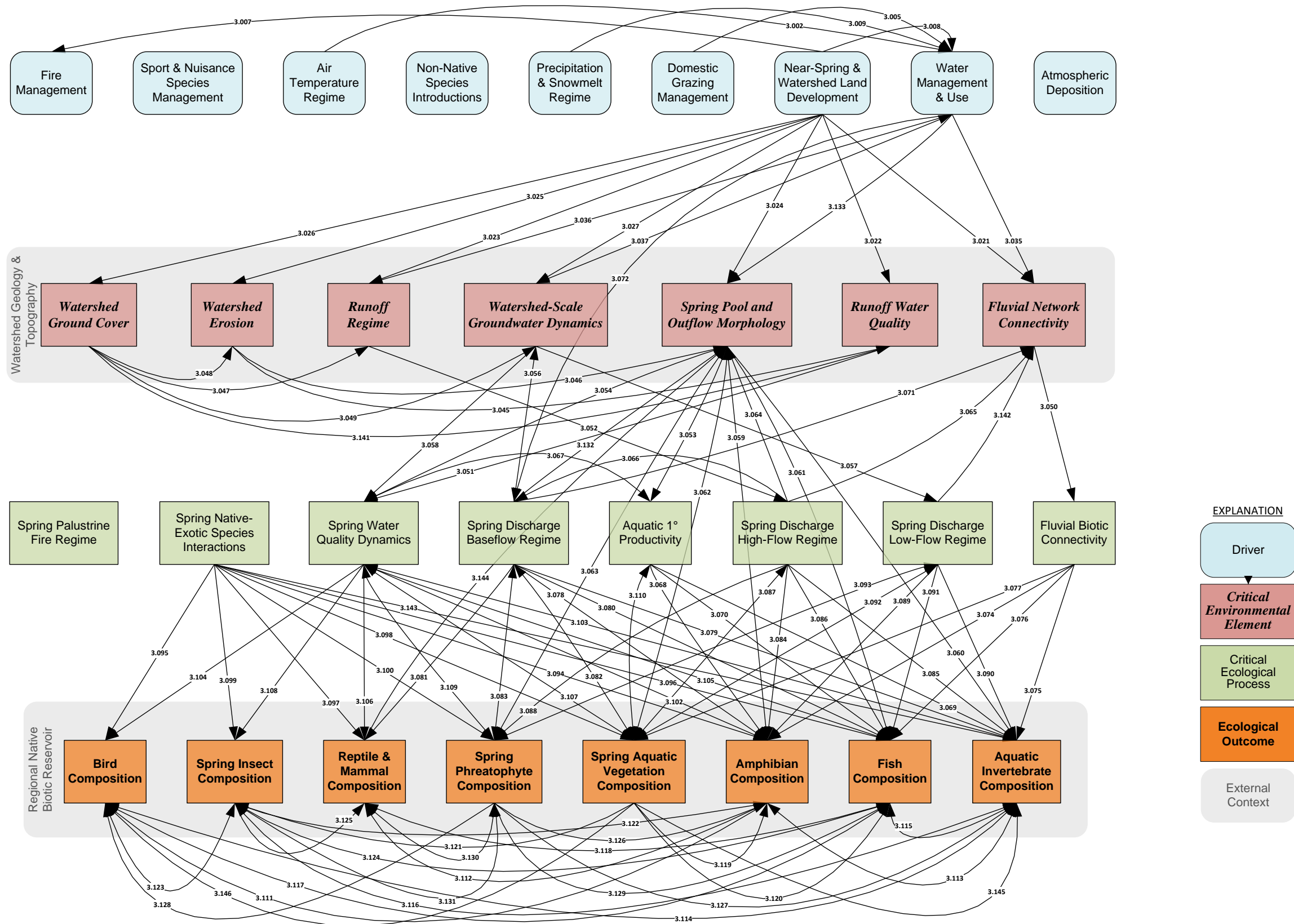
Chapters 2-3 discuss the ecological consequences of land and water development across the U.S. portion of the ecoregion in general, and Chapter 8 discusses the ecological consequences for perennial streams. Figure 10-6 presents the stressor model for spring-emergent wetland systems in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected two drivers that address the impacts of development on this CE: (1) near-spring and watershed land development, and (2) water management and use. Figure 10-6 shows the causal relationships through which these two drivers directly or indirectly affect every critical environmental element represented in the spring-emergent wetland systems stressor model, changes in which will affect almost every critical ecological process and ecological outcome. Appendix 1 presents the rationale and citations for each causal link shown in the diagram.

Near-spring and watershed land development directly affects all seven critical environmental elements in the stressor model for spring-emergent wetland systems: watershed ground cover, watershed erosion, runoff regime, watershed-scale groundwater dynamics, spring pool and outflow morphology, runoff water quality, and fluvial network connectivity (see Appendix 1 and Chapters 2-3 and 5-8). Changes in water management and use will directly affect four critical environmental elements in the stressor model for spring-emergent wetland systems: runoff regime, watershed groundwater dynamics, spring pool and outflow morphology, and fluvial network connectivity (see Appendix 1 and Chapters 2, 3, and 8). The impacts of water management and use thus compound the effects of near-spring and watershed land development on three critical environmental elements: runoff regime, watershed groundwater dynamics, and fluvial network connectivity. In fact, near-spring and watershed land development necessarily affects spring-related water management and use. Many if not most forms of watershed land development in the ecoregion require a water supply to sustain the activities associated with the development. This can result in withdrawals from the aquifers that also supply springs, or diversions directly from springs. Some springs in the ecoregion also have been developed for uses that do not involve withdrawals, such as livestock grazing or recreation. Near-spring and watershed land development also affects another driver, fire management. Fire management decisions also must take into account the types and locations of developed land, as these evolve across the landscape and around springs (see Uncharacteristic Wildfire, above, this chapter).

The impacts of changes in near-spring and watershed land development and in water management and use on the critical environmental elements in the springs-emergent wetlands stressor model are the same in most respects as the impacts of watershed and riparian land development and water management and use on the critical environmental elements in the perennial streams stressor model (see Chapter 8 and Appendix 1 for full presentation). The two stressor models differ only in two important respects, in the ways they characterize the impacts of development: The springs-emergent wetlands stressor model includes a critical environmental element for spring pool and outflow morphology, and does not include a critical environmental element for water impoundments, diversions and returns. The springs-emergent wetlands stressor model addresses impoundments or diversions of springs as alterations to spring pool and outflow morphology, and recognizes the diversion or consumption of spring water as water uses directly affecting spring baseflow. The following paragraphs

focus on the impacts of near-spring and watershed land development and water management and use on spring pool and outflow morphology and spring baseflow (see Appendix 1 for full presentation). See Chapter 8 for its discussion of all other effects of watershed development and water use:

Figure 10-6. Chihuahuan desert perennial stream stressor model: Impacts of land and water development.



- Land development immediately around a spring-wetland site may confine the spring pool and outflow within artificial control structures, and/or artificially "harden" outflow channel banks to prevent flows from reshaping their morphology and/or enhance human uses of the water (e.g., for recreation). Development of springs for recreational use or diversion also may entail the construction of dams and other confining structures. These activities modify spring pool and outflow morphology and can affect fluvial network connectivity (e.g., Brune 1975, El-Hage and Moulton 1998, McLemore 1999, Sharp et al. 2003, Mills 2005, Heitmuller and Williams 2006, Hoagstrom et al. 2008, Porter et al. 2009, USFWS 2009;2010; 2011; 2013a; 2013b, TWDB 2016, Januchowski-Hartley et al. 2013; 2014, Diebel et al. 2015, Fuller et al. 2015, see Chapters 1-2). Restoration of stream habitat may involve the removal or improvement of road-crossing barriers or the re-introduction of beaver, the dams of which can also affect fluvial network connectivity (e.g., Wild 2011, Gibson and Olden 2014, Gibson et al. 2014).
- Diversions of spring waters, for example for use in irrigation or in artificial watering features for livestock, necessarily reduce spring baseflow downstream from the point of groundwater discharge.
- Water management and use in the ecoregion has long involved the construction and operation of groundwater wells. Local, state, and bi-national water management policies and practices across the ecoregion determine how much water is consumed from the aquifers of the ecoregion and where and when it is withdrawn (e.g., George et al. 2011, NMOSE 2013, TWDB 2016, see also Chapters 2-3). Withdrawals from aquifers that also supply springs and emergent wetlands necessarily reduce baseflow discharge from the groundwater systems to the surface (Tillman et al. 2011, USBR 2011, Hogan 2013, NMOSE 2013; 2016a; 2016b, Sheng 2013, TWDB 2016).

Changes in spring pool and outflow morphology and in spring baseflow will in turn affect numerous ecological outcomes through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above).

10.3.6 Excessive Domestic Grazing

Chapters 2-3 discuss the ecological consequences of excessive domestic grazing across the ecoregion in general, Chapters 5-7 discuss its consequences specifically across the terrestrial ecological systems of the larger landscape, and Chapter 8 discusses its consequences for perennial streams. Figure 10-4 presents the stressor model for springs and emergent wetlands in the U.S. portion of the ecoregion, simplified to show only those causal relationships that potentially will be affected by uncharacteristic wildfire and excessive domestic grazing. Appendix 1 presents the rationale and citations for every causal link shown in the diagram. As shown in Figure 10-4, domestic grazing management in the U.S. portion of the ecoregion affects water and fire management practices. Ranchers across the ecoregion provide water for their livestock by damming intermittent streams to form small ponds, diverting perennial streams to watering structures, and drilling small wells to supply watering tanks. As also shown in Figure 10-4, domestic grazing management in the U.S. portion of the ecoregion both affects and is affected by the spread of non-native vegetation, as discussed in detail in Chapter 5-7. For example, grazed livestock act as vectors for spreading non-native plants.

The impacts of excessive domestic grazing on the critical environmental elements in the springs-emergent wetlands stressor model are the same as the impacts of this driver on the critical

environmental elements in the perennial streams stressor model (see Chapter 8 and Appendix 1 for full presentation). For example, trampling by livestock can directly alter spring pool and outflow morphology (e.g., Kodric-Brown and Brown 2007, Kodric-Brown et al. 2007), just as it can alter stream channel morphology (e.g., Belsky et al. 1999, see Chapter 8). The direct impacts of excessive domestic grazing on watershed ground cover, watershed erosion, spring pool and outflow morphology, and runoff water quality, and on spring native-exotic species interactions and spring water quality in turn affect numerous ecological outcomes through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above).

10.3.7 Landscape Restoration

Chapter 3 discusses the types of landscape restoration projects taking place in the U.S. portion of the ecoregion, some of which currently or in the future could affect spring-emergent wetland systems just as they currently or potentially could affect perennial stream and rivers in the ecoregion. These projects include efforts to remove non-native aquatic and riparian species, and to ensure the replacement of removed non-native vegetation with native vegetation. Specifically such efforts include or potentially could include the following (see Figure 10-5 and Appendix 1):

- Control or removal of non-native fishes, removal or improvement of road-crossing barriers, exclusion of livestock, or re-introduction of beaver and locally extirpated fish, as with restoration along perennial streams and rivers (e.g., Kapuscinski and Patronski 2005, NMDGF 2006, Hoagstrom et al. 2008, USFWS 2009; 2010; 2012, Wild 2011, Januchowski-Hartley et al. 2013; 2014, BEEC 2014, Gibson and Olden 2014, Hershler et al. 2014, Pilger et al. 2015). (Captive breeding and re-introductions of native species is not considered a type of landscape or habitat restoration, and so is not included here).
- Flow restoration, exclusion of livestock, or removal of non-native vegetation, with or without active restoration of native vegetation in its place, as with restoration along perennial streams and rivers (Farley et al. 1994, Stromberg 1998; 2001, Belsky et al. 1999, BLM 2000, Krueper et al. 2003, Bateman et al. 2008a; 2008b; 2008c; 2009, Malcom and Radke 2008, Shafroth et al. 2008, Abelho and Molles 2009, Katz et al. 2009, Smith et al. 2009, Stromberg et al. 2009a; 2009b, Theobald et al. 2010, Nagler et al. 2011, Poff et al. 2011, Watts and Moore 2011, Brand et al. 2013, Reynolds et al. 2014, Cole and Cole 2015, Mosher and Bateman 2016). Non-native species targeted for control or removal include salt cedar (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*), *Arundo donax*, an invasive reed, and Phragmites (*Phragmites* spp.), another invasive reed. Removal methods may include biological control, prescribed fire, mechanical removal, and chemical control, and these methods can have their own effects on riparian biota (e.g., Nagler et al. 2012, Goolsby et al. 2016).

10.4 Springs-Emergent Wetlands System Key Ecological Attributes

As noted earlier, all ecological outcomes and critical ecological processes in a system stressor model constitute key ecological attributes for the system. The list below identifies 16 key ecological attributes for Chihuahuan desert the springs-emergent wetlands system based on these criteria. *Characterizing the present condition of a system requires data on indicators for its key ecological attributes.* The definitions for the key ecological attributes are the same as the definitions for these model components presented above.

- **Ecological Outcomes**
 - Amphibian Composition
 - Aquatic Invertebrate Composition
 - Bird Composition
 - Fish Composition
 - Reptile & Mammal Composition
 - Spring Aquatic Vegetation Composition
 - Spring Insect Composition
 - Spring Phreatophyte Composition
- **Critical Ecological Processes**
 - Aquatic 1° (Primary) Productivity
 - Fluvial Biotic Connectivity
 - Spring Discharge Baseflow Regime
 - Spring Discharge High-Flow Regime
 - Spring Discharge Low-Flow Regime
 - Spring Native-Exotic Species Interactions
 - Spring Palustrine Fire Regime
 - Spring Water Quality dynamics

11 Playas and Playa Lakes Conceptual Model

This chapter presents the conceptual ecological model for the Playas and Playa Lakes CE, consisting of barren to sparsely vegetated depressions in topographic lows with episodic to persistent ponding and no drainage outlet. As discussed in Chapter 3, the U.S. portion of the ecoregion contains thousands of playas and playa lakes, from the large Lordsburg Playa in New Mexico near the Arizona border to clusters of small playas on the uplands east of the Pecos River (e.g., BLM 1999) in New Mexico and Texas. The larger playa and playa lake sites in the U.S. portion of the ecoregion, such as Alkali Flat and Lake Lucero, Lordsburg Playa, Isaacks Lake, and Playas Lake, all in New Mexico, are remnants of Pleistocene lakes that filled large expanses of closed (endorheic) drainage basins (Hawley 1993, Wilkins 1997). The wetting of playas and playa lakes in the U.S. portion of the ecoregion results from runoff following seasonal or episodic storms (see Chapter 2), supported by a rise in the local water table during particularly wet periods. Variation in precipitation strongly affects inundation patterns, with some playas filling – to varying degrees – and drying multiple times per year (MacKay et al. 1990, Bennett and Wilder 2009, Harings and Boeing 2014) while others may remain dry for years (Cooke et al. 1993).

Both basin-scale and local, near-playa aquifers may contribute to wetting at playas, but their interactions are complex and largely not well understood (KellerLynn 2012). Some playas occupy the lowest points in their entire basins and receive groundwater discharge from basin-fill aquifers (e.g., Rosen 1994, Hibbs et al. 2000, Johnson and Rappuhn 2002, Konieczki 2006, Porter et al. 2009, KellerLynn 2012). However, local, shallow aquifers may also exist, separate from deeper aquifers, recharged by local precipitation (Bennett and Wilder 2009, Porter et al. 2009, KellerLynn 2012).

The runoff and groundwater carry with them mixtures of salts derived from the surrounding and underlying geologic formations (George et al. 2005, Monger et al. 2006, Porter et al. 2009, NPS 2010, Szykiewicz et al. 2010, Sheng 2013, Sigstedt et al. 2016). Subsequent evaporation leaves behind a concentrated salt residue or evaporite (e.g., Langford 2003, KellerLynn 2012), the composition of which varies with the geochemistry of the setting (e.g., Langford 2003). Both the highly variable hydrology and the salinity of the resulting soils in turn strongly affect the vegetation and resident fauna.

Dune fields occur adjacent to some of the large playas in the U.S. portion of the ecoregion. The largest and best-studied such dune field – indeed, one of the best-studied playa basin dune fields in the world – is found at White Sands National Monument (WSNM) in the Tularosa Basin, New Mexico (Kocurek et al. 2007, KellerLynn 2008, Szykiewicz et al. 2010). Such dune fields are included in the definition of the Playas and Playa Lakes CE because they shape and are closely shaped by the hydrology, soils, and geochemistry of the playas and playa lakes (KellerLynn 2012, Kocurek et al. 2007, Szykiewicz et al. 2010, White et al. 2015).

Playas in the ecoregion vary in size, soil texture and geologic origin, soil and water chemistry, frequency and extent of ponding, and the presence and composition of vegetation (e.g., Muldavin et al. 2000, NPS 2010, NMDGF 2006, WWF-SIA 2007, USDA NRCS 2016): The innermost zone of most frequent wetting and drying – including lake shores – may be barren or populated with stands of three-square bulrush (*Scirpus pungens*) or common spikerush (*Eleocharis palustris*) depending on water quality. The open

waters and shores may then be surrounded by a zone dominated by iodinebush (*Allenrolfea occidentalis*) with varying admixtures of fourwing saltbush (*Atriplex canescens*), TransPecos sea lavender (*Limonium limbatum*), alkali seepweed (*Suaeda vera*), alkali mallow (*Malvella leprosa*), and other extreme halophytes that tolerate frequent or extended inundation. Inland saltgrass (*Distichlis spicata*) becomes increasingly dominant with less frequent or persistent inundation, joined at even greater distance from the wettest soils by other halophytic species including vine mesquite (*Panicum obtusum*), alkali sacaton grass (*Sporobolus airoides*), inland saltgrass, saltbush, spreading alkaliweed (*Cressa truxillensis*), TransPecos sea lavender, and low shrubs such as James' seaheath (*Frankenia jamesii*) and TransPecos false claddaisy (*Pseudoclappia arenaria*). Griffith's saltbush (*Atriplex griffithsii*), a rare species, occurs in this latter community exclusively in playas in the Chihuahuan Desert ecoregion and the adjacent Madrean Archipelago ecoregion near the Arizona-New Mexico border, including Lordsburg Playa (BLM 2000, WWF-SIA 2007). This community in turn may be surrounded by grasses such as buffalograss (*Buchloe dactyloides*), sparse shrubs, and phreatophytes including mesquite (*Prosopis glandulosa*) and the non-native salt cedar (*Tamarix* spp.) depending on soil chemistry and depths to the water table. The vegetation surrounding these concentric bands of halophytic and phreatic plants typically consists of upland scrub and grasslands (see Chapters 5-6), except where dunes are present, with their own characteristic vegetation (e.g., Muldavin et al. 2000, NMDGF 2006, WWF-SIA 2007, NPS 2010, NPS 2016).

Many playa lake sites in the U.S. portion of the ecoregion are important migratory stopover points for a diversity of shorebirds and other waterbirds (Drewien et al. 1995, Deason 1998, BLM 2000, Dinerstein et al. 2001, Desmond and Montoya 2006, NMDGF 2006, WWF-SIA 2007, see below). Many also provide habitat for unique assemblages of freshwater invertebrates and vertebrates including various branchiopods (clam shrimp, fairy shrimp), small molluscs, and a number of anuran species adapted to high salinities and unreliable hydrologic conditions (Lang and Rogers 2002, NMDGF 2006, KellerLynn 2008, Whitford and Bestelmeyer 2006, WWF-SIA 2007, NPS 2010, see below). A recent study of the gypsum dune fields at WSNM identified the white sands formation as home to more endemic Lepidoptera than any single habitat in North America (Metzler 2014). The White Sands pupfish (*Cyprinodon tularosa*), which occurs only in the closed Tularosa Basin in a few springs and their outflows, is not considered part of the playa ecological system for purposes of this REA, although it can temporarily colonize playa waters from its normal spring-fed habitat (Propst 1999, NatureServe 2015, Carman 2010, see Chapter 10). No other fish even temporarily inhabit playa waters in the U.S. portion of the ecoregion.

The REA initially considered distinguishing non-gypsiferous from gypsiferous playas and playa lakes as separate CEs, as discussed in Chapter 3. The unique geology of the U.S. portion of the ecoregion has created numerous zones in which gypsiferous geology, geochemistry, and soils affect ecological dynamics and foster potentially unique biota, with potentially distinctive management concerns (e.g., Monger et al. 2006). However, as discussed in Chapters 2-3, the REA chose to address these concerns instead through a Management Question applied to all CEs, to assess where gypsiferous geologic, geochemical, and soil conditions across the ecoregion may require management consideration.

The presentation below therefore addresses gypsiferous and non-gypsiferous playas and playa lakes,

together with any associated dune fields, as a single CE. The presentation follows the structure described in Chapter 4.

11.1 Sources of Information

The Playas and Playa Lakes CE control and stressor models integrate information from several sources:

- (1) The conceptual model for the North American Warm Desert Playa and Ephemeral Lake conservation element developed for the Madrean Archipelago REA immediately to the west of the Chihuahuan Desert ecoregion (Crist et al. 2014).
- (2) The type descriptions (Comer et al. 2003) for the North American Warm Desert Playa, North American Warm Desert Active and Stabilized Dune, and Western Great Plains Saline Depression Wetland ecological system types (International Ecological Classification Codes CES302.751, CES302.744, and CES303.669, respectively) (NatureServe 2015).
- (3) The type descriptions (Comer et al. 2003) for additional ecological system types that can occur in association with playas and playa lakes in the ecoregion, including North American Warm Desert Pavement (CES302.750), North American Warm Desert Riparian Woodland and Shrubland (CES302.753), North American Arid West Emergent Marsh (CES300.729), and Chihuahuan-Sonoran Desert Bottomland and Swale Grassland (CES302.1504) (NatureServe 2015).
- (4) The Ecological Site Descriptions developed by the U.S. Department of Agriculture, Natural Resources Conservation Service for the Alkali Flats, Gyp Duneland Barrens, Vegetated Gypsum Dunes, Gyp Interdune (Wet), Gyp Interdune (Dry), Gyp Playa, and Salt Flats ecological site types for the Southern Desertic Basins, Plains, and Mountains Major Land Resource Area (MLRA No. 42, site types R042XB001–R042XB005, R042XB008, and R042XB036, respectively) (USDA NRCS 2006; 2016).
- (5) Previous ecoregional assessments (e.g., Dinerstein et al. 2001, NMDGF 2006 WWF-SIA 2007).
- (6) A compendium of scientific research and information about the Jornada Basin within the Chihuahuan Desert ecoregion (Havstad et al., eds. 2006).
- (7) Additional literature on playa and playa lake hydrology, hydrogeochemistry, and biota both across North American in general and within the Chihuahuan Desert ecoregion in particular (e.g., Neal 1975, MacKay et al. 1990, Davis and Hopkins 1992, Haukos and Smith 1992, Rosen 1994, Briere 2000, Hibbs et al. 2000, Muldavin et al. 2000, Brostoff et al. 2001, Allen 2005, Desmond and Montoya 2006, Monger et al. 2006, Rango et al. 2006, Wainwright 2006, Whitford and Bestelmeyer 2006, WWF-SIA 2007, KellerLynn 2008; 2012, Bennett and Wilder 2009, NPS 2010, Boeing et al. 2014, Harings and Boeing 2014).

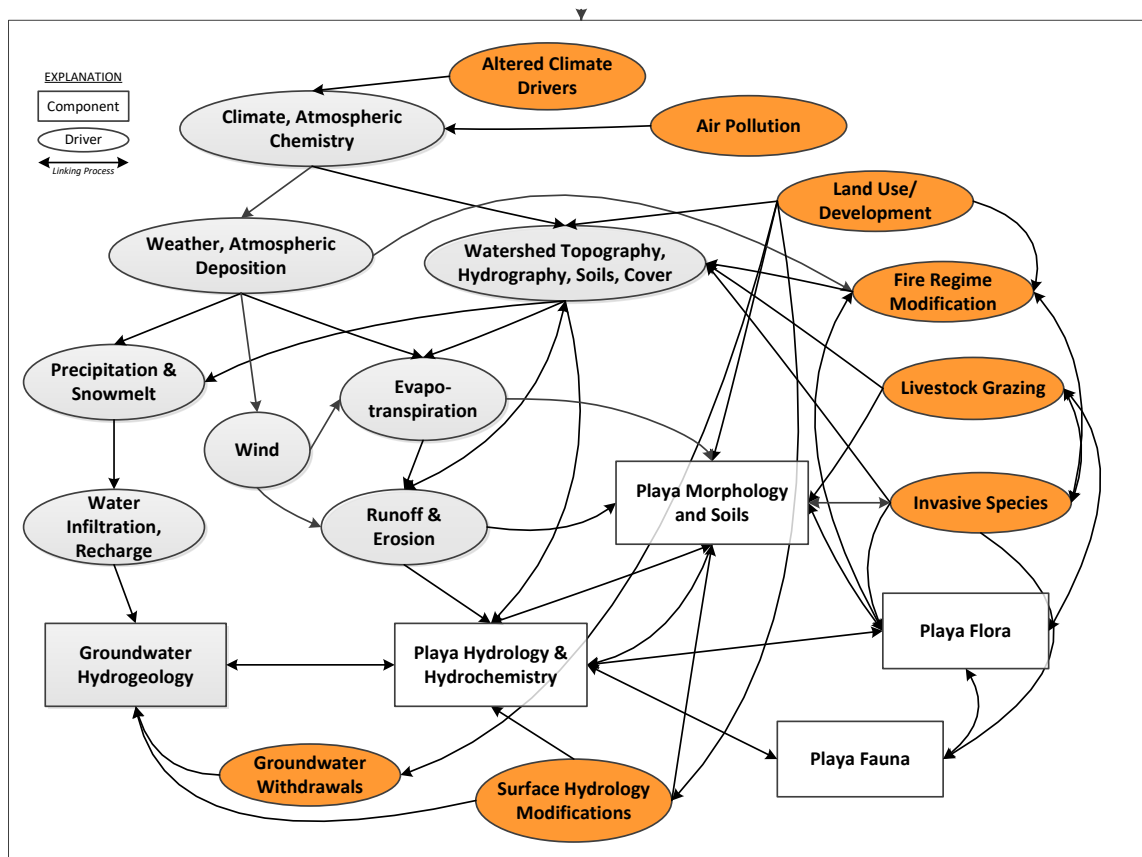
11.2 Playa and Playa Lakes Control Model

Figure 11-1 shows the control model for the Chihuahuan Desert playa and playa lakes system. The model identifies important physical, biological, and ecological characteristics of the resource, its abundance, and its distribution. Anthropogenic and natural drivers of the system are colored orange and grey, respectively. As in the overarching wet system model, arrows represent relationships in which one model component affects or influences another. The stressor model addresses the details of these relationships. Playas and playa lakes in the ecoregion exhibit a range of hydrologic characteristics (duration and consistency of inundation) and exhibit a wide range of chemical properties. This diversity is constrained by the geology (including geochemistry), hydrology, and topography of the region. The

playa and playa lakes system control model specifically identifies the following system components:

- Playa Morphology & Soils** refer to the geomorphology and soil composition of the playa site. Morphology includes depth, shape and areal extent of the playa as well as other aspects of geometry (e.g., manmade dams and diversions) as well as the geological setting they form in. Soil characteristics refer to the surface properties of the soils such as their particle size ranges, their disturbance/erosion and fracturing patterns during drying cycles, and the presence or absence of a surface crust, among other properties. The surface properties of the playas, including the formation of salt crusts, are a major determinant of soil erodibility and the likelihood of desertification (Kocurek et al. 2007, NPS 2010, Baddock et al. 2011, KellerLynn 2012, White et al. 2015).
- Playa Hydrology and Hydrochemistry** refer to the pattern of variation in the area, timing, and duration of wetting of the playas over time (seasonal, annual, and longer-term). Hydrochemistry includes the daily, seasonal, annual, and longer-term variability in playa water quality in terms of presence and concentrations of nutrient and salts and pH. These factors shape the native flora and fauna that occur in the playas in terms of frequency, duration and timing of occurrence (Haukos and Smith 1992).

Figure 11-1. Chihuahuan desert playa and playa lakes system control model.



- Playa Flora and Playa Fauna** refer to the density, composition, and structure of, respectively, the plant and animal assemblages in and on the edges of the playa, including aquatic and terrestrial species. The plant and animal species native to these playas possess unique

adaptations to the hydrology and soil/water chemistry of these environments. Specifically, the playa supports only aquatic species that can tolerate or take advantage of the unique hydrochemistry of the site. Assemblage composition varies along the gradient from the wettest areas in a playa basin to drier areas immediately surrounding the playa. Playas support rich assemblages of insects, reptiles, amphibians, birds, and mammals attracted to the oases created by the presence of water in a desert environment (e.g., Davis and Hopkins 1992, Haukos and Smith 1992, Whitford 2002, Boeing et al. 2014).

Environmental components and natural drivers that shape these system components in turn include the following:

- **Groundwater Hydrogeology** refers to the chemical composition, temperature, pressure, flow rates and other characteristics of groundwater that affect playas and playa lakes. This water may have a distinctive chemical signature. The time from recharge to discharge for an aquifer may span years to millennia. Both basin-wide and local (including dune-field) groundwater tables are important. Basin-scale groundwater discharge may contribute to wetting playas at the lowest points in a basin. Shallow, local groundwater systems may also occur, consisting of perched water tables separate from regional aquifers (e.g., KellerLynn 2012). Fluctuating groundwater levels and influxes of runoff (see below) dissolve and re-precipitate (cycle) salts, including gypsum and selenites, continually reworking the lake bed (e.g., Bennett and Wilder 2009). In the absence of human interventions, the amount of water stored in an aquifer that affects a particular playa reflects long-term averages for precipitation, ground surface conditions, and recharge.
- **Runoff & Erosion** across watershed surfaces delivers surface water to playas and playa lakes, along with sediment, particulate organic matter, and dissolved inorganic and organic matter. Runoff is the most crucial driver shaping playa surface hydrology and soil moisture.
- **Wind** erosion is the most important factor driving basin deflation and dune formation. Shifting and migrating dunes can alter drainage patterns across the landscape (e.g., Bennett and Wilder 2009, KellerLynn 2012).
- **Watershed Topography, Hydrography, Soils, and Cover** affect playas indirectly through their effects on watershed processes that shape groundwater recharge, surface water movement, chemistry, temperature; watershed soil erosion and deposition; and the transport of sediment and organic matter in runoff. However, this environmental element also can affect playas and playa lakes directly by determining the topographic location of the playa, connectivity of playa networks and by shaping the potential for upland wildfires to spread into the vegetation in and around the playa.

The following anthropogenic drivers shape these system components, environmental elements, and natural drivers:

- **Groundwater Withdrawals** alter aquifer system storage and flow gradients in ways that may reduce groundwater discharges to and elevations beneath a playa, and alter its water quality. Since the aquifer systems affecting playas may extend beneath large areas (hundreds to thousands of square km), groundwater withdrawals may affect playas hundreds of km distant from the site(s) of the withdrawals.
- **Surface Hydrology Modifications** include both alteration of playa morphology as well as alteration of surface inflows to playas or playa lakes through impoundment and diversion of water, or through road development, military construction, or installation of culverts and fiber

optic cables (e.g., USAF 1999, KellerLynn 2012). The most common morphologic modification is deepening of the playa basin to increase storage capacity and decrease evaporation. These subject the affected portion(s) of the playa basin to premature and prolonged drying, impeding the growth and development of associated wetland vegetation (Davis and Hopkins 1992, Harings and Boeing 2014). Impoundments and diversions of surface water inflow to a playa for agricultural and other purposes alter the inundation regime and therefore the habitat.

- **Invasive Species** directly alter the composition of the biotic communities within the watershed, including within and immediately around individual playas and playa lakes. Invasive plant species may also indirectly impact playas and playa lakes through alteration of ecological processes including wildfire regimes and grazing management. For example, salt cedar, introduced into the U.S. in the early 1900s as a windbreak and ornamental plant, has invaded the vegetated edges of water bodies throughout the southwestern U.S. (Tamarisk Coalition 2015, see Chapters 2, 3, 8-10). It tolerates salty soils and water, and occurs in – and can dominate – soils around playas normally occupied by native phreatophytes (USDA NRCS 2016). Individual plants can create a vegetative pedestal by drying out the sand in its immediate vicinity, which then erodes away. The resulting “tamarisk pedestals” then trap sand and form dunes in front of the natural dune system, fundamentally altering the morphology of the dune field (KellerLynn 2008).
- **Livestock Grazing** can remove vegetation within the watershed in ways that alter runoff and evapotranspiration rates. Livestock may also directly remove vegetation from playas and/or introduce non-native vegetation. Trampling by livestock can compact playa soils and create micro-topographic inconsistencies in the surface particularly in saturated soils with high clay content. In heavily grazed watersheds, increased sediment input to playa basins from runoff may reduce basin depth (Davis and Hopkins 1992). Cattle trampling also may simply disturb the surface crust (if present), increasing the wind erodibility of the soils (Baddock et al. 2011). Livestock-grazed playas may experience on-site and catchment runoff pollution by animal wastes (e.g., (BLM 2000, Dinerstein et al. 2001, WWF-SIA 2007, USDA NRCS 2016)).
- **Fire Regime Modification**, both through wildfire management and through the effects of altered watershed vegetation and climate, alter the frequency, timing, and severity of wildfires across a landscape. Such changes may affect vegetation around playas directly through changes in the wildfire regime and indirectly through the effects of upland wildfire on the spread of invasive species across watersheds. Fire regime modifications also indirectly affect playas by altering land surface permeability and soil vulnerability to erosion, which in turn affect watershed processes such as recharge and runoff.
- **Land-use and watershed development** directly affects playas when the development occurs directly at and alongside the playa, including recreational development. Human development such as road and railroad grades within or adjacent to playas, can impact hydrology including the inundation regime by unnaturally impounding or diverting water. Military activities at White Sands Missile Range impact gypsum subsystems by creating impact craters from military ordinance (KellerLynn 2012). Conversely, playa dynamics also directly affect development in their vicinity. For example, park infrastructure at WSNM including roads requires regular maintenance to remove sand that accumulates on the pavement surface. Development in playa areas must take into consideration special conditions caused by blowing sand, changing levels of the water table, and the corrosive nature of saline groundwater. At a watershed scale, land-use/development drives the intensity and geography of groundwater withdrawals and location of transportation and other infrastructure corridors. Land-use/development also alters watershed ground cover, land surface permeability, soil vulnerability to erosion, and releases of

chemical pollutants into both watershed soils and watercourses. These effects cascade through the entire ecological system and shape wildfire management policies and actions.

The playa and playa lakes system control model also recognizes the impacts of climate change and air pollution on this system. These drivers affect conditions at playa sites indirectly through their effects on weather and atmospheric deposition, which in turn have cascading effects on upland soils, ground cover, and watershed processes that affect groundwater recharge and surface runoff quantity and chemistry.

11.3 Playa and Playa Lakes System Stressor Model

Table 11-1 presents and defines the drivers, critical environmental elements, critical ecological processes, and ecological outcomes included in the playa and playa lakes stressor model. The stressor model subdivides and categorizes the system components in greater detail than the system control model. The stressor model follows the methodology described in Chapter 4.

Table 11-1. Playa and playa lakes stressor model components.

Model Component	Definition
Drivers	
Air Temperature Regime	The pattern of variation in air temperature, including daily, seasonal, annual, and longer-term variation in central tendencies, maxima, and minima.
Atmospheric Deposition	The pattern of variation in the deposition of potential pollutants from the atmosphere onto the land and water surfaces of the Chihuahuan Desert ecoregion, including variation in pollutant types and in their rates of wet, dry, and total deposition.
Domestic Grazing Management	The pattern of management of the spatial distribution and intensity of domestic livestock grazing immediately at and across watersheds surrounding playas.
Fire Management	The pattern of management of wildfire suppression and prescribed burns immediately at and across watersheds surrounding playas.
Land Use and Watershed Development	The pattern of land use and development of the land surface within and immediately around playas, and across the watersheds surrounding playas, to support human activities involving intentional modification of vegetation, soils, drainage, or topography and/or construction and maintenance of structures and engineered surfaces; and recreational uses.
Non-Native Species Introductions & Management	The types, origins, and patterns of introduction and management (where, when, how) of non-native plant and animal species into the ecoregion. This driver does not include domesticated livestock or species intentionally introduced by fish and game managers for recreational sport.
Precipitation & Snowmelt Regime	The form (rain, ice, snow) and pattern of variation in precipitation, including daily, seasonal, annual, and longer-term variation in magnitude, frequency, timing, and rate (intensity); and the annual pattern of variation in the rate and timing of snowmelt.
Water Management & Use	The pattern of management of surface- and groundwater storage, transport, and use (where, when, at what magnitudes) by public agencies, private organizations, and private individuals, controlled by structures such as dams, diversions, pipelines, and well fields.
Wind Patterns	The pattern of variation in wind forces, including daily, seasonal, annual, and longer-term variation in central tendencies, maxima, and minima.
Critical Environmental Elements	
Runoff Water Quality	The chemical and physical properties of the water that runs off a watershed into a playa, including temperature, pH, and concentrations of dissolved and suspended constituents; and the patterns of variation in these properties, including daily, seasonal, annual, and longer-term variation in their magnitudes.

Model Component	Definition
Runoff Regime	The pattern of variation in the amount of water flowing off the surface of a watershed into a playa or its outflow, including daily, seasonal, annual, and longer-term variation; and including the frequency, timing, and duration of particular high and low flow rates.
Playa Morphology	The overall shape and stability of playas including depths, widths, area-volume relationships, areal extent and artificial features such as impoundments and diversions.
Playa Network Connectivity	Playa networks consist of playas within a landscape, among which overland or surface hydrologic connections may occur depending on landscape-scale hydrologic and ground-cover conditions. The existence and frequency of connections in a playa network affect the ability of playa-adapted species to move among playa sites within the network.
Watershed Erosion	The pattern of daily, seasonal, annual, and longer-term variation in the amount (mass and volume) and particle size distribution of sediment eroded off the surface of a watershed and transported as runoff into a playa.
Watershed Ground Cover	The abundances and spatial distributions of classes of vegetated, disturbed, and artificial surfaces across a watershed that differ in their permeability to water infiltration, hydraulic roughness to water runoff, ability to inhibit soil erosion, and provision of shade (the latter of which can affect runoff temperatures and snowmelt).
Watershed-Scale Groundwater dynamics	The locations and rates of recharge of precipitation to groundwater systems; the storage volumes, inter-connections, and flow path lengths and duration of the aquifers that comprise these groundwater systems; the geochemical and hydrothermal dynamics of these groundwater systems; and the locations and rates of discharge to playas from these groundwater systems.
Critical Ecological Processes	
Aquatic Primary Productivity	The pattern of daily, seasonal, annual, and longer-term variation in the rate of production of biomass in the aquatic food web within the playa through photosynthesis by aquatic flora, including diatoms and algae.
Dune Formation	The pattern of seasonal, annual, and longer-term variation in dune formation driven by wind erosion and deposition.
Fire Regime	The frequency, timing, and severity of wildfires across a landscape, in this case including both the watershed surrounding a playa and the playa itself.
Native-Exotic Species Interactions	The ways, magnitudes, and spatial and temporal extent to which native and exotic species at a playa site compete for habitat, food, and other materials; prey on, infect, or otherwise harm each other; or interact in mutually beneficial ways.
Playa Inundation Regime	The pattern of seasonal, annual, and longer-term variation in playa inundation driven by all hydrologic inputs and affected by evapotranspiration and water use.
Playa Water Quality & Soil Dynamics	The chemical properties of the water and soil at playa site, including temperature, pH, and concentrations of dissolved and suspended constituents; and the patterns of daily, seasonal, annual, and longer-term variation in these properties, including variation over distance.
Ecological Outcomes	
Amphibian Composition	The taxonomic, functional, and size composition; spatial and temporal distribution; abundance; health; and activity level of the amphibian assemblages of a playa site.
Aquatic Invertebrate Composition	The taxonomic, functional, and size composition; abundance; spatial and temporal distribution; and activity level of the aquatic invertebrate assemblage of a playa site.
Biotic Soil Crust Composition & Structure	The taxonomic and functional composition and spatial and temporal distribution of the biota comprising soil crusts including cyanobacteria, green algae, lichens, mosses, microfungi, and other bacteria.
Bird Composition	The taxonomic and functional composition, spatial and temporal distribution, abundance, health, and activity level of the bird assemblage of a playa site.
Reptile & Mammal Composition	The taxonomic, functional, and size compositions, spatial and temporal distributions, abundances, health, and activity levels of the assemblages of reptiles and mammals that occupy or visit a playa site.

Model Component	Definition
Vegetation Composition & Structure	The taxonomic composition, size range, spatial and temporal distribution, and abundance of both emergent (aquatic) and terrestrial vegetation at a playa site.
Insect Composition	The taxonomic, functional, and size composition; abundance; spatial and temporal distribution; and activity level of the insect assemblage of a playa site. (Note: aquatic larvae of insects are addressed as aquatic invertebrates)

Figure 11-2 shows the full stressor model for the playa and playa lakes system in the Chihuahuan Desert ecoregion, built using the model components shown in Table 11-1. Specifically, it displays all model components listed in Table 11-1 along with their causal relationships. Appendix 1 describes and presents the rationale for including every causal relationship in the stressor model, with citations for each causal relationship, providing a comprehensive presentation of the stressor model. Figure 11-2 indicates the presence or absence of causal relationships between the system model components but does not indicate the potential magnitude or other characteristics of these relationships, as explained in Chapter 4.

The stressor model (1) identifies the causal relationships that have affected how the condition of the system likely has changed in the past, in response to changes in its drivers; and (2) provides a means for articulating how the condition of the system will likely change in response to changes in its drivers. As discussed and illustrated below, the stressor model makes it clear: (a) which critical environmental elements would likely be affected by a change in one or more particular drivers, including change agents; (b) which critical ecological processes would likely be affected by the cascading effects of these changes in critical environmental elements; and (c) which system characteristics (ecological outcomes) would likely be affected by the cascading effects of these changes in critical environmental elements and ecological processes. By doing so, in turn, the stressor model also highlights those components of the model – drivers, environmental elements, ecological processes, and ecological outcomes – that demand indicator data.

As defined earlier, a causal relationship exists when a change in one component of the system results in a change in some other component. Change in the first component is said to “cause a change in the second component. Each chain of causation, from driver to outcome, describes how the condition of the system likely has changed in the past, or likely would change in the future, in response to changes in its drivers.

The following paragraphs discuss how each Change Agent affects Chihuahuan Desert playas and playa lakes, as articulated in the stressor model for this CE. A “sub-model” diagram for each Change Agent presents a simplified version of the master stressor model diagram, showing only the direct and indirect effects of the Change Agent of interest. The text accompanying each sub-model diagram summarizes information from Appendix 1 concerning the impacts of each Change Agent, including selected citations. For each Change Agent, the paragraphs summarize present understanding of: (a) which critical environmental elements would likely be affected—directly or indirectly—by each Change Agent and (b) which critical ecological processes would likely be affected—directly or indirectly—by the cascading effects of these changes in critical environmental elements. However, the ecological characteristics of

the system (ecological outcomes) affect each other and are affected by critical environmental elements and ecological processes in the same way regardless of which Change Agent is involved in altering these elements and ecological processes. For this reason, the presentation below begins with a discussion of the interactions among ecological outcomes, and between critical ecological processes and ecological outcomes.

11.3.1 Critical Ecological Processes and Ecological Outcomes

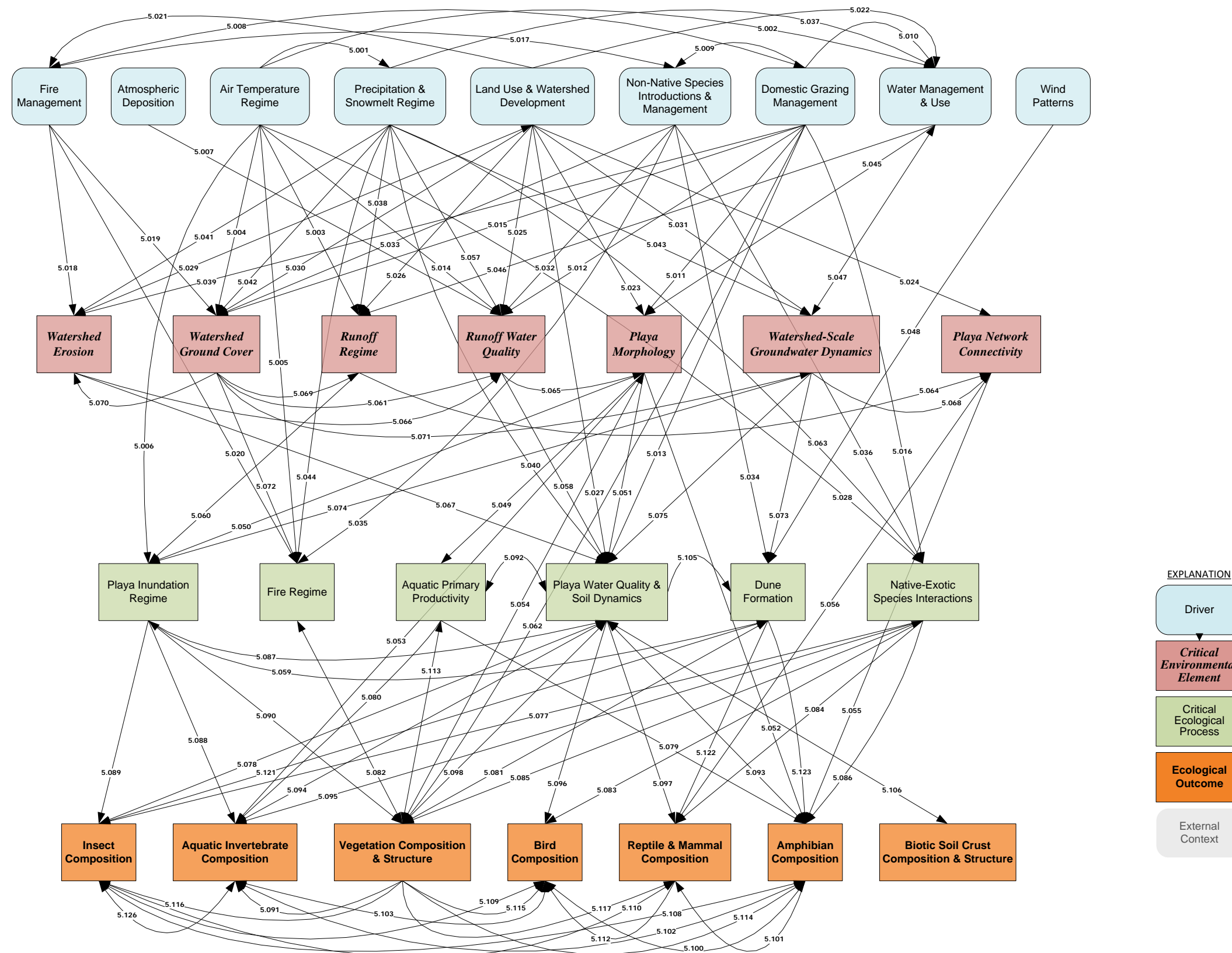
The playa and playa lakes stressor model includes seven ecological outcomes: amphibian composition, aquatic invertebrate composition, biotic soil crust composition & structure, bird composition, insect composition, reptile & mammal composition, and vegetation composition & structure. Table 11-1, above, defines these seven model components. These seven ecological outcomes directly affect each other in numerous ways – and therefore also affect each other indirectly in even more numerous ways – as shown in Figure 11-2. See Appendix 1 for a full presentation of the stressor model. For example:

- Organic matter that falls or gets blown into a playa lake from the vegetation surrounding the water or that gets inundated during an episode of playa inundation can provide food for aquatic invertebrates and amphibian larvae. However, non-native phreatophytes such as salt cedar may produce litter with different characteristics compared to the litter produced by native plants. Conceivably, this litter may function differently as a food resource when it falls into the water, thereby changing the taxonomic and functional composition of the aquatic invertebrate and/or amphibian assemblages (Kupferberg 1997, Wallace and Anderson 2008, Boeing et al. 2014).
- Adult amphibians may use vegetation submerged within and on the edges of playas as shelter and hunting zones, and females may attach egg masses within submerged aquatic vegetation. Immature amphibians may use submerged vegetation for cover for hunting or avoiding predation. The taxonomic composition; size range; spatial and temporal distribution; and abundance of vegetation at a playa site therefore helps shape the taxonomic composition; size range; spatial and temporal distribution; and abundance of amphibians there (Boeing et al. 2014).
- Some insects may use vegetation in and around playa sites as resting, feeding, and mating/egg-laying habitat. The types of plants present at a site therefore presumably strongly shape the taxonomic, functional, and size composition; abundance; and spatial and temporal distribution of the insect assemblage of the locality. Conversely, some riparian insects may play roles in plant reproduction (e.g., pollination, seed burying). Non-native salt cedar may also attract the Tamarisk beetle introduced as a biocontrol (see Chapters 8-9). Consequently, the types of vegetation present in a playa affect and may be affected by the composition, abundance, and spatial and temporal distribution of the insect assemblage at the site (Muldavin et al. 2000, Wallace and Anderson 2008, McCluney and Sabo 2012; 2014, see also Chapters 8-9 for discussion of Tamarisk beetle).
- The composition and structure of vegetation at a playa site help determine the attractiveness of the site to birds. Seed-eating birds in particular have been found to respond to changes in the abundance and distribution of seeds within playas which may be affected by the abundance of seed eating rodents in the habitat as well as rainfall events that may concentrate seeds in low areas within the playa (Raitt and Pimm 1977). Vegetation composition and structure at playa sites also presumably directly determine the availability of nesting, protective, and feeding habitat, and nest materials for both terrestrial and wetland birds at the locality, including migratory birds that use such settings in the ecoregion as stopover habitat, as is the case with

the vegetation along perennial streams in the ecoregion. Non-native phreatophytes such as salt cedar, may provide different habitat conditions and opportunities than those provided by native vegetation, with consequent effects on what birds use affected wetland sites and at what densities. By analogy with other wet system types in the ecoregion, the taxonomic composition, physiognomy and abundance of vegetation around and within a playa site presumably also shape the availability and distribution of protective habitat and herbivore feeding opportunities for both reptiles and mammals of the locality (Raitt and Pimm 1977, Hunter et al. 1985, Johnson and Haight 1985, Kozma and Mathews 1997, Skagen et al. 1998, Muldavin et al. 2000, Hinojosa-Huerta et al. 2004, Rosen and Caldwell 2004, Rosen 2005, Rosen et al. 2005, Brand et al. 2006, NMDGF 2006, Walker 2006, Bateman et al. 2008a; 2008b; 2008c; 2009, Ruth et al. 2010, Nagler et al. 2011, Minckley et al. 2013, Oring et al. 2013, Hagen and Sabo 2012; 2014, Refsnider et al. 2013, Smith and Finch 2014, USDA NRCS 2016).

- Amphibians, some birds, and some reptiles and mammals (including bats) feed on insects. As a result, the taxonomic, functional, and size composition; abundance; and spatial and temporal distribution of the insect assemblage of a playa can affect the abundance, composition, and distribution of amphibians, birds, reptiles, and mammals at the site. Feeding pressure may also affect the composition of the insect assemblage. Additionally, wetland insects may bioaccumulate contaminants that contribute to contaminant body loads in birds that feed on these insects at wetland sites. This set of interactions is better studied along riparian corridors in the ecoregion (see citations) but the concepts apply to playa settings as well (Raitt and Pimm 1977, Kupferberg 1997, Deason 1998, Muldavin et al. 2000, MacRae et al. 2001, Whitford 2002, Rosen and Caldwell 2004, Rosen 2005, Rosen et al. 2005, NMDGF 2006, WWF-SIA 2007, Bateman et al. 2008a; 2008b, Wallace and Anderson 2008, NPS 2010, Hagen and Sabo 2012; 2014, Oring et al. 2013, Boeing et al. 2014, Forstner et al. 2014).
- Amphibians, some birds, and many aquatic invertebrates themselves feed on aquatic invertebrates. The taxonomic, functional, and size composition; abundance; and spatial and temporal distribution of the aquatic invertebrate assemblage at a playa site therefore affects the abundance/frequency of use and species composition of amphibians and birds at the site, and vice versa. Migrating birds (e.g., sandhill cranes) often feed intensively on the branchiopods (e.g., brine shrimp) during stopovers. Additionally, some aquatic macroinvertebrates may compete for particulate organic matter with amphibian larvae. Aquatic invertebrates may bioaccumulate contaminants that contribute to contaminant body loads in amphibians and birds that feed on these insects. Finally, many wetland insect species begin their lives as aquatic larvae. The aquatic invertebrate assemblage of a playa lake/wetland site thus may affect the insect assemblage at the site, and vice versa (Davis and Hopkins 1992, Drewien et al. 1995, Deason 1998, BLM 2000, MacRae et al. 2001, Mora et al. 2002, Whitford 2002, Witte 2005, NMDGF 2006, WWF-SIA 2007, Wallace and Anderson 2008, NPS 2010, Bergeron et al. 2011, Moody and Sabo 2013, Oring et al. 2013, Boeing et al. 2014).
- Seed-eating mammals at a playa site affect the amount of seed available to seed-eating birds (Raitt and Pimm 1977).

Figure 11-2. Chihuahuan desert playa and playa lakes full stressor model.



- Amphibians at playa sites may provide food options for carnivorous birds, such as herons, and reptiles and mammals depending on what amphibians are available, at what times, and in what abundances. The composition and abundance of the amphibian assemblage therefore can affect the composition of the bird and reptile and mammal assemblages at these sites (e.g., Davis and Hopkins 1992, MacRae et al. 2001, Mora et al. 2002, Schmitt et al. 2005, White et al. 2006, Bateman et al. 2009; 2013)

The playa and playa lake stressor model includes six critical ecological processes that directly affect the ecological outcomes discussed above, shown in the following order in Figure 11-2: playa inundation regime, fire regime, aquatic primary productivity, playa water quality & soil dynamics, dune formation, and native-exotic species interactions. Table 11-1, above, defines these six model components. These six critical ecological processes directly affect – and in some cases are also affected by – the seven ecological outcomes in numerous ways, as shown in Figure 11-2 and documented in Appendix 1. Alterations to these critical ecological processes as a result of changes in drivers and critical environmental elements necessarily lead to altered ecological outcomes. The following paragraphs provide examples of the interactions of the six critical ecological processes with ecological outcomes in relatively unaltered systems (see Appendix 1 for full presentation):

- The wetted area, depth, and duration of playa inundation determine the amount, spatial distribution, and quality of wetted area suitable for different aquatic macroinvertebrates that establish in playa habitat, including the aquatic larvae of insects. Insects emigrate early to filled playas, for example, but do not become abundant unless the playa remains filled for an extended period (Sublette and Sublette 1967). Some keystone insects, particularly termites, may inhabit the fringes of playas but not the basins of playas that experience flooding (Whitford and Bestelmeyer 2006) (see also Patrick et al. 1977, Lang and Rogers 2002, NMDGF 2006, WWF-SIA 2007).
- The wetted area, depth, and duration of playa inundation also determine the amount, spatial distribution, and quality of wetted area and areas with shallow water tables suitable for different plants that may establish in playa habitat (Wondzell et al. 1990, Muldavin et al. 2000), and the spatial extent of surrounding land habitable by phreatophytes. Playa sites with higher run-on from the surrounding landscape (and therefore longer inundation periods) have been found to develop high net primary productivity relative to other vegetation types (Huenneke et al. 2002). Conversely, evapotranspiration by aquatic vegetation in playas and by surrounding phreatophytes affects the duration of inundation (see also Davis and Hopkins 1992, Muldavin et al. 2000, Boeing et al. 2014).
- Fire in the vegetation in and surrounding a playa directly affects vegetation composition, structure, and succession at the site. Fire outside of the natural range of variation will change composition, structure, and succession (Peters and Gibbens 2006). Conversely, the vegetation on any part of the landscape affects the fire regime of that part of the landscape (see also Muldavin et al. 2000, and see Chapters 8 and 10 on how altered wildfire affects vegetation along perennial streams and at springs).
- Aquatic primary productivity, including aquatic macrophytes and their litter, provides food and cover for amphibian larvae, and is the foundation of the aquatic invertebrate food web. The rate of aquatic primary productivity at a playa therefore strongly shapes the composition of the aquatic invertebrate and amphibian assemblages at the site (Kupferberg 1997, Lang and Rogers 2002, Whitford 2002, NMDGF 2006, Mitsch and Gosselink 2007, WWF-SIA 2007, Wallace and

Anderson 2008, Porter et al. 2009, NPS 2010, Hershler et al. 2011; 2014, Boeing et al. 2014, Harings and Boeing 2014, Wood et al. 2016).

- Dissolved and particulate organic matter carried into a playa in watershed runoff or submerged during playa inundation of surrounding vegetation (aka allochthonous inputs) provides food for amphibian larvae. The rate of input of allochthonous organic matter at a playa site therefore can shape the composition of its amphibian assemblages. Additionally, amphibians typically have very narrow ranges of tolerance for the chemical properties of water, including temperature, pH, salinity, and concentrations of specific chemical constituents such as metals, organochlorines, etc. These properties presumably can affect amphibian health, development, reproduction, feeding activities, and vulnerabilities to predation; and/or can cause them to depart from or avoid affected playas – or, if they are in fact adapted to these extreme conditions, can allow them to occupy playa sites unsuitable for other species. These combined effects shape the taxonomic, functional, and size composition; spatial and temporal distribution; abundance; health; and activity levels of the amphibians in playas. Reciprocally, amphibians inhabiting a playa may play a significant role in redistributing nutrients in the playa via the mortality of overwintering juvenile toads burrowing in the soils within and surrounding the playa. Given the thousands of juveniles that may die each year, this might be an important mechanism for moving nutrients from areas of high concentration to the surrounding landscape (Davis and Hopkins 1992, MacKay et al. 1992, Witte 2005, NMDGF 2006, White et al. 2006, Whitford and Bestelmeyer 2006, Mitsch and Gosselink 2007, Gregory and Hatler 2008, USFWS 2009, Connally, ed. 2012, Boeing et al. 2014, Bogan et al. 2014, Harings and Boeing 2014).
- Dissolved and particulate organic matter carried into a playa in watershed runoff or submerged during playa inundation (aka allochthonous inputs) also provides food for aquatic invertebrates, including the aquatic larvae of insects. The rate of input of allochthonous organic matter into a playa therefore can shape the composition of its aquatic invertebrate assemblage as well. Aquatic invertebrates also are very sensitive to many chemical properties of water, including temperature, pH, turbidity, salinity, and concentrations of specific chemical constituents such as metals, organochlorines, etc. These properties can affect individual aquatic invertebrate species' health, development, reproduction, feeding activities, and vulnerabilities to predation; can cause them to avoid or flee chemically unusual playas; or, if they are in fact adapted to extreme conditions of water chemistry, can allow them to safely occupy naturally chemically unusual waters at the expense of other aquatic invertebrate species. Natural or anthropogenic elements and compounds in playa water may bioaccumulate in the larvae of playa insects and/or in predatory insects that feed on them, at playa sites with natural or anthropogenic inputs of these bioaccumulative substances (e.g., mercury, pesticides). As a result, invertebrates at playa sites may accumulate relatively high body loads of these substances, although not necessarily to harmful levels. These combined effects shape the taxonomic, functional, and size composition; spatial and temporal distribution; abundance; health; and activity level of the aquatic invertebrate assemblages in playas. In general, however, the aquatic invertebrates of the ecoregion are poorly studied but recognized as highly diverse with a high level of endemism (e.g., Sublette and Sublette 1967, Loring et al. 1988, MacKay et al. 1990, Davis and Hopkins 1992, MacKay et al. 1992, Nash and Whitford 1993, MacRae et al. 2001, Lang and Rogers 2002, Whitford 2002, NMDGF 2006, WWF-SIA 2007).
- Birds may be affected by natural or anthropogenic elements and compounds that bioaccumulate in the smaller fauna on which they prey, in playas with natural or anthropogenic inputs of these bioaccumulative substances. Additionally, birds in the ecoregion may suffer salt toxicosis from use of playas with unusually high concentrations of salts as documented in Laguna Gatuna in Lea County, New Mexico in 1993. This playa was used as a brine disposal site

which led to unnaturally high concentrations of salts (Davis and Hopkins 1992, MacRae et al. 2001, Oring et al. 2013).

- The chemistry of playa waters presumably affects the range of reptiles and mammals that visit or reside around individual wetland sites, as a result of differing tolerances for waters with different temperatures and salinities. However, this review did not locate any studies addressing this possible relationship for or relevant to the ecoregion. The stressor model includes this possible relationship by analogy with the other wet system CEs (e.g., Schmidly and Ditton 1978, MacRae et al. 2001, NMDGF 2006, White et al. 2006, Connally, ed. 2012, Forstner et al. 2014).
- The surface properties of the soils within and immediately surrounding a playa, such as their particle size ranges, their disturbance/erosion and fracturing patterns during drying cycles, and the presence or absence of a surface crust, among other properties impacts the structure and composition of vegetation within and adjacent to the playa. Additionally, native and non-native plant species in the ecoregion have specific ranges of tolerance for soil and water salinity. These circumstances shape the composition and abundance of vegetation at the site. Conversely, salt-adapted species such as salt cedar expel salt, raising soil salinities in their immediate vicinity, further affecting the composition and abundance of vegetation at the site (El-Hage and Moulton 1998, Deloach et al. 2000, Muldavin et al. 2000, Mills 2005, Grunstra and Van Auken 2007, Patten et al. 2008, Baddock et al. 2011, Minckley et al. 2013).
- Biological soil crusts consist of hydrophobic filaments of cyanobacteria and microfungi that wind through the upper two or three millimeters of soil binding soil particles together into aggregates. They thus contribute to playa soil stability. Filaments of cyanobacteria also are hydrophobic, so crusts made of cyanobacteria promote lateral redistribution of water and inhibit infiltration. Finally, the cyanobacteria and microfungi in biological soil crusts take up and release nutrients within playa soils, contributing to nutrient cycling (KellerLynn 2003).
- Dune fields are not typically a suitable habitat for amphibians outside of inter-dunal wetlands. However, dune encroachment on a wetland presumably can make a site uninhabitable for amphibians. On the other hand, the ecoregional fauna include several insects and reptiles tolerant of or adapted to dune conditions, including the Sand-treader camel cricket, Apache pocket mouse, Bleached earless lizard. The ecoregional flora also includes several species tolerant of or adapted to dune conditions, including trees and shrubs, cacti, grasses, and other herbaceous species. Reciprocally, several grass species help anchor dunes and contribute to their stability and growth. Salt cedar, in turn, can form pedestals that alter dune formation. Dune formation at a playa site thus affects both the faunal and floral composition of the site and is affected by the flora in turn (Kelley 1971, Monger 1993, Muldavin et al. 2000, NMDGF 2006, WWF-SIA 2007, NPS 2010, NPS 2016).
- Native-exotic species interactions have pervasive effects across all ecological characteristics of playas and playa lakes, as discussed later in this chapter (see Invasive Species, below).

Two critical ecological processes also affect each other, as shown in Figure 11-2 and described in Appendix 1. Specifically, playa water quality & soil dynamics directly affect dune formation and both affect and are affected by aquatic primary productivity. Playa soil stability significantly affects dune stability: eroded playa soils are the largest source of the mineral particles that form the dunes associated with playas in the ecoregion (Guo et al. 2008, NPS 2010, Szykiewicz et al. 2010). Playa water chemistry defines the availability of nutrients for primary productivity; and primary productivity affects water chemistry through the consumption of nutrients and gases and production of dissolved oxygen (Davis and Hopkins 1992, MacRae et al. 2001, Lang and Rogers 2002, Porter et al. 2009).

Finally, two critical environmental elements, playa morphology and playa network connectivity, each directly affect multiple ecological outcomes, as also shown in Figure 11-2 and described in Appendix 1:

- Aquatic invertebrates and amphibians may have specific habitat requirements for water depths, areal extent of wetted habitat and substrate textures that affect the likelihood that they will occupy parts (or any) of any given playa. Aquatic macrophytes also may have specific habitat requirements for water depths, areal extent of wetted habitat, and substrate textures that affect the likelihood that they will occupy parts (or any) of any given playa. Additional vegetation zones around the most frequently wetted areas will be distributed based on the topography of the local depression in which the playa occurs. Conversely, vegetation can stabilize playa substrates and shorelines from disturbance by runoff pulses (Axtell 1977, Muldavin et al. 2000, Lang and Rogers 2002, NMDGF 2006, Mitsch and Gosselink 2007, Wallace and Anderson 2008, Porter et al. 2009, NPS 2010, Boeing et al. 2014).
- Desert amphibians may move several kilometers to access breeding sites with available water, even within the same breeding season (Boeing et al. 2014). Barriers to animal movement between playas would impact the amphibian composition at each playa site as well as at the larger, landscape scale. Barriers to animal movement between playas similarly would impact ability of reptiles specifically adapted to playa/playa lake and associated dunes conditions to move among suitable sites at the landscape scale (see also Axtell 1977, Whitford and Bestelmeyer 2006).

11.3.2 Climate Change

The final report for the Chihuahuan Desert REA will include a discussion of current forecasts of the ways in which climate change will affect the ecoregion and its ecological resources. The present chapter presents only the conceptual model of the causal relationships and outcomes that potentially will be affected for playa and playa lake sites. The causal relationships discussed here are essentially the same as those discussed for perennial streams, large rivers, and spring-emergent wetland sites in Chapters 8-10, but with emphasis on their effects specifically on playa and playa lake sites.

Figure 11-3 presents the stressor model for playa and playa lake sites in the U.S. portion of the ecoregion, simplified to show only those causal relationships that potentially will be affected by changes in the air temperature and precipitation and snowmelt regimes. Appendix 1 presents the rationale and citations for every causal link shown in the diagram and also provides a larger-format version of Figure 11-3.

Climate change will affect playa and playa lake sites in the U.S. portion of the ecoregion through its effects on two drivers in the playa and playa lakes stressor model: the air temperature regime; and the precipitation and snowmelt regime. Changes in these regimes may include changes in annual and seasonal averages, in the timing and magnitude of annual and seasonal extreme temperatures, and in the timing and magnitude of precipitation, as discussed in Chapter 2. Conceivably, climate change could also affect wind patterns in the ecoregion. However, this is not yet a topic of analysis and forecasting.

Changes in the air temperature and precipitation and snowmelt regimes in turn will directly affect five critical environmental elements in the playa-playa lakes stressor model, as in the stressor models for perennial streams, large rivers, and spring-emergent wetland sites presented in Chapters 8-10:

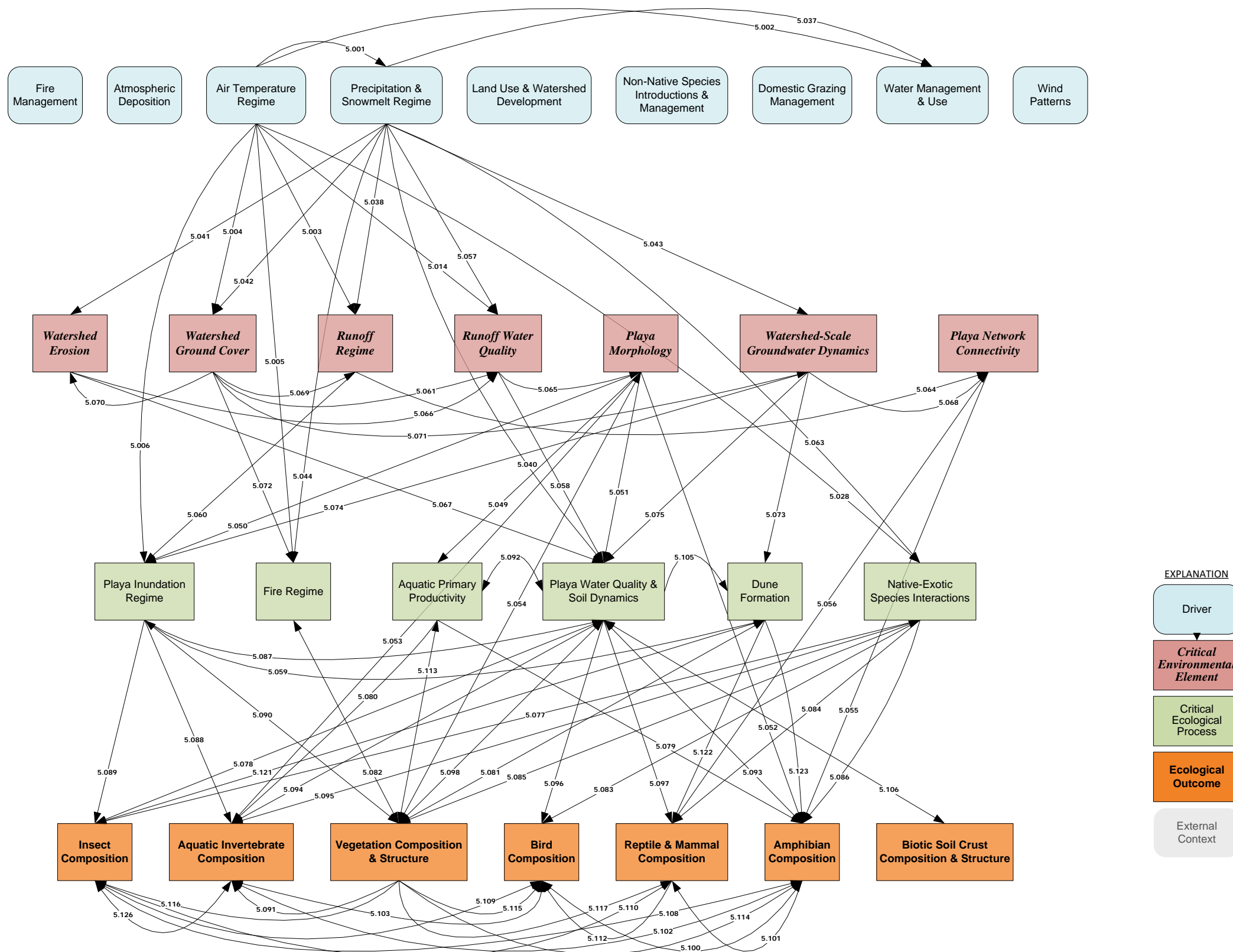
watershed ground cover, watershed erosion, runoff regime, watershed-scale groundwater dynamics, and runoff water quality. The air temperature regime also affects the precipitation and snowmelt regime. Specifically, air temperatures affect whether precipitation falls as rain or snow, whether precipitation even reaches the ground or evaporates as it falls (termed “virga” precipitation), and how much water runs off or infiltrates following precipitation versus simply evaporating.

Changes in the air temperature and precipitation and snowmelt regimes also will directly affect another driver, water management and use. Specifically, changes in air temperatures and precipitation will affect annual and seasonal water supply and demand. The resulting changes in water management and use will have their own, further effects on the runoff regime, watershed erosion, and watershed-scale groundwater dynamics, as discussed in Chapters 8-10 and in the section on the impacts of Development, below, this chapter.

The impacts of changes in the air temperature and precipitation regimes on the five critical environmental elements in the playa and playa lakes stressor model noted above are the same as in the perennial stream systems stressor model presented in Chapter 8:

- Changes in air temperature and precipitation will affect: (1) watershed ground cover by affecting the types, density, and rates of mortality of upland vegetation across a watershed (see Chapters 2-3 and 5-10); (2) runoff water quality by affecting water temperature – which affects other aspects of water quality – and the concentrations of both particulate organic matter (litter) and soluble matter transported in the runoff that contributes to organic inputs to playas (Loring et al. 1988); and (3) the rate at which salts accumulate across soil surfaces as a consequence of natural evaporative processes, and therefore the rate at which such salts are available for dissolution and transport in runoff, further affecting runoff water quality (Manahan 1991, see Chapters 2-3).
- Changes in precipitation, including storm intensity, will affect: (1) the runoff regime by altering the timing, amounts, forms, and rates of accumulation of the precipitation on the watershed surface (see Chapters 2, 3, 8); and (2) the rate and spatial extent of soil erosion caused by individual storm events.

Figure 11-3. Chihuahuan desert playa and playa lakes stressor model: Potential impacts of climate change.



- Recharge to non-alluvial aquifers in the ecoregion mostly takes place at higher elevations across the mountains and foothills, and varies both with the amount of precipitation received and whether the precipitation occurs as rain or snow. Melting snow recharges more effectively than does rainfall. Changes in precipitation therefore will affect the spatial distribution and rates of recharge, which may affect the groundwater dynamics crucial to inundation and soil wetting at some playa sites (e.g., Stonestrom et al., eds. 2007, Wolaver et al. 2008, Porter et al. 2009, USBR 2011, Szykiewicz et al. 2012; 2015a; 2015b, Friggens et al. 2013a, Sheng 2013, Friggens and Woodlief 2014, Jaeger et al. 2014, Eng et al. 2016, Meixner et al. 2016).

Changes in the air temperature and precipitation regimes will also directly affect four critical ecological processes in the playa and playa lakes stressor model:

- Changes in air temperature and precipitation will both directly and indirectly affect playa inundation regimes. Changes in air temperatures will affect the rates of evaporation of water from playa lake surfaces and wetted soils evapotranspiration by playa vegetation (e.g., Serrat-Capdevila et al. 2007, NPS 2010, Tillman et al. 2011, USBR 2011; Friggens and Woodlief 2014). Changes in precipitation patterns will affect the runoff regime and recharge to the groundwater system at higher elevations, as noted above. Playa inundation patterns are determined by watershed runoff and groundwater discharge, as discussed earlier in this chapter. Climate models predict warmer temperatures, including warmer nighttime temperatures, fewer frost days and increased frequency of extreme weather events in the southwestern United States. While total precipitation may not change, increased variability in the amount of precipitation is likely (Kunkel et al. 2013). Therefore, playa inundation may become less predictable for species that depend on predictable fill, such as migratory birds. However, years of high precipitation could favor aquatic invertebrates and primary productivity, and amphibians that can survive years of severely low precipitation (see discussion of Critical Ecological Processes and Ecological Outcomes, above).
- Changes in air temperature and precipitation patterns will affect the probability of wildfires initiated in or spreading into playa sites (see Uncharacteristic Wildfire, below) (Pyne et al. 1996, Luce et al. 2012, see Chapters 2-3 and 5-8).
- Changing precipitation patterns potentially could directly affect playa water quality and soil dynamics by affecting nutrient cycling, moisture retention, and the erosion resistance capacity of playa soils (KellerLynn 2003). Increased thunderstorm activity could also impact soil surfaces by diminishing soil crusts and increasing erosion (see also Bennett and Wilder 2009). Otherwise, changing precipitation patterns will affect playa water quality and soil dynamics indirectly through the impacts of changing precipitation on watershed ground cover, watershed erosion, and runoff water quality discussed above.
- Changes in air temperature and precipitation will directly affect native-exotic species interactions. Air temperature affects water demand in plants and thermal regulation in land animals, and native species may differ in their abilities to adjust to changes in air temperature patterns compared to non-native species. Similarly, precipitation directly on playa and playa lake sites also may affect water availability for both plants and land animals. Native species may differ in their abilities to adjust to changes in precipitation patterns compared to non-native species (e.g., Price et al. 2005, NMDGF 2006, CCSP 2008, Enquist et al. 2008, Jones et al. 2010, Nagler et al. 2011, Connally, ed. 2012, Friggens et al. 2013a; 2013b, Friggens and Woodlief 2014, see Chapter 8).

These direct impacts of changes in the air temperature and precipitation and snowmelt regimes on

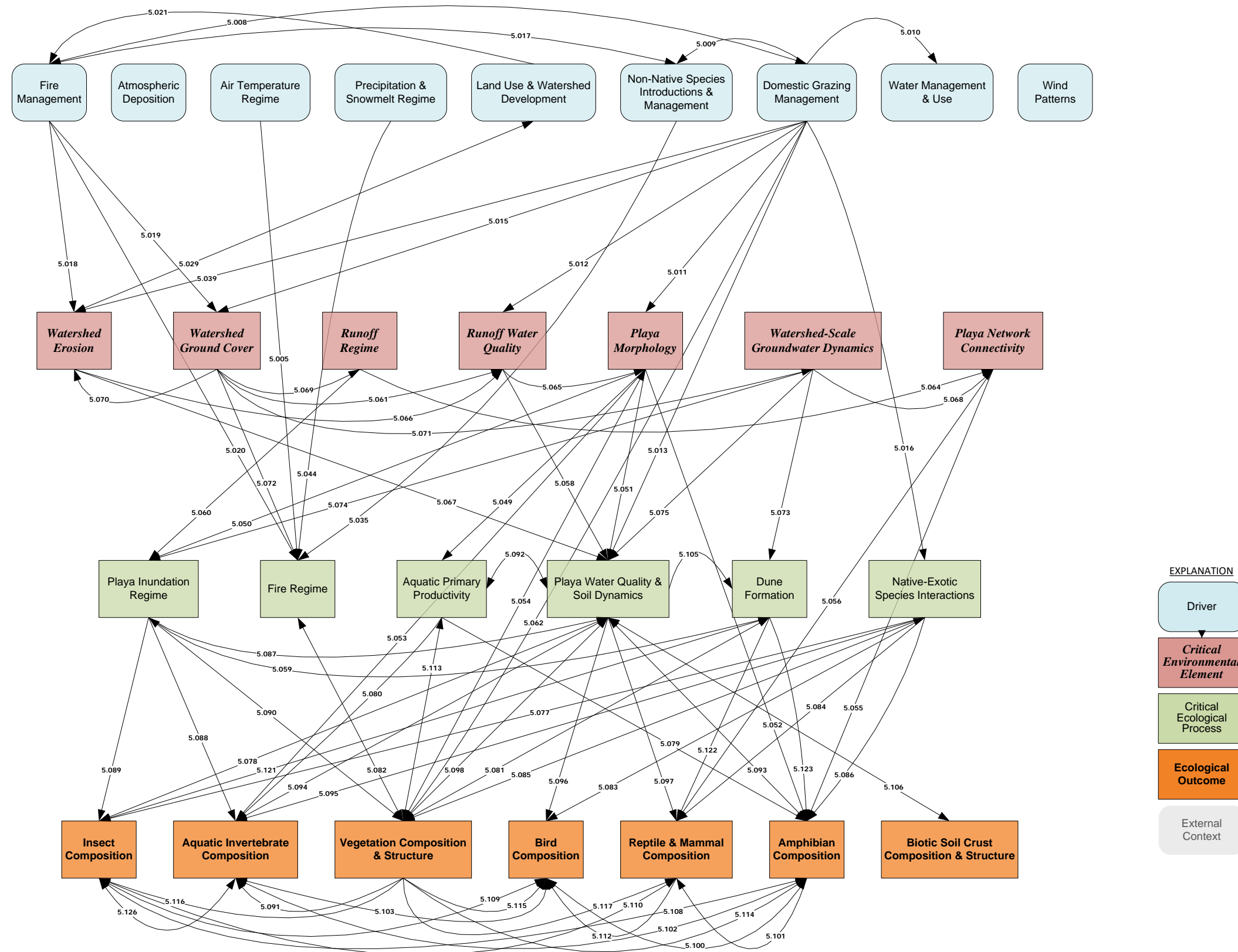
critical habitat elements and critical ecological processes in turn will have further cascading impacts on other critical environmental elements, other critical ecological processes, and all ecological outcomes identified in the playa and playa lakes stressor model. These cascading impacts will occur through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above) (see Appendix 1 for full presentation). For example:

- Changes in recharge to non-alluvial aquifers will result in changes to the dynamics of any aquifers that contribute to wetting and inundation at playa sites, affecting not only individual playa sites but playa network connectivity at the landscape scale. Groundwater elevations at playa sites also affect dune stability and movement, because saturated sediments are more resistant of the erosive forces of wind (Barud-Zubillaga 2000, Bennett and Wilder 2009, Szykiewicz et al. 2010). Changes in the water table could impact dune formation and retention. However, the geologic flow paths that deliver groundwater to playa sites in the ecoregion vary greatly in length, with flow path durations of days to millennia. As a result, the effects on playa inundation resulting from altered recharge will emerge over time spans of years to centuries or even millennia, with each playa uniquely affected (e.g., Heitmuller and Williams 2006, Webb and Leake 2006, Magruder et al. 2009, Porter et al. 2009, Kennedy and Gungle 2010, Tillman et al. 2011, USBR 2011, Friggens and Woodlief 2014, Jaeger et al. 2014, Eng et al. 2016, Meixner et al. 2016).
- Changes in surface-groundwater interactions and in watershed runoff will also affect playa network connectivity at the landscape scale.
- Changes in surface-groundwater interactions at individual playa sites also could affect playa water quality. Aquifer geochemistry and groundwater flow paths and residence times imprint groundwater along different flow paths with different chemistries, which in turn shape the water quality of any playa site supported by groundwater (Alley, ed. 1993, Mills 2005, Miyamoto et al. 2005, Wolaver et al. 2008, Porter et al. 2009, George et al. 2011, Partey et al. 2011, Szykiewicz et al. 2012; 2015a; 2015b, Stafford 2013). However, as noted above, the effects on playa water quality resulting from altered surface-groundwater interactions at playa sites will emerge over time spans of years to centuries or even millennia, with each playa uniquely affected.

11.3.3 Uncharacteristic Wildfire

The fire regime of the U.S. portion of the Chihuahuan desert has changed as a result of the interaction of several drivers, as discussed in several previous chapters. Chapters 2 and 3 discuss the causes and consequences of uncharacteristic wildfire across the ecoregion in general, Chapters 5-7 discuss the causes and consequences of altered fire regimes specifically across the terrestrial ecological systems of the region, and Chapters 8-10 discuss the consequences of uncharacteristic wildfire for perennial streams, large rivers, and springs and emergent wetlands. Figure 11-4 presents the stressor model for playa and playa lake sites in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by uncharacteristic wildfire and excessive domestic grazing. See Appendix 1 for a full presentation of the stressor model.

Figure 11-4. Chihuahuan desert perennial stream stressor model: Potential impacts of uncharacteristic wildfire and excessive domestic grazing.



Four drivers identified in the stressor model directly affect the fire regime at playa and playa lake sites: fire management, the air temperature regime, the precipitation regime, and non-native species introductions. Fire management includes management of fires both immediately at playa sites and across the larger landscape. Fire management practices directly determine whether and how fires at playa and playa lake sites are managed, including the use of prescribed fire, as is the case with fire at sites along perennial streams, along large rivers, and at springs and emergent wetland sites (see Chapters 8-10). Wildfires are closely watched and managed across the ecoregion; and fire managers may also use prescribed burns to help them control the fire regime at individual sites as part of efforts to control or remove non-native vegetation and restore native vegetation, sometimes in conjunction with the exclusion of livestock (Belsky et al. 1999, NMDGF 2006, BLM 2000, Connally, ed. 2012, see Chapters 2, 3, and 5-7).

Air temperature and precipitation patterns affect fire probabilities and intensities at playa and playa lake sites as elsewhere across the U.S. portion of the ecoregion (Pyne et al. 1996, Luce et al. 2012, see Climate Change, above, this chapter). Fire management practices may create conditions that favor invasion or spread of non-native species, and non-native plant species at playa and playa lake sites may have different susceptibilities or adaptations to fire and/or may contribute at different rates to fuel loads, compared to native plants, thus affecting the fire regime (D'Antonio and Vitousek 1992, Steidl et al. 2013, see Chapters 2, 3, 5-7 and the section on Invasive Species, below, this chapter). Fire management practices at the landscape scale also may indirectly affect playa and playa lake sites, as everywhere across the U.S. portion of the ecoregion, by affecting watershed ground cover and watershed erosion patterns, which have their own impacts on playa and playa lake sites (Pyne et al. 1996, Whitford 2002, Havstad et al. 2006, NPS 2010, Luce et al. 2012, see Chapters 2-3). These effects in turn affect the runoff regime and runoff water quality, which affect several critical ecological processes at playa and playa lake sites.

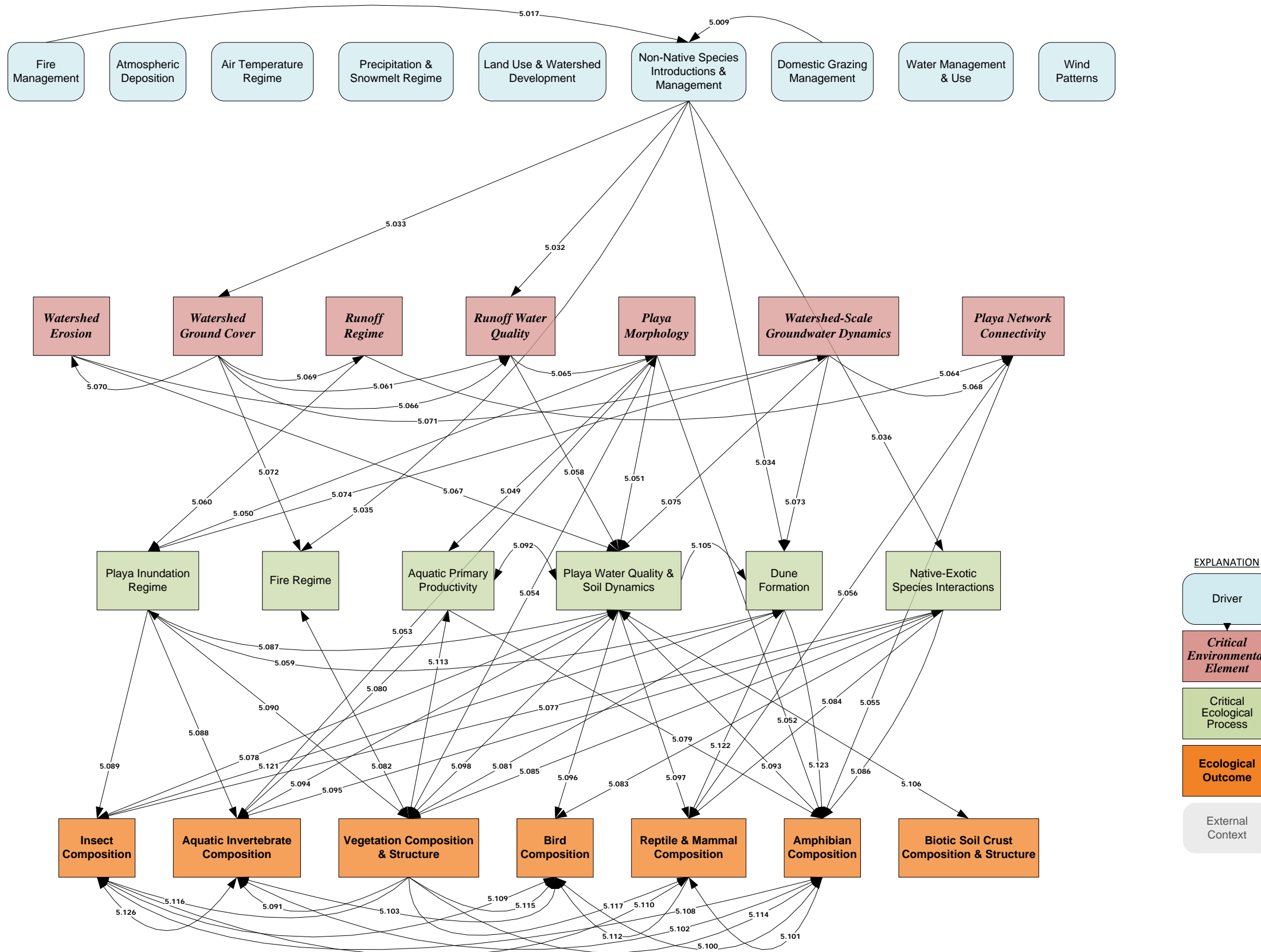
The resulting changes in the fire regime at playas will in turn affect and in turn be affected by the vegetation at playa and playa lake sites across the U.S. portion of the ecoregion (Muldavin et al. 2000, Peters and Gibbens 2006). The consequences of these effects will then cascade through the rest of the ecological outcomes identified in the playas and playa lakes stressor model, through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above). However, the potential effects of wildfire specifically on the ecology of playa sites in the ecoregion have not yet received scientific attention.

11.3.4 Invasive Species

Chapters 2-3 discuss the ecological consequences of non-native species across the U.S. portion of the ecoregion in general, and Chapters 8-10 discuss these consequences for perennial stream, large river, and spring-emergent wetland sites. Figure 11-5 presents the stressor model for playa and playa lake sites in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by non-native species and their management. Figure 11-5 includes a single driver that addresses the ways in which non-native species affect playa and playa lake sites in the U.S. portion of the ecoregion: non-native species introductions. Appendix 1 presents the rationale and citations for each causal link shown

in the diagram associated with this driver. As shown in Figure 11-5, non-native species introductions are directly shaped in part by two other drivers, fire management and domestic grazing management, the effects of which are discussed separately above and below, respectively, this chapter.

Figure 11-5. Chihuahuan desert perennial stream stressor model: Potential impacts of invasive species and landscape restoration.



Non-native species introductions directly shape the ecological status of playa and playa lake sites in the U.S. portion of the ecoregion, at two spatial scales. At the watershed scale, non-native plants affect watershed cover and runoff water quality, as discussed in Chapters 8-10. At the local scale of individual playa sites, in turn, non-native plants directly affect two critical ecological processes: the fire regime, as discussed above (see Uncharacteristic Wildfire, this chapter), and dune formation. Salt cedar creates vegetative pedestals by drying out the surrounding sand. The excessively dried sand then erodes away, altering dune formation patterns and distribution. The resulting pedestals also change local wind patterns, further altering where dunes form (Bennett and Wilder 2009). Otherwise, non-native species introductions affect ecological dynamics locally at individual playa sites through the effects and interactions of individual introduced species with native species. For example (see Appendix 1 for full presentation):

- The extreme conditions of hydrology and chemistry posed by playas and playa lakes make them somewhat less vulnerable to invasions by non-native plants, than is the case with other freshwater ecosystems in the region. Nevertheless, some non-native plants do tolerate or even thrive at playa sites, where they may compete with native plant species for habitat space and materials; affect aquatic substrate conditions; or interact in other ways potentially harmful or beneficial to native phreatic and emergent plants, including competing with native plants for water and, at least in the case of salt cedar, altering soil salinity. In turn, non-native fauna and domestic livestock may feed on native vegetation to different degrees and in different ways than do native fauna or interact in other ways potentially harmful or beneficial to native plants (Hunter et al. 1985, Di Tomaso 1998, Muldavin et al. 2000, Whitford 2002, Fleishman et al. 2003, Sogge et al. 2008, NPS 2010, Rogalski and Skelly 2012, Minckley et al. 2013, see also Excessive Domestic Grazing, below, this chapter, see also Chapter 8).
- Amphibians in the ecoregion, including at playa sites, are affected by non-native diseases such as the now-widespread chytridiomycosis. Amphibians at playa lakes could also be vulnerable to poisoning by the golden alga, which could be carried accidentally into playa sites, where warm, saline waters could provide suitable habitat (Carman 2010, Israël et al. 2014). Non-native amphibians such as the American bullfrog may compete with native amphibians and native aquatic invertebrates for habitat space and materials, including food; prey on natives amphibians; or interact in other ways potentially harmful or beneficial to native amphibians (Rosen et al. 1994, Rosen and Caldwell 2004, Rosen et al. 2005, Witte 2005, NMDGF 2006, Gregory and Hatler 2008, Carman 2010, Hershler et al. 2014).
- Non-native birds may compete with native birds for habitat space and materials, including food; prey on or serve as prey for native birds; interbreed with natives; or interact in other ways potentially harmful or beneficial to native birds. Non-native plant species may alter habitat conditions for native birds, e.g., by excluding plants with which the native birds are associated. Such interactions have not been studied specifically at playa sites in the ecoregion but are proposed in the stressor model by analogy with the impacts of non-native birds in riparian systems (e.g., Raitt and Pimm 1976, Hunter et al. 1985, Blossey 1999, Fleishman et al. 2002; 2003, Mora et al. 2002, Sogge et al. 2008, Merritt and Poff 2010).
- Non-native insects may compete with native insects for habitat space and materials, including food; prey on or serve as prey for natives; infect or otherwise harm natives; interbreed with natives; or interact in other ways potentially harmful or beneficial to native insects. Additionally, non-native vegetation in and surrounding playas may provide habitat and food differentially preferred by different insect species (native and non-native). The non-native Tamarix leaf beetle

(*Diorhabda carinulata*) in fact was introduced to the western U.S. specifically because of its attraction to the non-native salt cedar as its food source, making the beetle a potentially useful biological control agent (e.g., Ellis et al. 2000, Smith et al. 2006, Abelho and Molles 2009, Nagler et al. 2012, Moody and Sabo 2013, Mosher and Bateman 2016). Although not released intentionally here, the beetle has spread throughout the U.S. portion of the Chihuahuan Desert east of the Continental Divide, as of 2015 (Tamarisk Coalition 2015).

- Non-native reptiles and mammals conceivably may compete with native reptiles and mammals for habitat space and materials at playa sites, including food; prey on or serve as prey for natives; infect or otherwise harm natives; interbreed with natives; or interact in other ways potentially harmful or beneficial to native reptiles and mammals. Such interactions have not been studied specifically at playa sites in the ecoregion but are proposed in the stressor model by analogy with the possible impacts of non-native reptiles and mammals in riparian systems (Deason 1998, NMDGF 2006, Whitford and Bestelmeyer 2006, WWF-SIA 2007, Bateman et al. 2008a; 2008b, Connally, ed. 2012, Forstner et al. 2014).

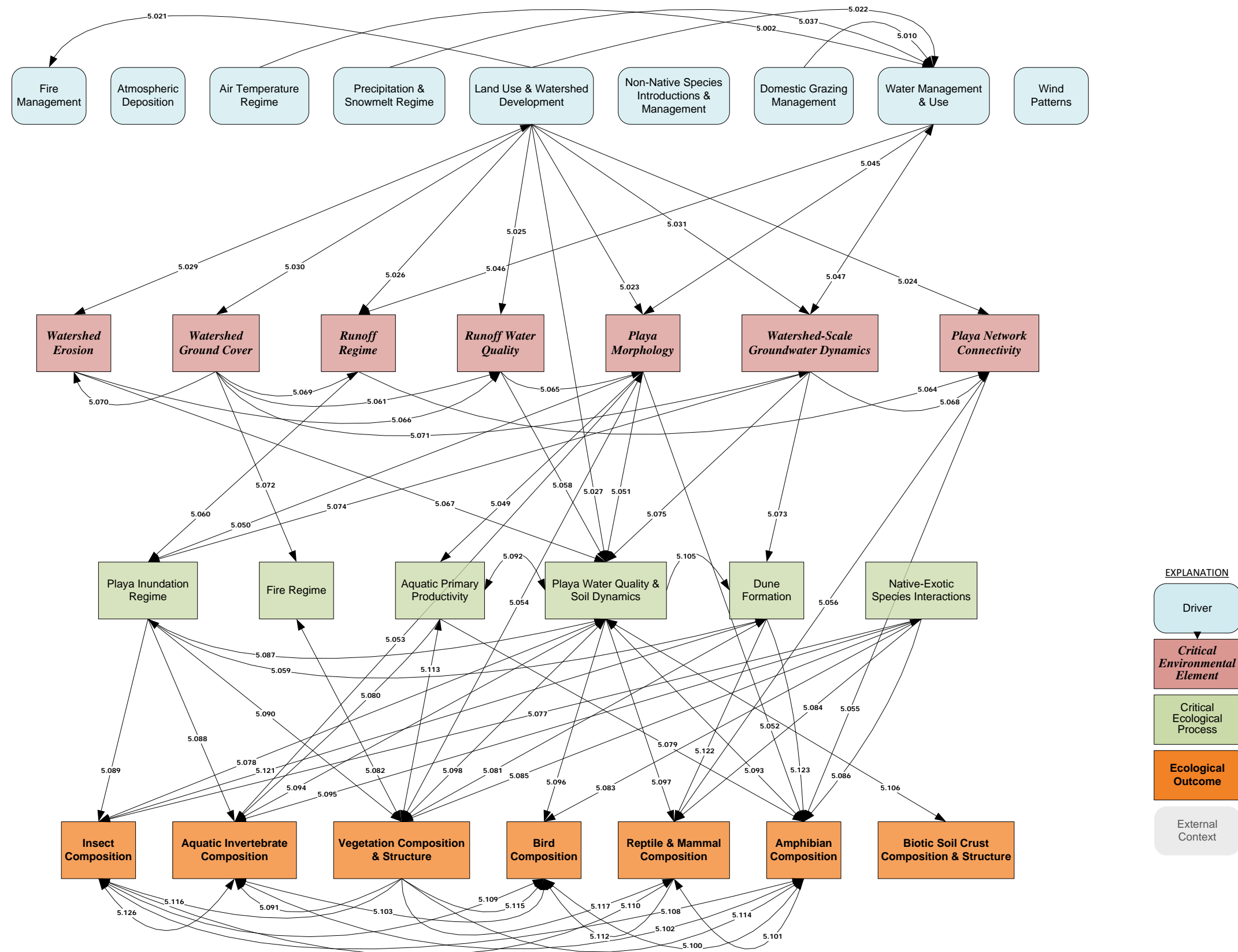
11.3.5 Development

Chapters 2-3 discuss the ecological consequences of land and water development across the U.S. portion of the ecoregion in general. Chapters 8-10 discuss the ecological consequences of land and water development across the U.S. portion of the ecoregion for perennial streams, large rivers, and spring-emergent wetland sites in particular. Figure 11-6 presents the stressor model for playas and playa lakes in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by two drivers that address the impacts of development on this CE: (1) land use and watershed development, and (2) water management and use. Figure 11-6 shows the causal relationships through which these two drivers directly or indirectly affect every critical environmental element represented in the playa and playa lake stressor model, changes in which will affect every critical ecological process and ecological outcome. Appendix 1 presents the rationale and citations for each causal link shown in the diagram.

Water management and use across the U.S. portion of the ecoregion are affected by climate and domestic grazing practices, as discussed earlier in this chapter. Land-use patterns also affect fire management, as discussed earlier in this chapter, and also affect water management and use.

Land and watershed development affect playas and playa lakes across the U.S. portion of the ecoregion at the scales of entire watersheds and individual playa localities. At the watershed scale, land use and watershed development affects six of the seven critical environmental elements identified in the stressor model for playas and playa lakes: watershed erosion, watershed ground cover, runoff regime, runoff water quality, watershed-scale groundwater dynamics, and playa network connectivity. Only one of these six critical environmental elements, playa network connectivity, is unique to the stressor model for playas and playa lakes. Chapters 2-3 and 5-10 discuss the ways in which land use and watershed development affect the other five of the six critical environmental elements affected at the watershed scale (see Appendix 1).

Figure 11-6. Chihuahuan desert perennial stream stressor model: Impacts of land and water development.



Land development across watersheds, including the construction and maintenance of structures and engineered surfaces, impacts the connectivity of playa networks by introducing barriers to animal movement and changing the hydrodynamics of the watershed (Januchowski-Hartley et al. 2013; 2014, Diebel et al. 2015, Fuller et al. 2015, see Chapters 2-3). Land and watershed development also affect playa morphology, at the scale of individual playa localities. Land development may result in the construction of highway and railroad grades through playas. Lordsburg Playa, for example, is fragmented by Interstate Highway 10 and NM State Road 338, as well as by an abandoned railroad grade that cuts across its northern half. The use of playa beds by off-road recreational vehicles (ORVs), including any associated construction of access roads, also modifies playa morphology (BLM 2000, NPS 2010). Playas also have been modified to receive potash process water and oil extraction wastes, municipal wastewater, and cattle feedlot wastes (Davis and Hopkins 1992).

Infrastructure at White Sands National Monument concentrates the effects of. Road culverts accelerate erosion and other man-made structures such as buried fiber optic cables, water lines, roads, and trails also affect erosion. Dissolution around buried water lines and fiber optic cables may cause the formation of sink holes. Parking lots and buildings are also at risk as rainwater and water from other sources dissolves underlying evaporite layers (Bennett and Wilder 2009).

Military operations within White Sands Missile Range, which surrounds White Sands National Monument, may also affect the playa system in the Monument. Military munitions tests are conducted in the Range, which leave unexploded ordinance in playas and dune fields. Impact craters in gypsum-rich evaporites can form fulgurite. Roads and other infrastructure associated with the Range alter surface water flow, inhibiting flow into topographic lows and increasing erosion (KellerLynn 2012).

Water management and use similarly affect playas and playa lakes across the U.S. portion of the ecoregion at the scales of entire watersheds and individual playa localities. At the watershed scale, water management and use can affect groundwater recharge and divert surface runoff into stock ponds and similar detention structures; and can also involve pumping from aquifers that supply individual playas (Hibbs et al. 2000, see Chapters 2-3 and 8-10). Groundwater pumping from the immediate vicinity of playas can have even larger effects on playa hydrology. For example, surface diversion and groundwater withdrawal rates are high in the catchment for the Lordsburg playa complex, primarily due to irrigation farming demand as well as municipal and industrial demand (Allen 2005, Konieczki 2006). A fossil carbon-fired electrical generating facility adjacent to the Lordsburg playa complex uses groundwater for cooling (Konieczki 2006). Excavations of livestock watering holes in playa beds also alter playa morphology and hydrology (Davis and Hopkins 1992, Havstad et al. 2006, NPS 2010 Boeing et al. 2014, see Chapters 2-3).

Watershed land development also affects another driver, fire management. Fire management decisions also must take into account the types and locations of developed land, as these evolve across the landscape and around springs (see Uncharacteristic Wildfire, above, this chapter).

The impacts of land use and watershed development and water management and use on the seven critical environmental elements identified in the stressor model for playas and playa lakes—watershed

erosion, watershed ground cover, runoff regime, runoff water quality, playa morphology, watershed-scale groundwater dynamics, and playa network connectivity—in turn indirectly affect numerous critical ecological processes and ecological outcomes through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above). However, land use and watershed development does directly affect one critical ecological process, playa water quality and soil dynamics. Land development immediately in and around playas can result in releases of chemical pollutants directly into playa sites. In the Southeast corner of New Mexico, for example, potash process water and oil extraction wastes have been discharged into playa lake basins; playa basins have been used as waste effluent ponds by communities; city stormwater is often diverted to playas; and cattle feedlot operations have diverted wastewater into playas (Bristol 1992, Davis and Hopkins 1992, AECOM 2011). Playa soil characteristics may be affected by the use of ORVs within the bed of the playa (BLM 1998, 2000). Organized ORV events are held each year in Doña Ana, Socorro, Otero, Eddy, and Chaves counties in New Mexico, some on playas (NMDGF 2006). While the specific effects of ORV use on playa habitats are poorly understood, ORV travel can cause damage to soils and vegetation and impact wildlife habitat and behavior (Taylor 2006), and possibly leave the surface soils more susceptible to wind erosion (BLM 1998).

11.3.6 Excessive Domestic Grazing

Chapters 2-3 discuss the ecological consequences of excessive domestic grazing across the ecoregion in general, Chapters 5-7 discuss its consequences specifically across the terrestrial ecological systems of the larger landscape, and Chapters 8-10 discuss its consequences for perennial streams, large rivers, and spring-emergent wetland sites. Figure 11-4 presents the stressor model for playas and playa lakes in the U.S. portion of the ecoregion, simplified to show only those causal relationships that potentially will be affected by uncharacteristic wildfire and excessive domestic grazing. Appendix 1 presents the rationale and citations for every causal link shown in the diagram. As shown in Figure 11-4, domestic grazing management in the U.S. portion of the ecoregion affects water and fire management practices. Ranchers across the ecoregion provide water for their livestock by damming intermittent streams to form small ponds, diverting perennial streams to watering structures, and drilling small wells to supply watering tanks. As also shown in Figure 11-4, domestic grazing management in the U.S. portion of the ecoregion both affects and is affected by the spread of non-native vegetation, as discussed in detail in Chapter 5-8. For example, grazed livestock act as vectors for spreading non-native plants.

The impacts of excessive domestic grazing on the critical environmental elements and critical ecological processes identified in the playas and playa lakes stressor model are the same as the impacts of this driver on the equivalent critical environmental elements and critical ecological processes in the perennial streams and springs-emergent wetlands stressor models (see Chapter 8 and 10 and Appendix 1 for full presentation). As noted in those other stressor models, excessive domestic grazing can alter watershed erosion, watershed ground cover, and runoff water quality.

Historic livestock grazing in the Chihuahuan Desert ecoregion has left a number of legacy effects on the desert ecosystem including alterations in ground cover, erosion, runoff patterns, and runoff water quality at the watershed scale that indirectly affect playas and playa lake sites. At the scale of individual

playa sites, livestock grazing management may significantly impact playa plant species composition, when palatable species such as vine mesquite (*Panicum obtusum*) or alkali sacaton grass (*Sporobolus airoides*) are present at a playa site, making such sites important rangeland resources but also vulnerable to effects of excessive grazing (Whitford 2002). Further, trampling by livestock can compact playa soils and create micro-topographic inconsistencies in the bed surface, particularly in saturated soils with high clay content. In some heavily grazed areas, increased sediment input to playa basins from runoff may reduce basin depth. Cattle trampling may also simply disturb the surface crust (if present) of some playas and increase the wind erodibility of the soils. Livestock grazing may also require physical alterations to playas to retain water longer for livestock watering (see Development, above) (Davis and Hopkins 1992, Belsky et al. 1999, BLM 2000, Baddock et al. 2011, see Chapters 2, 3, 8-10).

The direct impacts of excessive domestic grazing on the critical environmental elements and critical ecological processes identified in the playas and playa lakes stressor model in turn affect numerous other critical ecological processes and all ecological outcomes through the causal relationships described earlier (see Critical Ecological Processes and Ecological Outcomes, above).

11.3.7 Landscape Restoration

Chapter 3 discusses the types of landscape restoration projects taking place in the U.S. portion of the ecoregion, some of which currently or in the future could affect playa and playa lake sites just as they currently or potentially could affect perennial streams, rivers, and springs in the ecoregion. These projects include efforts to remove non-native aquatic and riparian species, and to ensure the replacement of removed non-native vegetation with native vegetation.

Very little information is available regarding playa restoration projects in this ecoregion and the examples are mostly for very small projects. For example, the city of Las Cruces, New Mexico, has undertaken a playa restoration project as mitigation for some of the unintended habitat modifications associated with the construction of the Las Cruces Dam in the 1970s. This restoration includes the planting of vegetation based on the assumption that the existing barren playas are seed limited. The goal for this restoration project is to improve wildlife habitat in an otherwise barren landscape (USACE 2011).

Other efforts to restore playa sites in the ecoregion potentially could include the following (see Figure 11-5 and Appendix 1):

- Restoration of surface and/or groundwater hydrology through changes to water management and use.
- Removal of road, railroad grade, or other types of barriers to connectivity and hydrologic integrity within individual playas and across larger playa network.
- Exclusion of livestock and/or ORVs (e.g., BLM 2000).
- Prohibition or control of waste disposal into playa sites, with or without removal of existing contaminants.
- Removal of non-native vegetation, with or without active restoration of native vegetation in its place. The list of non-native species targeted for control or removal includes salt cedar (*Tamarix* spp.; Muldavin et al. 2000). Removal methods may include biological control, prescribed fire,

mechanical removal, and chemical control, and these methods can have their own effects on playa biota (e.g., Nagler et al. 2012, Goolsby et al. 2016).

Regardless of scale or desired outcome, each restoration project should have clearly defined objectives with an analysis of desired and potential unintended consequences of the restoration project. Most importantly, understanding how the restoration may impact the hydrodynamics of the playa is critical since the various species that inhabit the playa will be differentially affected by the changes in inundation period (WWF-SIA 2007). Figure 11-5 clearly shows the cascade of effects to ecological processes and outcomes that might stem from alterations to one or more of the environmental elements through the process of restoration.

11.4 Playa and Playa Lakes Key Ecological Attributes

As noted earlier, all ecological outcomes and critical ecological processes in a system stressor model constitute key ecological attributes for the system. The list below identifies 12 key ecological attributes for the Chihuahuan Desert playa and playa lakes system based on these criteria. Characterizing the present condition of a system requires data on indicators for its key ecological attributes. The definitions for the key ecological attributes are the same as the definitions for these model components presented above.

- **Ecological Outcomes**
 - Amphibian Composition
 - Aquatic Invertebrate Composition
 - Bird Composition
 - Insect Composition
 - Reptile & Mammal Composition
 - Vegetation Composition and Structure
- **Critical Ecological Processes**
 - Aquatic Primary Productivity
 - Fire Regime
 - Native-Exotic Species Interactions
 - Playa Inundation Regime
 - Playa Water Quality & Soil Dynamics

12 Pronghorn Conceptual Model

This chapter presents the conceptual ecological model for the pronghorn, *Antilocapra americana* (Ord 1815, Brown and Ockenfels 2007). Pronghorn range from Alberta and Saskatchewan south through the western United States, west of the Mississippi, into portions of California, through portions of the mountain and Great Plains states, and south into Arizona, New Mexico, Texas, and Mexico. They are absent from mountainous areas and inhabit primarily flat prairies, shrub steppes, and semiarid grasslands (Yoakum 2004a). Five subspecies have been described. In the Chihuahuan Desert, *Antilocapra americana mexicana* has been described but there is controversy over all of these subspecies and genetic evidence indicates they may be clines rather than subspecies (O’Gara and Janis 2004). The presentation of the pronghorn conceptual model follows the structure described in Chapter 4, with sections on sources of information, a species overview, the stressor model, and key ecological attributes. As noted in Chapter 4, the conceptual models for species and species assemblages do not include a separate control model.

12.1 Sources of Information

The pronghorn overview and stressor model integrate information from several sources, including summaries of the ecology of the species both in general across its entire range and specifically within the Chihuahuan Desert (e.g., Buechner 1950, Nelson et al. 1999, Morris 2003, Yoakum 2004a; 2004b; 2004c; 2004d; 2014, Nelle 2006, Richardson 2006, Brown and Ockenfels 2007, Tluczek 2012, Arizona Game and Fish Department 2013, NatureServe 2015).

12.2 Pronghorn Overview

The Chihuahuan Desert is a biologically diverse ecoregion and an important part of the known historic range of pronghorn. The Chihuahuan Desert has a diversity of natural communities, several of which are threatened by increasing development and fragmentation, changes in climate and a shift from grasslands to more shrub dominated communities. The ability of pronghorn to continue to exist within the Chihuahuan Desert will depend on landscape scale actions to assure that sufficient habitat exists within the changing range of variation of drivers, critical environmental elements and critical ecological processes resulting from current and future threats.

12.2.1 Distribution

Pronghorn are wide ranging animals, traveling in herds and are highly visible because they occupy open habitat. Pronghorn are also an important prey species for several predators as discussed below. Competition between pronghorn and other native ungulates appears to be minimal, though there is dietary overlap with mule deer (Yoakum 2004b). Pronghorn prefer low vegetation, feeding primarily on forbs and small shrubs and eat very little grass, yet they require cover for fawns that are nearly immobile shortly after birth. Pronghorn are the fastest land mammal in North America, capable of reaching speeds in excess of 50 mph.

Pronghorn are an important game species, and hunting provides economic benefits to landowners and

area commerce. They forage on a diversity of plants, the abundance of which is highly variable in the Chihuahuan Desert. Years with high precipitation provide abundant food sources while drought can result in limited resources for pronghorn. They also can be limited by human created barriers, particularly fences, so development activities reduce their ability to find and occupy suitable habitat. Habitat has been degraded with a continuing increase in the abundance of shrubs due to historic grazing and fire suppression (Arizona Game and Fish Department 2013).

12.2.2 Habitat

Pronghorn require opportunities for foraging, sufficient water, and terrain that allow for safe habitat selection and opportunities to see predators and escape from predation (Morris 2003, Yoakum 2004c, Yoakum et al. 2014). Pronghorn prefer primarily open, even terrain with sparse shrubs, few trees and primarily grasses and forbs. They avoid steep or broken terrain, canyons, and steep slopes (Brown and Ockenfels 2007).

The Chihuahuan Desert is composed of several ecological systems consisting of shrublands, mixed shrub and grass lands and grasslands. The dominant plant species throughout the Chihuahuan desert is creosote bush (*Larrea tridentata*). Depending on latitude and other factors, other co-dominant shrubs include tarbush (*Flourensia cernua*), honey mesquite (*Prosopis glandulosa*), four-wing saltbush (*Atriplex canescens*), and several agave and yucca species, including the endemic lechuguilla (*Agave lechuguilla*), candelilla (*Euphorbia antisiphilitica*), ocotillo (*Fouquieria splendens*), and sotol (*Dasyilirion* spp.). Cacti include cholla (*Cylindropuntia* spp.), opuntia (*Opuntia* spp.), barrel cactus (*Ferocactus* spp.) and Arizona rainbow cactus (*Echinocereus polyacanthus*) (Yoakum 2004d). Most studies classify cacti as forbs (Yoakum 2004c). Grasslands include these shrubs and a mixture of grasses including side oats grama (*Bouteloua curtipendula*), black grama (*Bouteloua eriopoda*), tobosa grass (*Hilaria mutica*), and purple three-awn (*Aristida purpurea*) (NatureServe 2015).

The diversity of these shrub and forb assemblages provides highly nutritional food for pronghorn, which have been found to use 160 species of forbs, 53 species of shrubs and 15 species of grasses. In the Trans-Pecos region, pronghorn are fond of flowers and fruit and consume five species poisonous to livestock (Yoakum 2004d). The specific species used varies between seasons (Buechner 1950, Nelle 2006, Tluczek 2012, Yoakum 2004d). In the Trans-Pecos, stemmed bitterweed (*Actinea linearis* var. *scaposa*), cutleaf daisy (*Aplopappus spinulosus*), side-oats grama (*Bouteloua curtipendula*), blue grama (*Bouteloua gracilis*), black dalea (*Dalea frutescens*), tall buckwheat (*Eriogonum tenellum*), scarlet beeblossom (*Gaura coccinea*), deervetch (*Lotus oroboides*), paper flower (*Psilostrophe tagetina*), coneflower (*Ratibida columnaris*), and woolly senecio (*Senecio longilobus*) were used throughout the year (Buechner 1950). At White Sands Missile Test Range, species included Bigelow sage (*Artemisia bigelovii*), Sonoran scrub oak (*Quercus turbinella*), littleleaf sumac (*Rhus microphylla*), canaigre dock (*Rumex hymenosepalus*), *Ephedra* spp., *Yucca* spp., *Opuntia* spp., several grass species and the forbs such as silver nightshade (*Solanum elaeagnifolium*), globemallow (*Sphaeralcea* spp.), *Croton* spp., Indian rushpea (*Hoffmannseggia glauca*) and other forbs (Avery 2012). Succulents are consumed in low abundance, and seem to be consumed during dry periods (Tluczek 2012, Yoakum 2004d). Perennial forbs are most important as they are available year-round compared to annual forbs available after

precipitation events. Browse or shrub species are used in the spring and summer or when forbs are not available. Grasses are least used because pronghorn digestive systems are best adapted to digesting forbs and fresh shoots of shrubs and are inefficient for digesting the low-quality fibrous forage of grasses (Richardson 2006).

In the Chihuahuan Desert, most wildlife including pronghorn are found in the mid-seral, grass-shrub community. This community consists of widely scattered shrubs that provide food for pronghorn but allows free movement and safe habitat selection where pronghorn can see, avoid and escape predation (Nelson et al. 1999). In the Trans-Pecos region, pronghorn avoid areas with vegetation greater than 24 to 30 inches tall and shrub cover greater than 10-20% (Brown and Ockenfels 2007). Optimal vegetation heights, including grasses and shrubs, were found to be 10-18" (Richardson 2006, Yoakum 2004c). Pronghorn avoid areas of bare soil (Brown and Ockenfels 2007). Some types of terrain such as steep slopes, woodland areas and human uses represent barriers to movement. Pronghorn will use slopes less frequented by cattle where flatter areas have been overgrazed. Pronghorn may also prefer south facing slopes for shelter from cold winds and where food availability is greater (Buechner 1950).

Mating occurs in late summer with the birth of fawns in spring. Fawns are born in March or April in southwestern deserts. Fawns are generally immobile during the first week or so with the doe providing milk. Fawn mortality is high, depending on food availability and predation. Bucks expend a great deal of energy defending territory as part of fall breeding and may have little fat reserve for winter resulting in a decrease in male pronghorn abundance (Brown and Ockenfels 2007). Cover for fawns is critical to reproductive success. Fawn survival is considered the most important contributor to overall abundance (Richardson 2006). Safe habitat selection for fawn bedding sites includes a height of the grass and forb assemblages of 10-22 inches with scattered plants in the shrub and succulent assemblages. An overabundance of shrubs provides stalking cover resulting in increased predation of fawns (Richardson 2006).

Pronghorn also select areas close to water with most found within two miles of a water source. As stated above, pronghorn can get much of their water from preformed water within the forb, shrub and succulent assemblages. However these sources may not be sufficient for lactating does (Richardson 2006).

While pronghorn in other parts of their range may migrate up to 200 miles to avoid deep winter snows, pronghorn in the Chihuahuan Desert are not migratory. However, they do move in response to seasonal availability of forage. Movement of only five or ten miles may be critical for foraging during dry periods when the abundance of the forb assemblage has declined or is insufficient and access to woody browse plants on an adjacent range becomes necessary for survival (Texas Parks and Wildlife Department 2010).

Predation is a major source of mortality for both adults and juveniles. Adult pronghorn are preyed upon by mountain lions (*Puma concolor*) and coyotes (*Canis latrans*), while fawns are preyed upon primarily by coyote, bobcat (*Lynx rufus*), and golden eagle (*Aquila chrysaetos*) (O'Gara and Janis 2004, O'Shaughnessy et al. 2014). Fawn mortality may be as high as 50-80%, primarily in the first 30 days of life (Yoakum et al. 2014). Fawns born during peak fawning period have a higher survival rate than those

born off peak. Barriers to movement can result in increased predation as predators may trap pronghorn that can't cross barriers, particularly fences (Sullins 2002). Human efforts to provide water for wildlife or cattle may also augment predation because livestock tanks provide opportunities for predators, particularly if vegetation conceals them (Tluczek 2012).

Terrain and the height and density of the grass and shrub assemblages that limit movement can combine to contribute to predation. Mountain lion kills in central AZ were documented within 100 meters of rugged terrain such as canyons or rock outcroppings and in grasslands or juniper woodlands but not in short-grass prairie (Ockenfels 1994). Coyote predation was positively correlated with the height and density of vegetation (Yoakum 2004c). Terrain and the shrub assemblage are also important in providing thermal cover with ridges, draws and swales providing protection from storms in the Trans-Pecos (Richardson 2006)

Pronghorn have similar food preferences with native ungulates such as mule deer (*Odocoileus hemionus*) and white-tailed deer (*Odocoileus virginianus*), but there appears to be little interaction or competition for resources between deer and pronghorn. Bighorn sheep (*Ovis canadensis*) have a similar diet, consuming primarily forbs and shrubs and little grass (Brewer and Harveson 2007), but they use different habitat (Yoakum 2004b). Prairie dogs (*Cynomys ludovicianus*) appear to consume fewer species in the forb assemblage preferred by pronghorn (Yoakum 2004b).

Pronghorn are subject to numerous diseases and parasites (Brown and Ockenfels 2007; Buechner 1950). Mortality is generally low, but adults and young can be weakened by diseases and parasites, making them more subject to predation. Periodically, there are major die-offs. During the 2009 die off in the Trans-Pecos, large numbers of pronghorn had excessive loads of *Haemonchus contortus* or barber pole worms, a parasitic round worm (Harveson et al. 2012a). The competitor assemblage may be a source of diseases and parasites. Pronghorn and cattle may share diseases, but there appear to be no significant disease agents that they pass between them, though cattle may be a reservoir for bluetongue which is fatal to pronghorn (Zwartjes et al. 2005). Pronghorn have higher incidences of parasites where sheep also graze (O’Gara 2004).

12.3 Pronghorn Stressor Model

Table 12-1 presents and defines the drivers, critical environmental elements, critical ecological processes, and ecological outcomes represented in the pronghorn stressor model. The stressor model follows the methodology for species described in Chapter 4.

Table 12-1. Pronghorn stressor model components.

Model Component	Definition
Drivers	
Fire Management	Refers to any human activities to control the size, timing, intensity, or frequency of fires. Fire management applies to the control and prevention of wildfire as well as the use of prescribed fire to achieve management goals and objectives.

Model Component	Definition
Grazing Management	Refers to human activities to manage and control where and how domestic livestock are grazed within the Chihuahuan Desert. This includes the number of livestock grazed per hectare as well as any grazing rotation strategies.
Invasive Species Management	Refers to any human activities to reduce the abundance of invasive species in the Chihuahuan Desert, with a primary focus on invasive grasses. This management may include mechanical removal, herbicide application, and native grass plantings. Invasive species management may overlap with fire management, grazing management, and wildlife management and landscape restoration when any of those activities also impact invasive species.
Land Conversion and Development	Refers to any human activity that converts native habitats (shrubland or grassland) into developed areas. Developed areas include urban, industrial, suburban development, areas of intense recreational activity, military activities, borderland security activities as well as rangelands used for grazing or agricultural croplands. This factor also refers to threats posed by human development including barriers to movement, collisions with motor vehicles, attraction of predators and other related threats. Developed areas include urban, industrial, and suburban development as well as rangelands used for grazing or agricultural croplands.
Legacy Effects of Historic Grazing	Refers to any ongoing impacts to vegetation community composition and structure, especially increasing conversion of grasslands to shrublands, resulting from the legacy effects of unsustainable domestic livestock grazing in the 19th and early 20th centuries.
Precipitation Regime	Refers to the pattern, timing, intensity, and average annual rate of precipitation in the Chihuahuan Desert within and across seasons.
Temperature Regime	Refers to the patterns of air temperature in the Chihuahuan Desert within and across all seasons.
Wildlife Management and Landscape Restoration	Refers to human activities to manage for wildlife and restore habitat including vegetation management to alter vegetation structure and composition, providing water sources, hunting, and predator control. This driver overlaps with fire management, grazing management and invasive species management.
Critical Environmental Elements	
Competitor Assemblage	Refers to the composition and abundance of wild and introduced animals that forage on the same plant species as pronghorn.
Fire Regime	Refers to the frequency, intensity, severity and seasonality of both wildfire and prescribed fire.
Forb Assemblage	Refers to the composition and abundance of herbaceous plants in the forb guild.
Grass Assemblage	Refers to the composition and abundance of grasses.
Grazing Intensity	Refers specifically to the number of livestock grazed per acre and the frequency of grazing rotations. In the literature, intensity is generally characterized as light to non-use, conservative, moderate, heavy and severe.
Predator Assemblage	Refers to the composition and abundance of animals that kill pronghorn.
Shrub Assemblage	Refers to the composition and abundance of woody plant species.
Succulent Assemblage	Refers to the composition and abundance and species of succulents (e.g., cacti), which are an important food and water source for pronghorn
Terrain	Refers to landscape features including rock formations, cliffs, slope, aspect and elevation of specific areas.
Water	Refers to sources of preformed, metabolic and free water.
Critical Ecological Processes	
Disease and Parasites	Refers to any condition affecting the health of individual pronghorn, including internal and external parasites, bacterial, fungal and viral infections, and environmentally-based toxins.
Foraging	Refers to the ability of pronghorn to actively find and consume food resources within their habitats

Model Component	Definition
Movement	Refers to movement by pronghorn between patches of habitat to avoid predators, for foraging, and for reproduction including using different terrain in different seasons. Movement may be affected by areas developed for agriculture or urban uses, as these areas provide unsuitable habitat, potential threats from vehicles, altered predator densities and barriers to movement.
Predation	Refers to mortality that pronghorn suffer due to predators such as mountain lions and coyotes.
Safe Habitat Selection	Refers to the preferential use of habitat to avoid predators, for thermoregulation, and to access water.
Ecological Outcomes	
Abundance	Refers to population size, ratios of bucks to does and does to fawns, and spatial distribution of pronghorn.
Reproductive Success	Refers to the ability of pronghorn to successfully give birth and raise young to self-sufficiency.

Figure 12-1 shows the full pronghorn stressor model, displaying the model components listed in Table 12-1 along with their causal relationships. Appendix 1 describes and presents the rationale for including every causal relationship in the stressor model, with citations for each causal relationship, providing a comprehensive presentation of the stressor model.

Figure 12-1. Pronghorn stressor model.

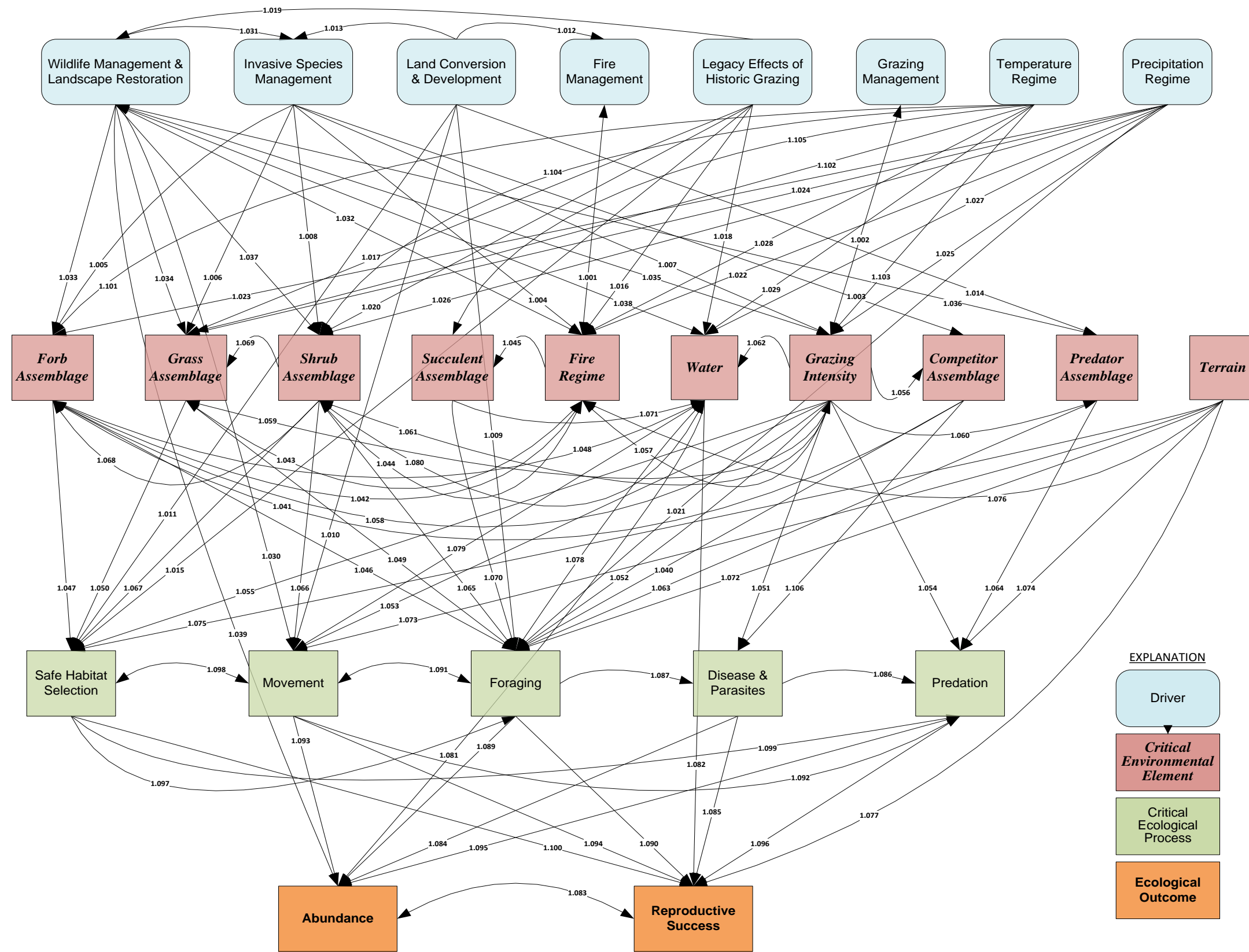


Figure 12-1 indicates the presence or absence of causal relationships between the system model components but does not indicate the potential magnitude or other characteristics of these relationships, as explained in Chapter 4. Figure 12-1 identifies the causal relationships that have affected how the system likely has changed in the past, in response to changes in its drivers. The diagram provides a means for articulating how the condition of the system will likely change in response to changes in its drivers. As discussed and illustrated below, the stressor model makes it clear: (a) which critical environmental elements would likely be affected by a change in one or more particular drivers, including change agents; (b) which critical ecological processes would likely be affected by the cascading effects of these changes in critical environmental elements; and (c) which system characteristics (ecological outcomes) would likely be affected by the cascading effects of these changes in critical environmental elements and ecological processes. By doing so, in turn, the stressor model also highlights those components of the model – drivers, critical environmental elements, critical ecological processes, and ecological outcomes – that demand indicator data.

As defined earlier, a causal relationship exists when a change in one component of the system results in a change in some other component. Change in the first component is said to “cause a change in the second component. Each chain of causation, from driver to outcome, describes how the condition of the system likely has changed in the past, or likely would change in the future, in response to changes in its drivers.

The following paragraphs discuss how each Change Agent affects pronghorn in the U.S. portion of the ecoregion, as articulated in the stressor model for this CE. A “sub-model” diagram for each Change Agent presents a simplified version of the master stressor model diagram, showing only the direct and indirect effects of the Change Agent of interest. The text accompanying each sub-model diagram summarizes information from Appendix 1 concerning the impacts of each Change Agent, including selected citations. For each Change Agent, the paragraphs summarize present understanding of: (a) which critical environmental elements would likely be affected—directly or indirectly—by each Change Agent and (b) which critical ecological processes would likely be affected—directly or indirectly—by the cascading effects of these changes in critical environmental elements.

12.3.1 Climate Change

The final report for the Chihuahuan Desert REA will include a discussion of current forecasts of the ways in which climate change will affect the ecoregion and its ecological resources. The present chapter presents only the conceptual model of the causal relationships and outcomes that potentially will be affected for pronghorn. Figure 12-2 presents the pronghorn stressor model for the U.S. portion of the ecoregion, simplified to show only those causal relationships that potentially will be affected by changes in the air temperature and precipitation regimes. Appendix 1 presents the rationale and citations for every causal link shown in the diagram.

Climate models for the southwest United States predict warmer temperatures, including warmer nighttime temperatures, fewer days of frost and increased frequency of extreme weather events. Predictions on precipitation are less certain because much larger, continental scale systems control the monsoon storms that bring moisture to the Chihuahuan Desert in the summer (Archer and Predick

2008). For the southwestern United States, the temperature regime showed a statistically significant increase during 1895-2011, while the precipitation regime showed no statistically significant trends in precipitation (Kunkel et al. 2013). However, the spatial extent of drought in the southwest from 2001 to 2020 (forecasted) was the second largest observed for any decade since 1901 (Gedir et al. 2015).

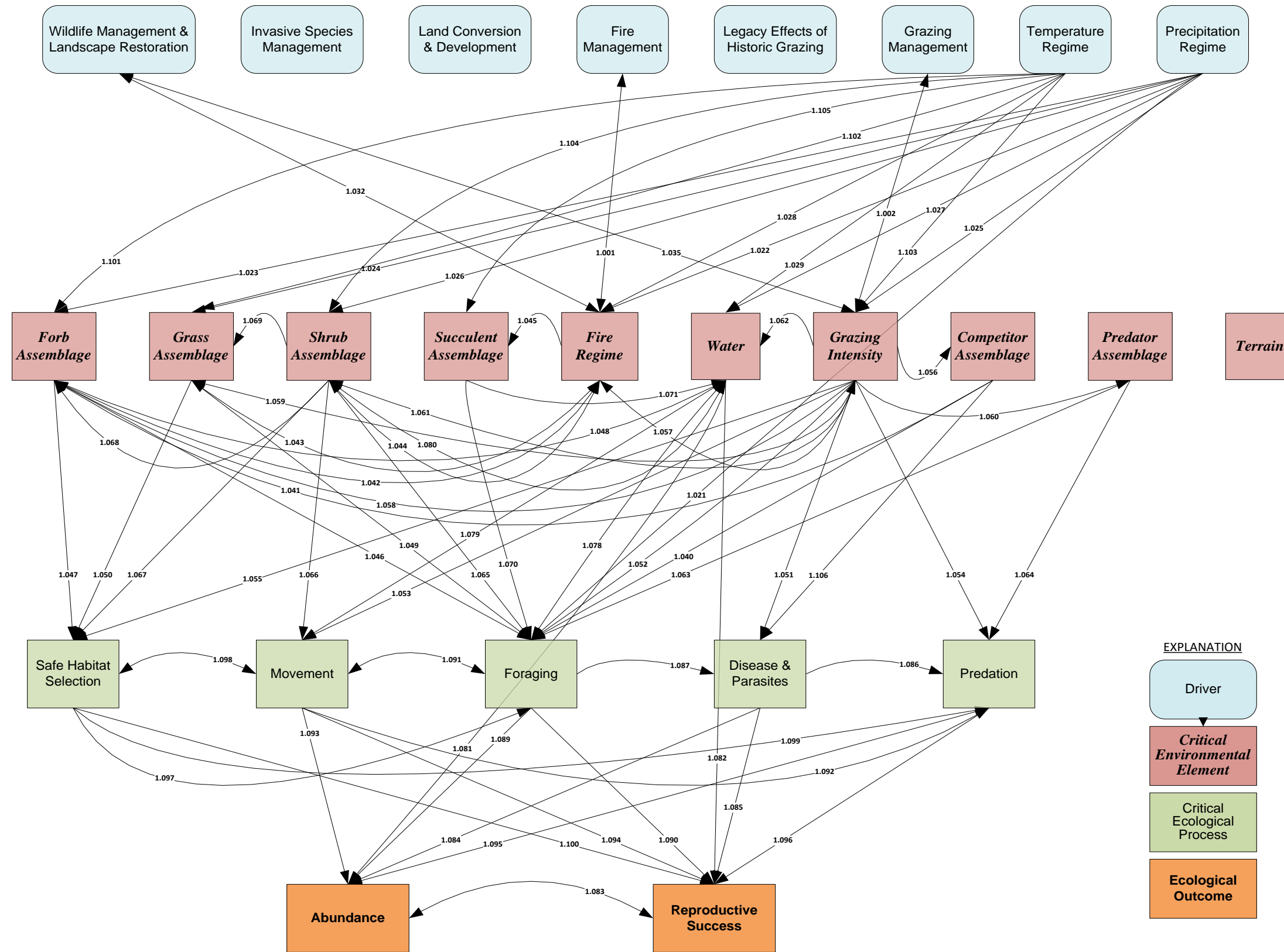
Temperatures are predicted to increase, though models differ in the extent of that increase. Evapotranspiration will therefore increase, putting stress on the ability of plants to grow and reproduce. This will also reduce occurrences of killing frosts (Hatfield et al. 2008; Ryan et al. 2008). The southern portions of the Southwest United States including the Chihuahuan Desert will experience the greatest decreases in average annual precipitation (Kunkel et al. 2013), though the precipitation regime has been predicted to possibly shift to higher precipitation in the winter and lower precipitation in the summer (Neilson 1986). This may favor shrub growth though models also predict an overall reduction in annual precipitation throughout the Southwest (Seager et al. 2007).

The precipitation and temperature regimes regulate plant growth in the Chihuahuan Desert. Most plant species exhibit extensive growth and reproduction early spring and late summer, and this increased plant density and cover is correlated with summer precipitation. Shrubs have adapted to diverse moisture regimes and grow quickly during years of above average precipitation and persist during droughts once established (Kemp 1983, McPherson and Weltzin 2000). For example, creosote bush and mesquite are C₃ species best adapted to wet winters while black grama is a C₄ species best adapted to wet summers. Shrubs can take advantage of winter precipitation, and increases in precipitation in that season could increase the abundance of the shrub assemblage (Neilson 1986). Shrubs recycle nutrients beneath their canopy creating resource islands where germination and growth of shrubs is favored over that of grasses or forbs (Van Auken 2000), which reduces the forb assemblage.

While shrubs are adapted to a diverse moisture regime due to their deeper roots, grasses and forbs respond to shorter term precipitation cycles (Kemp 1983) with both grasses and forbs increasing in years with high precipitation and decreasing in years with low precipitation (Buonopane et al. 2005, Gibbens and Beck 1988). Further, changes in the temperature regime, particularly temperature increases, will increase evapotranspiration and stress on plants and will increase water loss by pronghorn (Cain III et al. 2006, Yoakum 2004c). Water content of plants may also decrease, thereby reducing a source of preformed water for pronghorn at a time when surface water resources will also diminish.

An increase in shrub density driven by climate change could affect pronghorn in several ways. As stated above, pronghorn feed primarily on forbs but will feed on shrubs during dry seasons when forbs are not available (Kemp 1983, Richardson 2006, Yoakum 2004d). The timing of pronghorn reproduction is such that forage species richness is highest during lactation depending on precipitation (Tluczek 2012). Winter and/or spring deaths from malnutrition are the greatest cause of population fluctuation in the Trans-Pecos (Sullins 2002). Increases in shrub density and height could also impede movement and safe habitat selection and possibly make pronghorn more prone to predation.

Figure 12-2. Pronghorn stressor model: Potential impacts of climate change.



EXPLANATION

- Driver (Light Blue box)
- Critical Environmental Element (Red box)
- Critical Ecological Process (Green box)
- Ecological Outcome (Orange box)

Changes in the precipitation and temperature regimes will also affect the fire regime. The amount and spatial arrangement of fuels formed in the shrub, grass and forb assemblages will vary with the amount of growth and time of recovery following disturbance, such as grazing or fire. Increases in temperature and reductions in precipitation will increase that recovery time of grasses and forbs following fire, thereby reducing fine fuels. With lower fuel loads, fire frequency, intensity and spatial area will also decrease. As discussed below (see Uncharacteristic Wildfire, this chapter), changes in the fire regime in turn could have additional effects on pronghorn.

Both the temperature regime and precipitation regime affect the need for and the availability of water for pronghorn (Richardson 2006). Free water comes from surface water sources while preformed water is available in the forb, grass and shrub assemblages and represents a significant source of water for pronghorn (Bristow et al. 2006). Pronghorn may not drink free water, even if available, if the content of moisture content of forbs is at least 75% (Yoakum 2004d). Females may need more water than males. Supplemental water may be needed during lactation from May thru July during dry years (Nelle 2006, Tluczek 2012). Fawn: doe ratios are only slightly related to free water availability, which is most limited in June. Radio collared pronghorn ranged further from water in summer than fall, winter or spring. Yearling pronghorn remained closer to water than adults (Clemente et al. 1995). Plants in the succulent assemblage are also a source of water, particularly in the dry season.

The longer term precipitation regime appears to affect overall pronghorn abundance while short-term precipitation affects pronghorn reproductive success. Pronghorn population abundance is closely related to long-term moisture conditions as measured by the Palmer Hydrologic Drought Index, which is a measure of surface and groundwater availability (Simpson et al. 2007). There is also a relatively strong correlation between the July Palmer Drought Severity Index (PDSI), a measure of overall drought conditions, and the number of observed does the following year. The percentage of annual forbs found in pronghorn pellets declined during a drought that occurred between 2002 and 2004. The number of fawns/100 does was correlated with the number of does observed the following year. There was a strong relationship between the July PDSI and the number of does observed. Mid-summer droughts reduced doe numbers, while increased precipitation increased numbers. These impacts of summer drought on doe numbers might exceed the importance of winter precipitation on fawn survival especially as reductions in the survival of does will have a large impact on fawn reproduction. The variation in annual mortality of does may be more important than fawn recruitment in determining overall pronghorn population abundance (Brown et al. 2006). Rainfall from August-October is directly related to overwinter survival in the Trans-Pecos. Fawn survival, and therefore pronghorn reproductive success, is more closely related to immediate moisture conditions as measured by precipitation than long-term moisture supply as indicated by drought indices. Recruitment of pronghorns through late summer has been positively correlated with precipitation during the previous winter (McKinney et al. 2008). Low precipitation contributed to low weights of fawns in Trans-Pecos (Simpson et al. 2007, Weaver et al. 2012).

A model (Gedir et al. 2015) predicting changes in long-term population of 18 pronghorn populations in the southwest (Utah, Arizona, New Mexico and Texas) based on potential changes in precipitation and temperature indicated that precipitation during the summer period of lactation affected half of the

populations while the other were affected by precipitation during early or late gestation. While temperature was predicted to increase for all populations, and precipitation variable, precipitation was more important in affecting these populations. Sixteen of these populations had declined since the early 1990's. The model compared an increasing amount of CO₂ with a stabilized CO₂ level declining to 1990 levels by 2100. Five of the six populations within the Chihuahuan Desert would be extirpated with the higher CO₂ levels, and stabilize with lower levels. The populations that remained stable in the study were near agricultural lands where pronghorn can use those additional food resources (Gedir et al. 2015).

12.3.2 Uncharacteristic Wildfire

The fire regime of the U.S. portion of the Chihuahuan desert has changed as a result of the interaction of several drivers, as discussed in several previous chapters. Chapters 2 and 3 discuss the history, causes, and consequences of uncharacteristic wildfire across the ecoregion in general, and Chapters 5-7 discuss the causes and consequences of altered fire regimes specifically across the terrestrial ecological systems of the region, in which pronghorn live. Figure 12-3 presents the stressor model for pronghorn in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by uncharacteristic wildfire. See Appendix 1 for a full presentation of the stressor model.

In general, changes in the fire regime—in the spatial extent, frequency, and severity of wildfires—affect pronghorn by shaping the relative abundance of shrubs and forbs versus grasses, and shrub stand height. However, fire extent, frequency, and severity depend on many factors including the plant (fuel) types present, the amount and arrangement of fuels, fuel moisture, weather patterns, and topography. In addition, differences in the season in which fires occur and the frequency of fire can result in very different fire effects as the response of plants to fire depends on their phenology, as discussed in detail in Chapters 2, 3, and 5-7. As a result, the effects of any change in fire patterns on pronghorn in the ecoregion depend on highly localized conditions and histories of fire and vegetation change.

Frequent summer thunderstorms in the Chihuahuan Desert create a natural source of lightning ignitions. Combined with the fine fuels of the grasslands, these ignitions likely caused fires that had significant effects on vegetation composition and structure, creating and maintaining a grass dominated system. Because of the dominance of grasses and shrubs, there are few ways to determine historic fire frequencies. Based on historic observations and where tree ring data can be uncovered, fire frequency has been estimated to range from 5-20 years preceding European settlement (McPherson 1995, Parker 2002, Paysen et al. 2000, Rice et al. 2008). However, there is some discord in the literature relative to desert fire regimes. Long-term studies of the effects of prescribed fire have shown negative impacts of fire on black grama, spike dropseed (*Sporobolus contractus*), purple three-awn and sand muhly (*Muhlenbergia arenicola*). The responses of shrub species, including creosotebush, have been highly variable. Recovery of these species occurred over several years and was very dependent on precipitation (Parmenter 2008).

The use of fire by humans in the borderlands of the Southwest has varied as population densities, settlement patterns and cultural practices have shifted. Native Americans may have altered vegetation at the local level by shifting the frequency and seasonality of fire, but there is no evidence that Native

Americans altered vegetation at a broad spatial scale (Parker 2002). Overgrazing in the late 1800s and early 1900s greatly reduced the grasses that maintained fine fuel loads and supported grassland fires (Jones 2000). Fire suppression is practiced for the protection of structures and to prevent the loss of grasses for grazing of cattle.

Intensive grazing can compact the soil, limit water infiltration, and reduce litter and overall vegetation cover. Grazing-related reductions in the grass assemblage resulted in reductions in fine fuel loads limiting both the chance for ignition and for fire to spread (Jones 2000). Increasing dominance by shrubs over the past 150 years is partially due to changes in climate but the major driver has been the reduction in fine fuels resulting from grazing and the resulting reduction in grassland fires that would have reduce abundance of the shrub assemblage (Van Auken 2000).

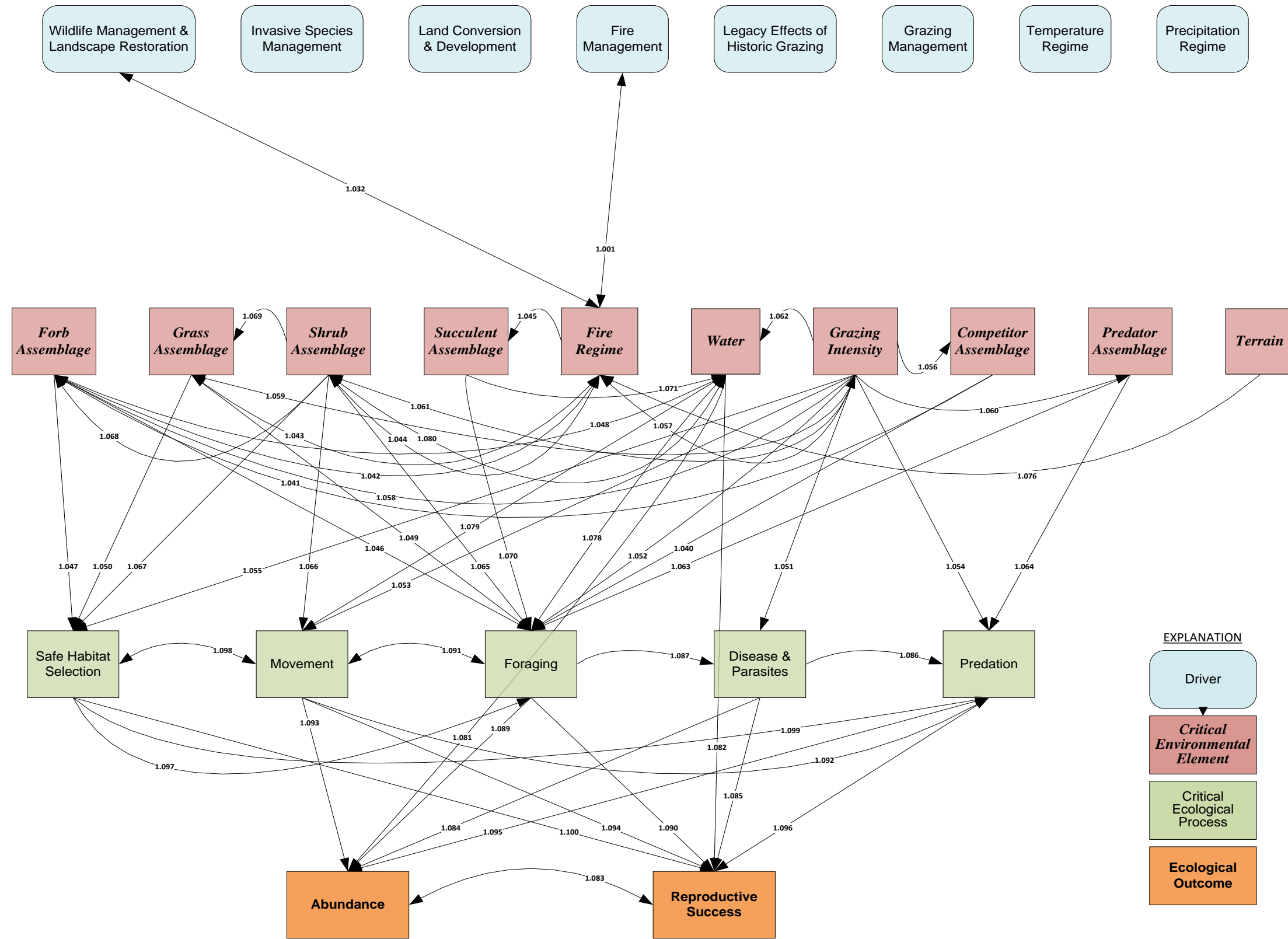
As shown in Figure 12-3, precipitation is a key driver in how plants respond following fire. Droughts following fire can significantly slow recovery of the grass and forb assemblages (Ladwig 2014). Most plant growth occurs during and following the summer monsoon and natural fires occur in early summer at the beginning of the monsoon. The temperature regime, as expressed by seasonal changes in temperature, is also a significant driver of the fire regime because the effects of cool season (fall, winter, early spring) fires will differ from warm season (late spring, summer) fires. In general, fire is less detrimental to dormant plants and many plants are dormant in the cool season compared to plants actively growing in the warm season. While in one study, there were no significant differences in fire effects recorded between spring, summer or fall burns, forb abundance increased most following summer burns (Ladwig 2014). Cool season fires before rains also may encourage annual and perennial forbs (Texas Parks and Wildlife Department 2010). In general, native grasses will be reduced in abundance by summer fires, and recovery is dependent on post-fire precipitation (Parmenter 2008).

While the current abundance of the shrub assemblage is attributed to the lack of fire following historic grazing practices, the degree of mortality and the ability of shrubs to recover from fire are highly variable. Fire can be used to control creosote bush which is intolerant of fire. However, mesquite is more difficult to control, and may require mechanical, herbicide or repeated burns for significant reduction. The ability to apply repeated burns depends on the timing of and amount of accumulation of fine fuels (Lyon et al. 2000). The height and cover of the shrub assemblage will be reduced more by summer than cool season fires (Rice et al. 2008). Therefore, effective control of the shrub assemblage to favor the forb and grass assemblages will require summer fires. If conditions are dry and windy, as may be typical during that period, prescribed burns may be difficult to manage, so fire may be used during the cool season and, hence less effective in shrub control (Brown and Ockenfels 2007). Managers will need to balance the use of fire to reduce the shrub assemblage with potential effects on the grass and forb assemblages incorporating the potential limits on post fire recovery posed by the precipitation regime (Parmenter 2008).

The increase in abundance of non-native grasses represents another biological change in these systems. These species have created systems that are different from those that existed prior to Euro-American settlement and different from those studied in the 1950s through the mid-1970s, when most of the research on fire ecology was conducted (McPherson 2006). These grasses produce more biomass than

native grasses, especially during dry years. The biomass is also highly lignified and decomposes more slowly than native grasses, so the fuel is available for a longer period than where native grasses dominate. This increases the potential for fire spread. Since these nonnative grasses respond favorably to fire, fire facilitates their increased abundance and the concomitant accumulation of fuel (McPherson 2006, Paysen et al. 2000, Rice et al. 2008).

Figure 12-3. Pronghorn stress or model: Potential impacts of uncharacteristic wildfire.



12.3.3 Invasive Species

Chapters 2-3 discuss the ecological consequences of non-native species across the U.S. portion of the ecoregion in general and Chapters 5-7 discuss the consequences of invasive species introductions specifically across the terrestrial ecological systems of the region in which pronghorn live. Figure 12-4 presents the stressor model for pronghorn in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by non-native species and their management. Appendix 1 presents the rationale and citations for each causal link shown in Figure 12-4.

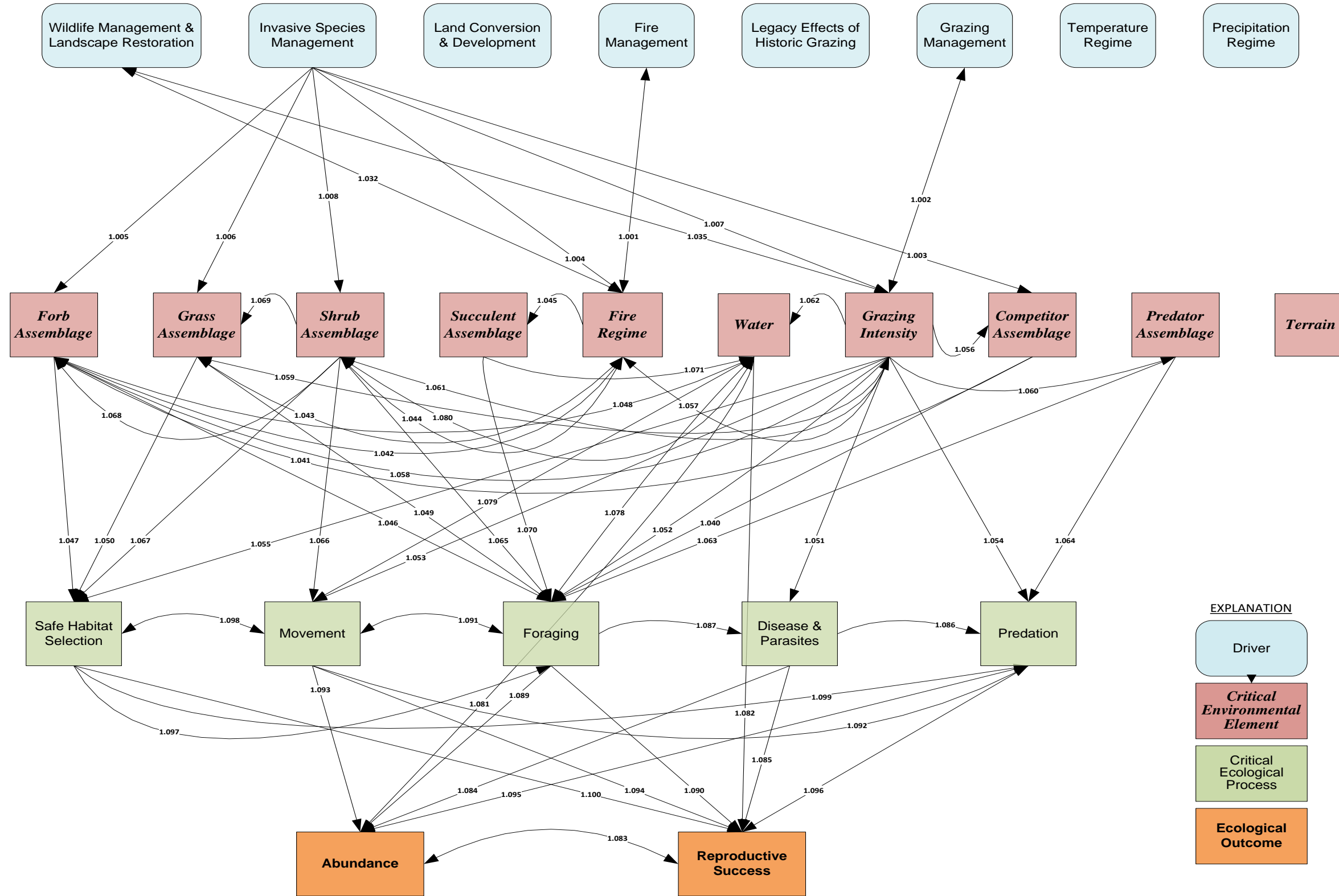
There have been few studies on the effects of invasive plants on pronghorn or other large native herbivores (Steidl et al. 2013). However, the reduced abundance and diversity of forbs and key shrub species likely cause pronghorn to avoid areas where these non-native grasses have become dominant (Heffelfinger et al. 2006). Buffelgrass (*Pennisetum ciliare*), Lehmann's lovegrass (*Eragrostis lehmanniana*), and red brome (*Bromus rubens*) are three of the most significant invaders in southwest deserts, originally introduced to reduce soil erosion and provide forage for cattle (Cox et al. 1984, Heffelfinger et al. 2006, see Chapters 1-2 and 4-6). They have been shown to reduce the biomass, richness and diversity of native plants (McPherson 2006, Steidl et al. 2013). However, the geographic extent of these and other invasive plants in the Chihuahuan Desert is not well known (Rogstad et al. 2009). Other species of concern include giant reed (*Arundo donax*), Bermuda grass (*Cynodon dactylon*), Johnson grass (*Sorghum halepense*) and salt cedar (*Tamarix* spp.) (Rogstad et al. 2009).

Non-native plants also affect pronghorn indirectly, by altering wildfire dynamics (see above, this chapter, and Chapters 1, 2, and 4-6). Buffelgrass, Lehmann's lovegrass and red brome are highly flammable and create more fine fuels than do grasses native to the Chihuahuan Desert. Because they increase fine fuel loads, they can significantly alter the fire regime by increasing the potential for wildfire. They also recover relatively quickly after fire, compared to native grasses, thereby altering fire frequency as fine fuels accumulate more quickly than occurs with native grasses (McPherson 2006, Steidl et al. 2013). Lovegrass is an example of a fire tolerant species, because this species creates deep roots and fire stimulates seed production. By contrast, members of the native grass and forb assemblages may take many years post-fire to recover and that recovery is slowed if precipitation is below normal (Ladwig 2014, Parmenter 2008).

Management of these invasive plants is a complex set of activities that will likely require the use of multiple techniques including mechanical removal, fire management, herbicide treatment and possibly biological controls. Reseeding with native species will also be needed but success is dependent on precipitation. In addition, a system of early detection and response is important to protect intact areas and avoid focusing all resources on invaded areas (Sheley et al. 2011).

Non-native ungulates including cattle, domestic sheep, aoudad or Barbary sheep (*Ammotragus lervia*), Oryx or gemsbok (*Oryx gazella*) and feral hogs (*Sus scrofa*) potentially compete with pronghorn and damage shrubland and grassland habitat (Adkins and Harveston 2007, Schwertner 2002, Texas Parks and Wildlife Department 2012). Competition from cattle and sheep is discussed in Livestock Grazing below.

Figure 12-4. Pronghorn stressor model: Potential impacts of invasive species and associated management.



Oryx feed on some of the same plants as pronghorn. A study found that Oryx feed primarily on grasses, although they also fed on some succulent shrub species that pronghorn used (Smith et al. 1998). Overall diet overlap between Oryx and pronghorn was calculated at 37%; 20% for grasses, 4% for forbs and 13% for shrubs (Yoakum 2004b). In another study, dietary overlap was 37% during the cool-dry season, 17% in the warm dry season and 31% during a drought (Avery 2012). Other studies showed Oryx feeding more extensively on shrubs and forbs, though still using substantial amounts of grass (Marquez and Boecklen 2010, Reid and Patrick 1983). Oryx carry blue-tongue, bovine respiratory syncytial virus and parainfluenza-3 virus (Bender et al. 2003), with bluetongue a potential significant cause of mortality (O’Gara 2004).

12.3.4 Development

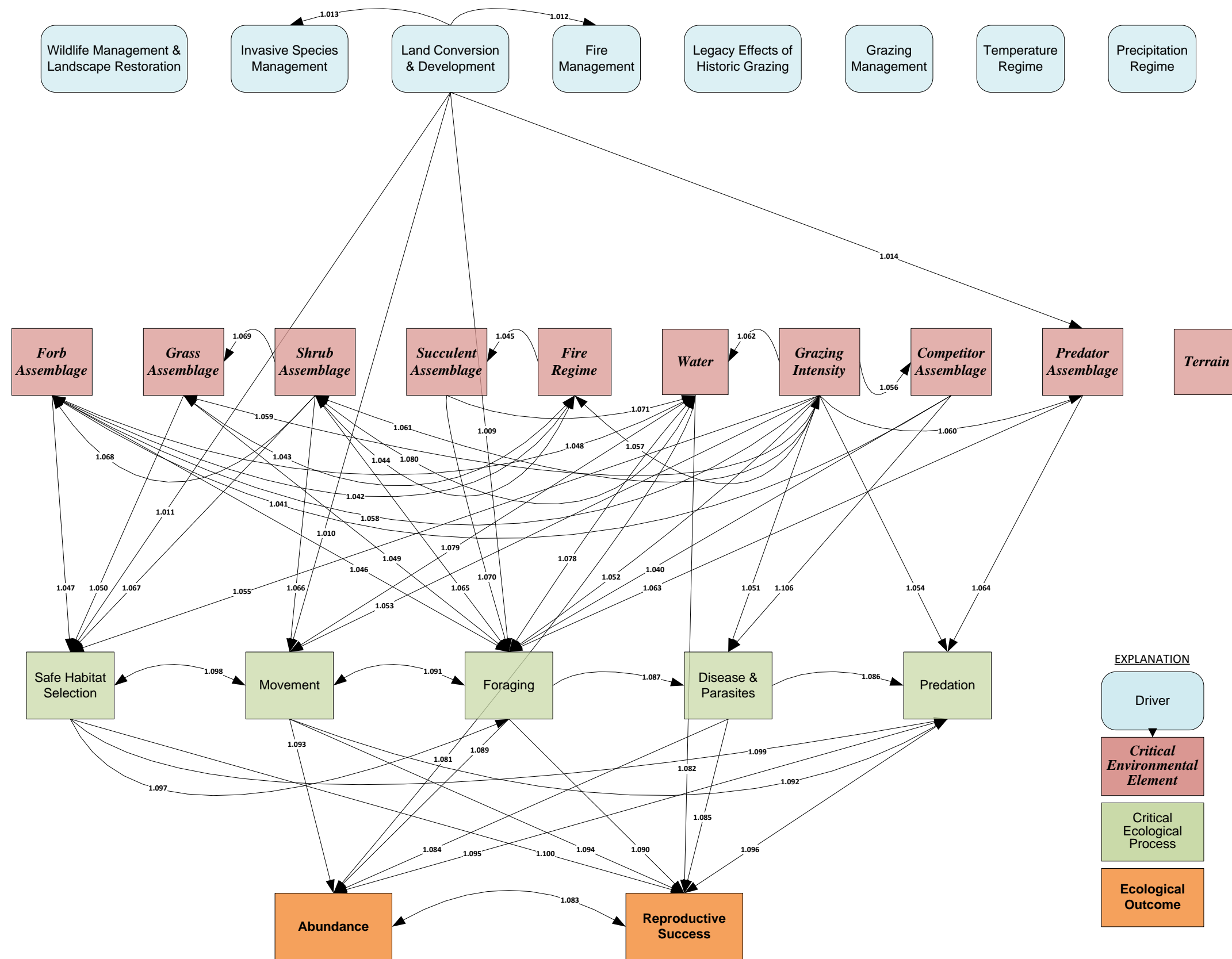
Chapters 2 and 3 discuss the history and consequences of land development across the ecoregion in general, and Chapters 5-7 discuss these consequences specifically across the terrestrial ecological systems of the region, in which pronghorn live. Figure 12-5 presents the stressor model for pronghorn in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by land development. See Appendix 1 for a full presentation of the stressor model.

In general, land development affects pronghorn by directly eliminating habitat for feeding, hiding, and moving among habitat patches, and by confining local populations within limited landscape fragments. Land development can also affect pronghorn indirectly, through impacts to wildfire patterns and the spread of non-native species.

Land conversion and development include residential and commercial development, roads, utility rights-of-way, energy extraction, mining and agricultural uses. Agriculture, urbanization, energy extraction, border security and military activities and off-road vehicle use can create barriers to both short and long-term movement of pronghorn. Increased road building and traffic in the borderlands region can cause habitat loss and fragmentation, reduce safe habitat selection, increase roadkill mortality, and increase illegal hunting. Loss of habitat for feeding and hiding, barriers to movement and reduced access to water and foraging opportunities reduce pronghorn abundance (New Mexico Department of Game and Fish 2006, Yoakum 2004d). In other parts of their range, pronghorn travel from long distances from winter to summer territories, usually established using corridors. If such corridors are important in the Chihuahuan Desert, then increased development could create barriers to seasonal movement (Richardson 2006). Where movement of pronghorn is restricted by roads, fences and other barriers, the predator assemblage may significantly affect small pronghorn populations (Yoakum et al. 2014).

Phase II of this REA will assess the extent and pace of development in the U.S. portion of the ecoregion. Ruhlman et al. (2012) report very little change in the amount of land in development in the U.S. portion of the Chihuahuan Desert from 1973 to 2000 yet Pool et al. (2014) report an approximate six percent annual loss of grassland to agriculture in a portion of the Chihuahuan Desert in Mexico. The U.S.-Mexico borderlands are a center of increased road building and traffic, causing additional habitat loss and fragmentation. This development is also increasing illegal poaching. Much of the habitat degradation is tied to energy extraction within the area that occurs on public as well as private lands (New Mexico Department of Game and Fish 2006). Further information on the extent of land conversion is critically needed because all types of habitat loss and land use conversion take place over long time periods so the effects are difficult to quantify (Bergman et al. 2015).

Figure 12-5. Pronghorn stressor model: Potential impacts of development.



Land development potentially also can affect pronghorn through its effects on fire management. Land development potentially can limit the use of prescribed fire near human developments for safety and health reasons including smoke impacts. Increasing development could also lead to the need for other types of fuel reduction treatments that could alter habitat. These changes in the fire regime would then influence the forb, grass, shrub and succulent assemblages. Increased land development also can contribute to the expansion of off-road vehicular use, which in turn can foster the spread and establishment of invasive species (Brooks and Pyke 2001).

12.3.5 Excessive Domestic Grazing

Chapters 2-3 discuss the ecological consequences of excessive domestic grazing across the U.S. portion of the ecoregion in general and Chapters 5-7 discuss the history and consequences of livestock grazing specifically across the terrestrial ecological systems of the region in which pronghorn live. Figure 12-6 presents the stressor model for pronghorn in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by non-native species and their management. Appendix 1 presents the rationale and citations for each causal link shown in Figure 12-6.

In general, excessive domestic grazing affects pronghorn by altering vegetation, both directly through the grazing itself and indirectly through secondary effects on wildfire and the spread of non-native plants. Livestock can also compete with pronghorn for space, food, and water, and transmit pathogens that also affect pronghorn.

The legacy effects of historic grazing in the 1800s and early 1900s resulted in significant reductions in the abundance of the grass assemblage, soil loss and erosion, and stream incision, which in turn lowered water tables and reduced the availability of natural watering sites (Bahre 1995; 1991, Kerley and Whitford 2000, Parker 2002, Van Auken 2000). Historic grazing practices also affected the fire regime by reducing fine fuels from the grass assemblage and reducing fire frequency, intensity and area affected, thereby allowing for greater abundance in the shrub assemblage, as discussed above, this chapter, and in Chapters 2, 3, and 5-7. This altered vegetation assemblage is also enhanced by cattle eating seeds of some shrubs, particularly mesquite, they pass through their gut and thereby spread across a much wider area (Bahre 1995; 1991, McPherson 2006). These interactions have created a landscape dramatically different from what existed prior to European settlement.

Grazing intensity is generally measured based on forage use, qualitative categories (light, moderate, severe) and quantitative measures of vegetation abundance (Holechek and Galt 2000). Grazing can alter plant community composition, reduce abundance of forage plants, and alter vegetation structure. Domestic cattle and sheep as well as introduced ungulates represent the competitor assemblage. As stated above, dietary overlap between cattle and pronghorn is relatively limited, but if grazing intensity is high and/or precipitation low, competition between cattle and pronghorn could increase (Yoakum 2004b).

Cattle and pronghorn can share the same areas. However, pronghorn females have been documented moving away from cattle during fawning and choosing sites with taller vegetation. In this way, livestock can reduce options for safe habitat selection and may increase pronghorn vulnerability to predation (Yoakum 2004c, Yoakum et al. 2014). The dietary overlap between cattle and pronghorn is relatively limited, but if grazing intensity is high and/or precipitation low, competition between cattle and pronghorn could increase (Yoakum 2004b).

Domestic sheep prefer forbs and so have diets similar to pronghorn potentially reducing forage for them. A review of studies indicates substantial dietary overlap, with abundance of the forb assemblage reduced by sheep grazing (Yoakum 2004b). Sheep also can carry disease and parasites that afflict pronghorn (Yoakum et al. 2014).

Other nonnative ungulates (aoudad, feral hogs) do feed on forbs and could limit foraging opportunities for pronghorn (Adkins and Harveson 2007, Schwertner 2002).

A literature review of grazing impacts on arid western systems indicates soil porosity, litter cover and vegetation cover are negatively affected. The reduction in forb assemblage, along with grass and shrub assemblages, reduces forage quality and the nutritional value of the plant community (Jones 2000). However, grazing impacts can be highly variable. In one study, ungrazed areas had lower cover of the forb assemblage, but light to moderate grazing resulted in higher cover of the forb assemblage (Holechek 1991). On the other hand, removing cattle from a northern Arizona grassland for five years did not significantly improve either cover of the forb assemblage nor cover that provided concealment from predators (Loeser et al. 2005). A study of an area where cattle had been excluded for 16 years found an increase the cover of annual forbs, but that increase was statistically significant for only a few species, possibly because annual forbs are highly dependent on precipitation (Kelt and Valone 1995).

Grazing may directly and indirectly create barriers to movement. Pronghorns prefer flat or shallowly sloped terrain, but will use sloped terrain to avoid overgrazed, flatter areas (Buechner 1950). Therefore, grazing may force pronghorn into areas where escape from members of the predator assemblage is more difficult. Fencing designed to contain cattle can limit movement, their ability to escape predators, and safe habitat selection. Pronghorn are unable to jump fences. They can generally get under cattle fences but not sheep fences (Buechner 1950). The bottom wires need to be at least 16" aboveground (Brown and Ockenfels 2007, Nelson et al. 1999, O'Shaughnessy et al. 2014).

The sustainability of grazing depends on precipitation. Grasses and forbs need time to recover from grazing. Range condition with low intensity grazing increased in years with sufficient precipitation and decreased during drought. Annual precipitation of approximately 26-35 cm is needed to allow for plant recovery and sustainable grazing (Molinar et al. 2011, Navarro et al. 2002). Light stocking is recommended following years of below average precipitation (Holechek et al. 2003; 1994, Khumalo et al. 2007). Short term, high intensity grazing is sometimes practiced and may compact the soil and decrease water infiltration.

12.3.6 Landscape Restoration

Chapters 3 and 5-10 discusses the types of landscape restoration projects taking place in the U.S. portion of the ecoregion, some of which currently or in the future could affect pronghorn; and pronghorn themselves are managed as a game species in the U.S. portion of the ecoregion. Figure 12-7 presents the stressor model for pronghorn in the U.S. portion of the ecoregion, simplified to show only those causal relationships that address landscape restoration and pronghorn management. Appendix 1 presents the rationale and citations for each causal link shown in Figure 12-7.

Management of pronghorn involves four activities: (1) regulating hunting to provide for recreational hunting while sustaining population numbers, (2) controlling predators to reduce predation, (3) improving habitat to enhance safe habitat selection for pronghorn, and (4) translocating pronghorn to areas that historically supported populations.

Pronghorn hunting in the U.S. portion of the ecoregion is regulated by the Texas Parks and Wildlife Department, New Mexico Department of Game and Fish, and Arizona Game and Fish Department. Each of these agencies has hunting regulations that specify the number of pronghorns that can be killed by each hunter. Hunting is therefore a source of adult pronghorn mortality, although the control of hunting permits limits hunting as a significant pressure on the pronghorn population. Most hunters seek trophy animals and only kill bucks. Some ranchers want to limit the number of pronghorns on their land while increasing income by charging hunting fees (Yoakum et al. 2014).

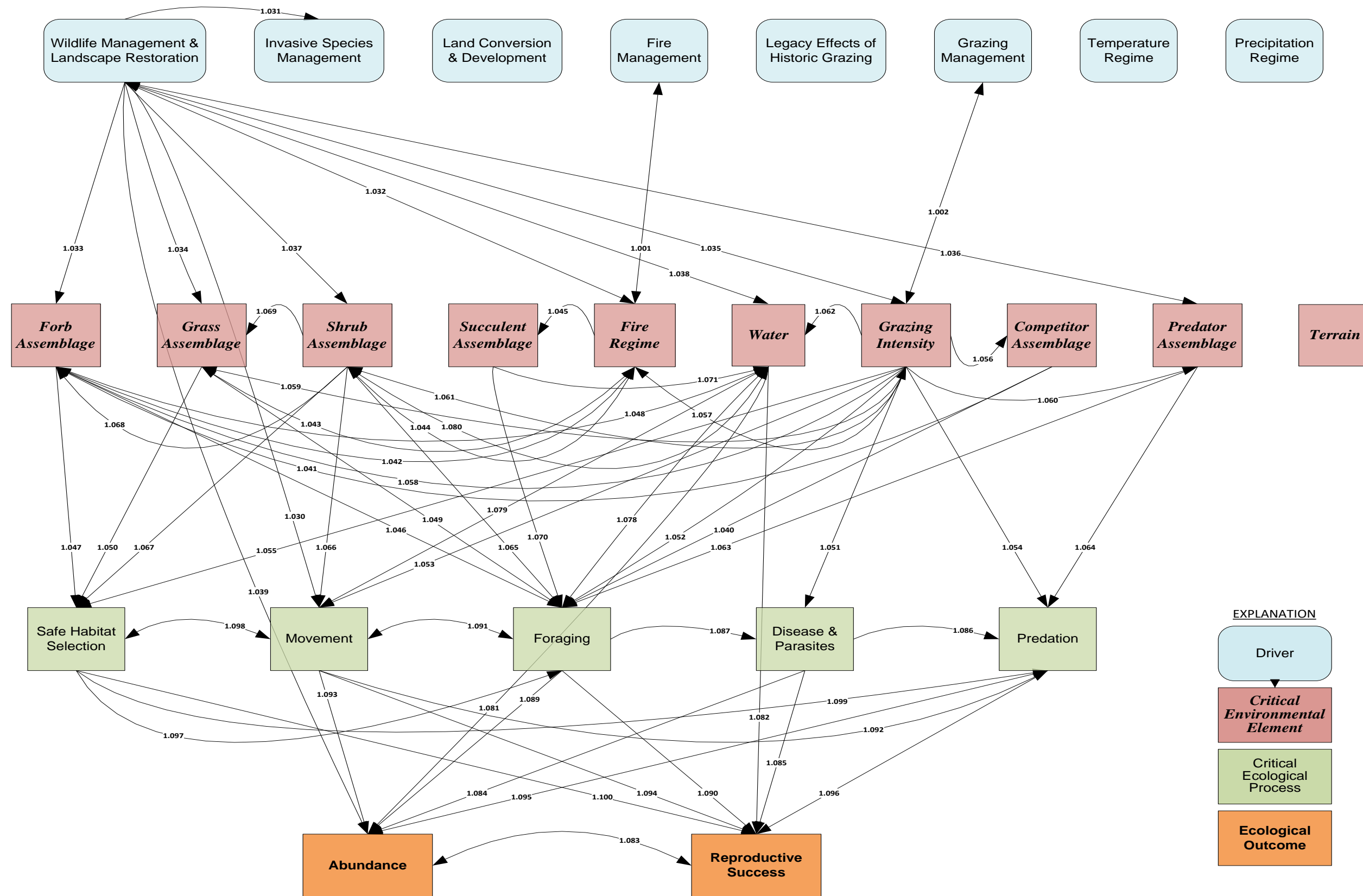
The state agencies determine harvest rates based on estimated buck: doe ratios. Agency wildlife biologists use surveys including spotlight surveys after dark or daylight helicopter surveys to determine the status of the population and set permit numbers for upcoming hunting seasons. Biologists also conduct buck: doe and fawn: doe surveys in the late summer for population modelling (Yoakum et al. 2014).

The state agencies may translocate pronghorn to repopulate areas where populations have decline and habitat is suitable. Care is taken when capturing and transporting animals to avoid causing them harm. Translocated animals may become disoriented in their new surroundings and may be more susceptible to predation. Translocated animals may also encounter unfamiliar barriers, particularly roads and fences (Harveson et al. 2012a).

Control of the predator assemblage may relieve some predation pressure on pronghorn. However, pronghorn mortality and survival are more closely related to forage quality, which is primarily driven by the precipitation regime. Control of predators may also have unintended consequences. For example, the litter size of coyote increases following local population control, and juveniles become sexually mature earlier, likely as a result of reduced competition. Coyote population control may be more successful if completed after the establishment of dominance and territorial patterns for the coming breeding season and prior to whelping to prevent establishment of new breeding pairs (Connolly 1978). Predator control benefits pronghorn more if carried out just before fawning, but the practice requires repeated control. Instances of coyote control have resulted in increased fawn survival. However, such control has rarely resulted in increased herd sizes, although it may be justified for translocated pronghorn (Yoakum et al. 2014).

Habitat restoration to benefit pronghorn may involve (1) providing adequate food, which is dependent on maintaining a diversity of forbs and small shrubs; and/or (2) manipulating vegetation density to balance the need for cover for fawns to avoid predation with the need for adults to see predators and escape predation. These two purposes may be achieved by appropriate application of grazing intensity or prescribed fire to increase the abundance of the forb assemblage and to maintain the shrub assemblage in a condition that allows movement and safe habitat selection. Providing cover for fawns is crucial, and requires balancing horizontal and vertical cover with sufficient visibility to prevent predators from stalking fawns through dense vegetation (Sumner 2006).

Figure 12-7. Pronghorn stressor model: Potential impacts of landscape restoration.



Prescribed fire can be applied to improve habitat for fawns in this manner. However, the fires should be relatively small to provide a patchy mosaic with limited shrub cover (Howard 1995). In addition, successful fire treatments require adequate precipitation to allow for recovery of native species (Morton and Melgoza 1991, Richardson 2006, Yoakum 2004e).

Mechanical treatments, the use of herbicides and fire can be used in various combinations to reduce shrub cover and height and increase the forb assemblage. Seeding may also be a necessary follow up to these techniques provided precipitation following treatment is adequate. Development of water sources can also aid pronghorn provided they do not have high levels of dissolved solids (Bristow et al. 2006), which may occur following use by livestock (Yoakum et al. 2014). Another key landscape restoration approach benefiting pronghorn involves eliminating or redesigning fences to allow for movement. Fences can present significant barriers as pronghorn generally will not jump fences, but prefer to go under them. The bottom of the fence should be 16 inches or higher, and should be smooth and not barbed. Fences become significant barriers when alongside roads (Arizona Game and Fish Department 2013, Yoakum et al. 2014). A larger and more difficult task is to reduce fragmentation from highways and urban development or from differences in land management between private and public lands that may isolate small populations and cause direct mortality from vehicular accidents (Arizona Game and Fish Department 2013).

At Duff Springs in Brewster County, Texas, 962 hectares were restored to benefit pronghorn. The first step was to apply herbicides to reduce mesquite using helicopters. Native grasses were then seeded. Two years afterwards, bare ground had been significantly reduced and herbaceous cover had replaced mesquite. Surface water also increased as a result of mesquite reduction, leading to a perennial water source (Warnock 2006).

Wildlife management and landscape restoration for pronghorn also must incorporate invasive species management as part of an overall effort to restore and maintain foraging resources and safe habitat selection opportunities, including taking into account the possible transmission of diseases and parasites from sheep to pronghorn (see above, Invasive Species, this chapter).

12.4 Pronghorn Key Ecological Attributes

All ecological outcomes and critical ecological processes in the pronghorn stressor model constitute the key ecological attributes for the CE. The list below identifies 7 key ecological attributes for pronghorn based on these criteria. *Fully characterizing the present condition of the CE will require data on indicators for its key ecological attributes.* Indicators are determined during Phase II of the REA process. The definitions for the key ecological attributes are the same as the definitions for these model components presented above.

- **Ecological Outcomes**
 - Abundance
 - Reproductive Success
- **Critical Ecological Processes**
 - Disease and Parasites
 - Foraging
 - Movement
 - Predation
 - Safe Habitat Selection

13 Mule Deer Conceptual Model

This chapter presents the conceptual ecological model for mule deer, *Odocoileus hemionus* (Rafinesque 1817). Mule deer range from southeastern Alaska and Canada south through the western conterminous U.S. and into Mexico. Their distribution is discontinuous in Mexico and the western conterminous U.S., with major gaps in the Mojave and Sonoran Deserts, southeastern California, southern Nevada, southwestern Arizona, and northwestern Sonora in Mexico (Innes 2013). Throughout their range, mule deer occupy a variety of habitats including agricultural lands, forests, grasslands, savannas and shrublands. In much of their range, mule deer migrate from high elevations in the summer to lower elevations in winter (Innes 2013). Some taxonomists refer to mule deer in the southwestern deserts, including the Chihuahuan desert, as *Odocoileus hemionus eremicus* (Heffelfinger 2006, Innes 2013). The presentation of the mule deer conceptual model follows the structure described in Chapter 4, with sections on sources of information, a species overview, the stressor model, and key ecological attributes. As noted in Chapter 4, the conceptual models for species and species assemblages do not include a separate control model.

13.1 Sources of Information

The mule deer overview and stressor model integrate information from several sources, including summaries of the ecology of the species both in general across its entire range and specifically within the Chihuahuan desert (e.g., Short et al. 1965, Short 1977, Smith and Lecount 1979, Hibler 1981, Urness 1981, Wallmo 1981, Krausman et al. 1985, Ordway and Krausman 1986, Krausman and Leopold 1988, Scarbrough and Krausman 1988, Krausman et al. 1989, Fox and Krausman 1994, Lawrence 1995, Krausman et al. 1997, Sanchez-Rojas and Gallina 2000a; 2000b, Lingle 2002, Avey et al. 2003, Morris 2003, Lawrence et al. 2004, Marshal et al. 2004, Yoakum 2004, Marshal et al. 2005, Cain III et al. 2006, Heffelfinger et al. 2006, Heffelfinger 2006, Marshal et al. 2006, Brewer and Harveson 2007, Bender et al. 2007; 2011, Esparza-Carlos et al. 2011, Tollefson et al. 2011, Innes 2013, Bergman et al. 2015, NatureServe 2015).

13.2 Mule Deer Overview

The Chihuahuan Desert is a biologically rich ecoregion and a significant part of the known historic range of mule deer. The Chihuahuan Desert has a diversity of natural communities, several of which are threatened by increasing development and fragmentation, changes in climate and a shift from grasslands to more shrub dominated communities. The ability of mule deer to continue to exist within the Chihuahuan Desert will depend on landscape scale actions to assure that sufficient habitat exists within the changing range of variation of drivers, critical environmental elements and critical ecological processes resulting from current and future threats.

13.2.1 Distribution

Mule deer are important economically and socially, particularly as a game species for hunters who live in or visit the area. Mule deer are also an important prey species for several predators as discussed below.

Mule deer seem to have minimal effects on native vegetation. In one study in the Chihuahuan Desert, mule deer and other native herbivores did not affect vegetation composition but did reduce vegetation height in comparisons between enclosed (deer excluded) versus control plots (Krausman and Leopold 1988). Mule deer have suffered declines throughout many parts of their range, and their population has fluctuated dramatically in the southwest. Mule deer occur at lower densities and with a more discontinuous distribution in the U.S. portion of the ecoregion than in other types of habitats throughout their range (Sanchez-Rojas and Gallina 2000a, Wallmo 1981).

13.2.2 Habitat

Mule deer require adequate and available foraging opportunities, access to water, including water from plants, good visibility and terrain allowing for movement for foraging, safe habitat selection and to avoid predation (Esparza-Carlos et al. 2011, Heffelfinger et al. 2006, Morris 2003). In the southwest, mule deer occur in desert shrublands, semi-desert shrubland-grasslands, chaparral, mountain shrublands and woodlands and forests at higher elevations. Additionally, washes are important for water, food, escape, resting and as corridors for travel (Innes 2013).

The Chihuahuan Desert is composed of several ecological systems consisting of shrublands, mixed shrub and grasslands and grasslands. The dominant plant species throughout the Chihuahuan Desert is creosote bush (*Larrea tridentata*). Depending on latitude and other factors, other co-dominant shrubs include tarbush (*Flourensia cernua*), honey mesquite (*Prosopis glandulosa*), four-winged saltbush (*Atriplex canescens*), several Agave and Yucca species, including the endemic lechuguilla (*Agave lechuguilla*), candelilla (*Euphorbia antisyphilitica*), ocotillo (*Fouquieria splendens*), and sotol (*Dasylirion* spp.). Cacti include pricklypear (*Opuntia* spp.), cholla (*Cylindropuntia* spp.), barrel cactus (*Ferocactus* spp.) and Arizona rainbow cactus (*Echinocereus polyacanthus*). Grasslands include these shrubs and a mixture of grasses including side-oats grama (*Bouteloua curtipendula*), black grama (*Bouteloua eriopoda*), tobosa grass (*Hilaria mutica*), and purple three-awn (*Aristida purpurea*) (NatureServe 2015).

Mule deer in the Chihuahuan Desert primarily browse on shrubs and forbs consuming very little grass (Heffelfinger 2006, Marshal et al. 2004). The high plant species diversity in the Chihuahuan Desert provides for many foraging choices. The growth and diversity of plant species is a function of their response to the precipitation and temperature regimes, and the nutrient quality of forage (crude protein content, dry matter digestibility and water content) are positively associated with rainfall (Marshal et al. 2005, Sanchez-Rojas and Gallina 2000a). The species of available forage change with the seasons. In the winter, wavyleaf oak (*Quercus undulata*), Mohr Shrub oak (*Q. mohriana*), and Emory oak (*Q. emoryi*), littleleaf sumac (*Rhus microphylla*), skunkbush (*R. trilobata*), Apache plum (*Fallugia paradoxa*), and mountain mahogany (*Cercocarpus* spp.) are important shrubs. Oaks and their acorns are important food sources along with other shrubs throughout the year. Forbs are most abundant in mule deer diets in the spring and decline through summer and fall (Marshal et al. 2004). While shrubs are always an important food source, they become more important in the fall and winter as forbs decrease in importance. Succulents are important food sources, though cacti are low in protein and phosphorus and consumed in low abundance in most studies, so these are likely more important as sources of preformed water (Heffelfinger 2006, Krausman et al. 1997, Short 1977).

Forbs important as mule deer food include spurge (*Euphorbia* spp.), Dalea (*Dalea* spp.), globemallow (*Sphaeralcea* spp.), tansymustard (*Descurainia* spp.) and milkwort (*Polygala* spp.). With green up following the summer rains, skeletonleaf goldeneye (*Viguiera cordifolia*), spurge, Dalea, buckwheat (*Eriogonum* spp.), dayflower (*Commelina* spp.), dogweed (*Thymophylla pentachaeta*) and needleleaf bluets (*Hedyotis acerosa*) become important. The abundance and diversity of forbs vary with the timing and amount of rainfall (Heffelfinger 2006). Many plant species exhibit extensive growth and reproduction in early spring and late summer. Increased plant density and cover have been correlated with precipitation. In winter and early spring, C₃ annuals utilize moisture from winter frontal storms. In summer, C₄ annuals respond to convective storms. Shrubs have adapted to diverse moisture regimes (Kemp 1983). As the summer forbs disappear in the dry fall, deer resume feeding on shrub species and the remaining forbs.

The ability of deer to find sufficient nutritional food and water, particularly during reproduction, strongly shapes their abundance and reproductive success. Deer select the most nutritious plants available, and the abundance and types (browse or shrub vs. forbs) varies with the season (Urness 1981). Browse provides most of the nutrition for deer throughout the year, but deer utilize forbs extensively when they become available in the spring and summer (Heffelfinger 2006, Krausman et al. 1997, Marshal et al. 2004) The volume of the rumen-reticulum in mule deer is relatively small and their metabolism high compared to other ruminants so that their ability to digest materials high in cellulose, like grasses, is limited (Heffelfinger et al. 2006, Short et al. 1965).

Mule deer depend on free standing water although the forb assemblage and succulent assemblage also provide important sources of preformed water (Krausman et al. 1997, Marshal et al. 2004). Accessibility to free water, as measured by travel distance, is a key factor in deer abundance in the Chihuahuan Desert. In a comparison of two sites, deer density was greater where the average distance to water was between 2.99 and 3.68 km (0.70 to 4.21 deer/km²) compared to a site where the average distance was from 5.44 to 5.69 km (0.77 to 2.73 deer/km²) (Sanchez-Rojas and Gallina 2000a; 2000b). Mule deer use different habitats across seasons (temperature regime) and use habitat that is a greater distance to water in winter and spring than during summer or fall. Deer also use washes where surface water may be available, the diversity and biomass of the shrub, forb and grass assemblages is greater due to the greater soil moisture, and shelter is also available (Heffelfinger et al. 2006, Lawrence 1995, Marshal et al. 2006, Scarbrough and Krausman 1988).

Deer movement patterns vary depending on the precipitation regime, temperature regime and terrain. In dry years, when food is less abundant, deer expand their habitat and may use areas where visibility and their ability to detect predators are reduced, which can increase predation risk (Esparza-Carlos et al. 2011). Terrain also affects safe habitat selection and seasonal movement. Male and female deer may also use different terrain at different times of the year. In Arizona, female deer used shallower slopes in the spring where forage was more abundant and steeper slopes in summer (Marshal et al. 2006). In the Trans-Pecos region, male deer tended to use low elevations and south and west facing slopes while female deer were less specific in terrain choices (Lawrence 1995), although an Arizona study found that they tended to use more mountainous terrain (Ordway and Krausman 1986). Males were observed

using higher elevations during rut from December to February while females remained at high elevations (Scarborough and Krausman 1988). Uneven terrain allows deer to escape predators due to their ability to bound great distances making predator pursuit difficult. Female deer may use higher elevations to protect their young. Terrain also helps deer avoid predation by allowing them to detect predators at a distance (Sanchez-Rojas and Gallina 2000a). Deer use riparian and xeroriparian areas for foraging due to the high diversity of forbs and important shrubs as well as the presence of free water (Heffelfinger et al. 2006). Deer may also use these areas for cover and as travel corridors (Krausman et al. 1985).

The density of the shrub assemblage also affects safe habitat selection. In the Trans-Pecos, where mule deer and white-tailed deer are sympatric, mule deer used areas where woody cover ranged from 32.2 to 44.2% (95% CI) and abandoned areas of woody cover greater than 75% (Avey et al. 2003). The extent that shrub height is a factor was not reported. Shrubs impede movement as well as visibility.

Terrain and vegetation density also affect reproductive success. Fawns have limited mobility (movement) early in life and bed sites are important for them to be able to avoid predators. Smaller fawns are more subject to predation (Tollefson et al. 2011) primarily from coyotes (*Canis latrans*), so that fawn survival depends on how fast they can grow to allow them to escape. Bed sites are generally on steep slopes, which are used less often by coyotes. Females were found to avoid creosote flats where shrub density is higher and visibility reduced (Fox and Krausman 1994).

Mule deer have similar food preferences with other native ungulates such as pronghorn (*Antilocapra americana*) but there appears to be little interaction or competition for resources between deer and pronghorn (Yoakum 2004). Bighorn sheep (*Ovis canadensis*) have a similar diet, consuming primarily forbs and shrubs and little grass (Brewer and Harveson 2007). Dietary overlap was minimal in one study, and habitat is generally mountainous and steep terrain (Krausman et al. 1989).

Most studies indicate that access to high quality food is the major factor in regulating deer populations. The lack of body fat as a result of inadequate foraging resources was shown to be the primary cause of adult female mortality, and resulting low fawn survival, in two studies in New Mexico (Bender et al. 2011; 2007). There has been suggestion, that water is also a critical factor, but experimental evidence is lacking. (Cain III et al. 2006) In addition, predation, the predator assemblage and disease and parasites do have impacts on mule deer populations. Deer make tradeoffs in the form of safe habitat selection between foraging resources and the risk of predation (Esparza-Carlos et al. 2011, Heffelfinger 2006, Lawrence et al. 2004, Smith and Lecount 1979). However, the number of predators is highly variable across the landscape and over time. Mountain lions (*Puma concolor*) are the primary predator of adult deer (Heffelfinger 2006). Coyotes also prey on adults (Lingle 2002) but are more of a threat to fawns. Fawns are also preyed upon by golden eagles (*Aquila chrysaetos*) and bobcats (*Lynx rufus*) (Heffelfinger 2006).

Mule deer are subject to numerous diseases and parasites and deer weakened by disease and parasites are more subject to predation (Hibler 1981). In a Colorado study, mountain lion and other predators selected for animals weakened by chronic wasting disease (Bergman et al. 2015). Internal and external

parasites are common but rarely cause widespread mortality (Heffelfinger 2006). Deer that are malnourished are more likely to become infected and/or to suffer debilitation or mortality. In particular, poorly nourished fawns may be susceptible to disease and parasites. As discussed above, the precipitation regime has a major effect on the forb assemblage, grass assemblage, and shrub assemblage and by affecting the nutrition of deer through foraging indirectly drives disease and parasites.

13.3 Mule Deer Stressor Model

Table 13-1 presents and defines the drivers, critical environmental elements, critical ecological processes, and ecological outcomes represented in the mule deer stressor model. The stressor model follows the methodology for species described in Chapter 4.

Table 13-1. Definitions of stressor model components for mule deer.

Model Component	Definition
Drivers	
Fire Management	Refers to any human activities to control the size, timing, intensity, seasonality or frequency of fires. Fire management applies to the control and prevention of wildfire as well as the use of prescribed fire to achieve management goals and objectives.
Grazing Management	Refers to human activities to manage and control where and how domestic cattle are grazed within the Chihuahuan Desert. This includes the number of cattle grazed per hectare or other measures of grazing intensity as well as any grazing rotation strategies.
Invasive Species Management	Refers to any human activities to reduce the abundance of plant and animal invasive species in the Chihuahuan Desert, with a primary focus on invasive grasses. This management may include mechanical removal, herbicide application, and native grass plantings. Invasive Species Management may overlap with Fire Management, Grazing Management, and Wildlife Management and Landscape Restoration when any of those activities also impact invasive species.
Land Conversion and Development	Refers to any human activity that converts native habitats (shrubland or grassland) into developed areas. Developed areas include urban, industrial, suburban development, areas of intense recreational activity, military activities, borderland security activities as well as rangelands used for grazing or agricultural croplands. This driver also refers to threats posed by human development including barriers to movement, collisions with motor vehicles, attraction of predators and other related threats.
Legacy Effects of Historic Grazing	Refers to the effects of unsustainable domestic cattle grazing in the 19th and early 20th centuries that significantly altered vegetation community composition and structure, especially increasing conversion of grasslands to shrublands.
Precipitation Regime	Refers to the pattern, timing, intensity, and average annual rate of precipitation in the Chihuahuan Desert within and across seasons.
Temperature Regime	Refers to the patterns of air temperature in the Chihuahuan Desert within and across all seasons.
Wildlife Management and Habitat Restoration	Refers to human activities to manage for wildlife and to restore habitat including vegetation management to alter vegetation structure and composition, providing water sources, hunting, and predator control. This driver overlaps with fire management, grazing management and invasive species management.
Critical Environmental Elements	
Competitor Assemblage	Refers to the composition and abundance of both wild and introduced animals that compete with mule deer by foraging on plants on which mule deer also forage.

Model Component	Definition
Fire Regime	Refers to the frequency, intensity, severity and seasonality of both wildfire and prescribed fire.
Forb Assemblage	Refers to the composition and abundance of herbaceous plants in the forb guild.
Grass Assemblage	Refers to composition and abundance of grass species.
Grazing Intensity	Refers to the number of cattle grazed per acre and the frequency of grazing rotations. In the literature, intensity is generally characterized as light, moderate, or heavy.
Predator Assemblage	Refers to the composition and abundance of predators that prey on mule deer.
Shrub Assemblage	Refers to the composition and abundance of woody shrub species.
Succulent Assemblage	Refers to the composition and abundance of succulent plant species, which are an important food and water source for mule deer.
Terrain	Refers to landscape features including rock formations, cliffs, slope, aspect and variations in elevation.
Water	Refers to sources of preformed, metabolic and free water.
Critical Ecological Processes	
Disease and Parasites	Refers to any condition affecting the health of individual mule deer, including internal and external parasites, bacterial, fungal and viral infections, and environmentally-based toxins.
Foraging	Refers to the ability of mule deer to actively find and consume food resources within their habitats
Movement	Refers to movement by mule deer between patches of habitat to avoid predators, for foraging, and for reproduction including using different terrain in different seasons. Movement may be affected by areas developed for agriculture or urban uses, as these areas provide unsuitable habitat, potential threats from vehicles, altered predator densities and barriers to movement.
Predation	Refers to mortality that mule deer suffer due to predators such as mountain lions and coyotes.
Safe Habitat Selection	Refers to the preferential use of habitat to avoid predators, for thermoregulation, for foraging and to access water.
Ecological Outcomes	
Abundance	Refers to population size, ratios of males to females and females to fawns, and spatial distribution of mule deer.
Reproductive Success	Refers to the ability of mule deer to successfully give birth and raise young to self-sufficiency.

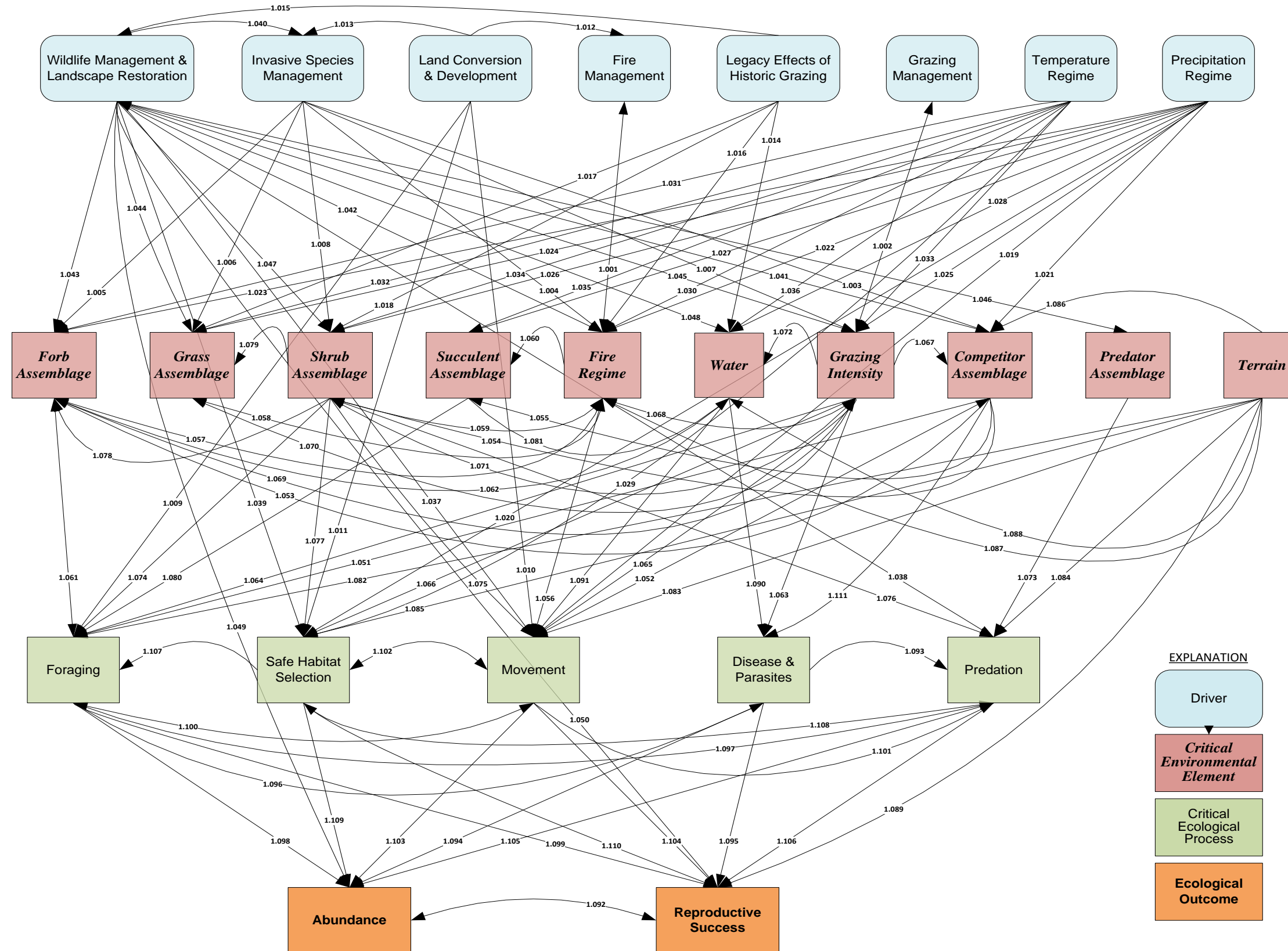
Figure 13-1 shows the full mule deer stressor model, displaying the model components listed in Table 13-1 along with their causal relationships. Appendix 1 describes and presents the rationale for including every causal relationship in the stressor model, with citations for each causal relationship, providing a comprehensive presentation of the stressor model. Appendix 1 also provides a larger-format version of Figure 13-1.

Figure 13-1 indicates the presence or absence of causal relationships between the system model components but does not indicate the potential magnitude or other characteristics of these relationships, as explained in Chapter 4. Figure 13-1 identifies the causal relationships that have affected how the system likely has changed in the past, in response to changes in its drivers. The diagram provides a means for articulating how the condition of the system will likely change in response to changes in its drivers. As discussed and illustrated below, the stressor model makes it clear: (a) which critical environmental elements would likely be affected by a change in one or more particular drivers,

including change agents; (b) which critical ecological processes would likely be affected by the cascading effects of these changes in critical environmental elements; and (c) which system characteristics (ecological outcomes) would likely be affected by the cascading effects of these changes in critical environmental elements and ecological processes. By doing so, in turn, the stressor model also highlights those components of the model—drivers, critical environmental elements, critical ecological processes, and ecological outcomes—that demand indicator data.

As defined earlier, a causal relationship exists when a change in one component of the system results in a change in some other component. Change in the first component is said to “cause a change in the second component. Each chain of causation, from driver to outcome, describes how the condition of the system likely has changed in the past, or likely would change in the future, in response to changes in its drivers.

Figure 13-1. Mule deer stressor model.



The following paragraphs discuss how each Change Agent affects mule deer in the U.S. portion of the ecoregion, as articulated in the stressor model for this CE. A “sub-model” diagram for each Change Agent presents a simplified version of the master stressor model diagram, showing only the direct and indirect effects of the Change Agent of interest. The text accompanying each sub-model diagram summarizes information from Appendix 1 concerning the impacts of each Change Agent, including selected citations. For each Change Agent, the paragraphs summarize present understanding of: (a) which critical environmental elements would likely be affected—directly or indirectly—by each Change Agent and (b) which critical ecological processes would likely be affected—directly or indirectly—by the cascading effects of these changes in critical environmental elements.

13.3.1 Climate Change

The final report for the Chihuahuan Desert REA will include a discussion of current forecasts of the ways in which climate change will affect the ecoregion and its ecological resources. The present chapter presents only the conceptual model of the causal relationships and outcomes that potentially will be affected for mule deer. Figure 13-2 presents the mule deer stressor model for the U.S. portion of the ecoregion, simplified to show only those causal relationships that potentially will be affected by changes in the air temperature and precipitation regimes. Appendix 1 presents the rationale and citations for every causal link shown in the diagram.

Climate models predict warmer temperatures, including warmer nighttime temperatures, fewer frost days and increased frequency of extreme weather events in the southwestern United States. Predictions of precipitation are less certain because much larger, continental scale systems control the monsoons that bring moisture to the Chihuahuan Desert in summer (Archer and Predick 2008). For the southwestern United States, the temperature regime shows a statistically significant increase in the period 1895-2011 while the precipitation regime shows no statistically significant trends (Kunkel et al. 2013). However, the spatial extent of drought in the southwest from 2001 to 2020 (forecasted) was the second largest observed for any decade since 1901 (Gedir et al. 2015).

Temperatures are predicted to increase, though models differ in the extent of that increase. Evapotranspiration will therefore increase, putting stress on the ability of plants to grow and reproduce. Temperature increases will also reduce occurrences of killing frosts (Hatfield et al. 2008, Ryan et al. 2008). The southern portions of the Southwest United States including the Chihuahuan Desert will experience the greatest decreases in average annual precipitation (Kunkel et al. 2013). Climate models predict a reduction in total annual precipitation (Seager et al. 2007) although the precipitation regime has been also predicted to shift to higher precipitation in the winter and lower precipitation in the summer (Neilson 1986), which may favor shrub growth.

Forbs are an important component of mule deer diet, and the abundance of the forb assemblage, particularly the abundance of annual forbs and small shrubs, is strongly correlated with winter precipitation (Smith and Lecount 1979). Nutritional content is highest during rapid forage growth associated with rainfall (Marshal et al. 2005). Since rainfall is not predictable, desert mule deer consume a variety of browse and forb species and may move great distances for food. The higher the diversity of plants, the more food possibilities are available. Localized summer storms create patches of higher

forage quality and deer respond by moving to these patches (Marshall et al. 2005). There is evidence that deer may be able to detect distant rainfall and move to areas where rain has fallen (Rautenstrauch and Krausman 1989) so that movement is key to foraging and to safe habitat selection.

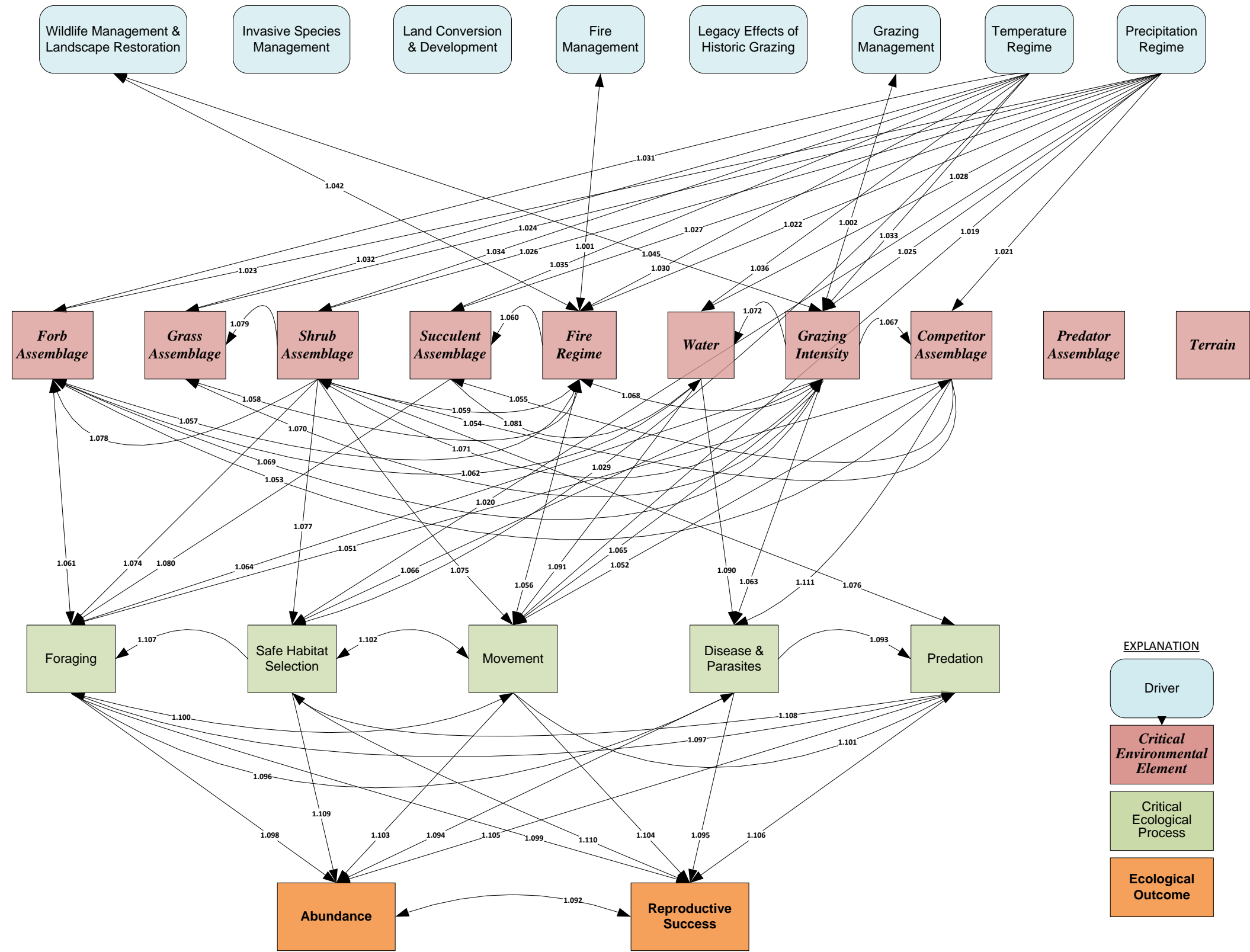
Nutrition of both males and females is critical to the birth and survival of fawns. The precipitation regime influences deer nutrition and reproduction by influencing the shrub and forb assemblages that provide the food for deer, which in turn affects foraging and ultimately the health and reproductive success of deer. The peak period for fawning is highest during the peak period of precipitation. In southwest Texas and southeast New Mexico, the peak period for fawning occurs from June to September (Leopold and Krausman 1991, Urness 1981). Fawn survival is correlated with winter forb abundance (though not with summer forb abundance) and the amount of rainfall from October to April preceding fawning (Smith and LeCount 1979). Female nutrient demands associated with gestation result in low body fat in late spring and early summer, so drought will result in aborted or absorbed fetuses. Lactation demands can be as high as 2.3 times the basal metabolic rate for adult females (Lawrence et al. 2004). As forage quality decreases, milk quality decreases, affecting fawn growth. Smaller fawns nurse more frequently than larger fawns and may be rejected more often by diet stressed does. They may also be more subject to predation as they grow more slowly (Tollefson et al. 2011). Multiple years of above average precipitation may be necessary for the population to increase in size due to the nutritional requirements needed for reproduction (Heffelfinger et al. 2006).

Further, changes in the temperature regime, particularly temperature increases, will increase evapotranspiration and stress on plants and will increase water loss by mule deer (Cain III et al. 2006). Water content of plants may also decrease, thereby reducing a source of preformed water for mule deer at a time when surface water resources will also diminish.

Increases in the shrub assemblage as a result of altered temperature and precipitation regimes will lead to reductions in the forb and grass assemblages, affecting mule deer food resources and cover. Shrubs have adapted to diverse moisture regimes and grow quickly during years of above average precipitation and persist during droughts once established (Kemp 1983, McPherson and Weltzin 2000). Shrubs can take advantage of winter precipitation, which could increase the cover and height of the shrub assemblage (Neilson 1986). Shrubs also recycle nutrients beneath their canopy creating resource islands where germination and growth of shrubs is favored over that of grasses or forbs (Van Auken 2000).

Changes in the precipitation and temperature regimes will also affect the fire regime. The amount and spatial arrangement of fuels formed in the shrub, grass and forb assemblages will vary with the amount of growth and time of recovery following disturbance, such as grazing or fire. Increases in temperature and reductions in precipitation will increase that recovery time of grasses and forbs following fire, thereby reducing fine fuels. With lower fuel loads, fire frequency, intensity and spatial area will also decrease. As discussed below (see Uncharacteristic Wildfire, this chapter), changes in the fire regime in turn could have additional effects on mule deer.

Figure 13-2. Mule deer stressor model: Potential impacts of climate change.



13.3.2 Uncharacteristic Wildfire

The fire regime of the U.S. portion of the Chihuahuan desert has changed as a result of the interaction of several drivers, as discussed in several previous chapters. Chapters 2 and 3 discuss the history, causes, and consequences of uncharacteristic wildfire across the ecoregion in general, and Chapters 5-10 discuss the causes and consequences of altered fire regimes specifically across the terrestrial and riparian ecological systems of the region, in which mule deer live or through which they move. Figure 13-3 presents the stressor model for mule deer in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by uncharacteristic wildfire. See Appendix 1 for a full presentation of the stressor model.

In general, changes in the fire regime – in the spatial extent, frequency, and severity of wildfires – affect mule deer by shaping the relative abundance of shrubs and forbs versus grasses, shrub stand height, and riparian corridor quality. However, fire extent, frequency, and severity depend on many factors including the plant (fuel) types present, the amount and arrangement of fuels, fuel moisture, weather patterns, and topography. In addition, differences in the season in which fires occur and the frequency of fire can result in very different fire effects as the response of plants to fire depends on their phenology, as discussed in detail in Chapters 2, 3, and 5-7. As a result, the effects of any change in fire patterns on mule deer in the ecoregion depend on highly localized conditions and histories of fire and vegetation change.

Frequent summer thunderstorms in the Chihuahuan Desert create a natural source of lightning ignitions. Combined with the fine fuels of the grasslands, these ignitions likely caused fires that had significant effects on vegetation composition and structure, creating and maintaining a grass dominated system. Because of the dominance of grasses and shrubs, there are few ways to determine historic fire frequencies. Based on historic observations and where tree ring data can be uncovered, fire frequency has been estimated to range from 5-20 years preceding European settlement (McPherson 1995, Parker 2002, Paysen et al. 2000, Rice et al. 2008). However, there is some discord in the literature relative to desert fire regimes. Long-term studies of the effects of prescribed fire have shown negative impacts of fire on black grama, spike dropseed (*Sporobolus contractus*), purple three-awn and sand muhly (*Muhlenbergia arenicola*). The responses of shrub species, including creosotebush, have been highly variable. Recovery of these species occurred over several years and was very dependent on precipitation (Parmenter 2008).

The use of fire by humans in the borderlands of the Southwest has varied as population densities, settlement patterns and cultural practices have shifted. Native Americans may have altered vegetation at the local level by shifting the frequency and seasonality of fire, but there is no evidence that Native Americans altered vegetation at a broad spatial scale (Parker 2002). Overgrazing in the late 1800s and early 1900s greatly reduced the grasses that maintained fine fuel loads and supported grassland fires (Jones 2000). Fire suppression is practiced for the protection of structures and to prevent the loss of grasses for grazing of cattle.

Intensive grazing can compact the soil, limit water infiltration, and reduce litter and overall vegetation cover. Grazing-related reductions in the grass assemblage resulted in reductions in fine fuel loads

limiting both the chance for ignition and for fire to spread (Jones 2000). Increasing dominance by shrubs over the past 150 years is partially due to changes in climate but the major driver has been the reduction in fine fuels resulting from grazing and the resulting reduction in grassland fires that would have reduce abundance of the shrub assemblage (Van Auken 2000). Alteration of the fire regime led to changes in plant species composition that were likely detrimental to mule deer (Heffelfinger et al. 2006).

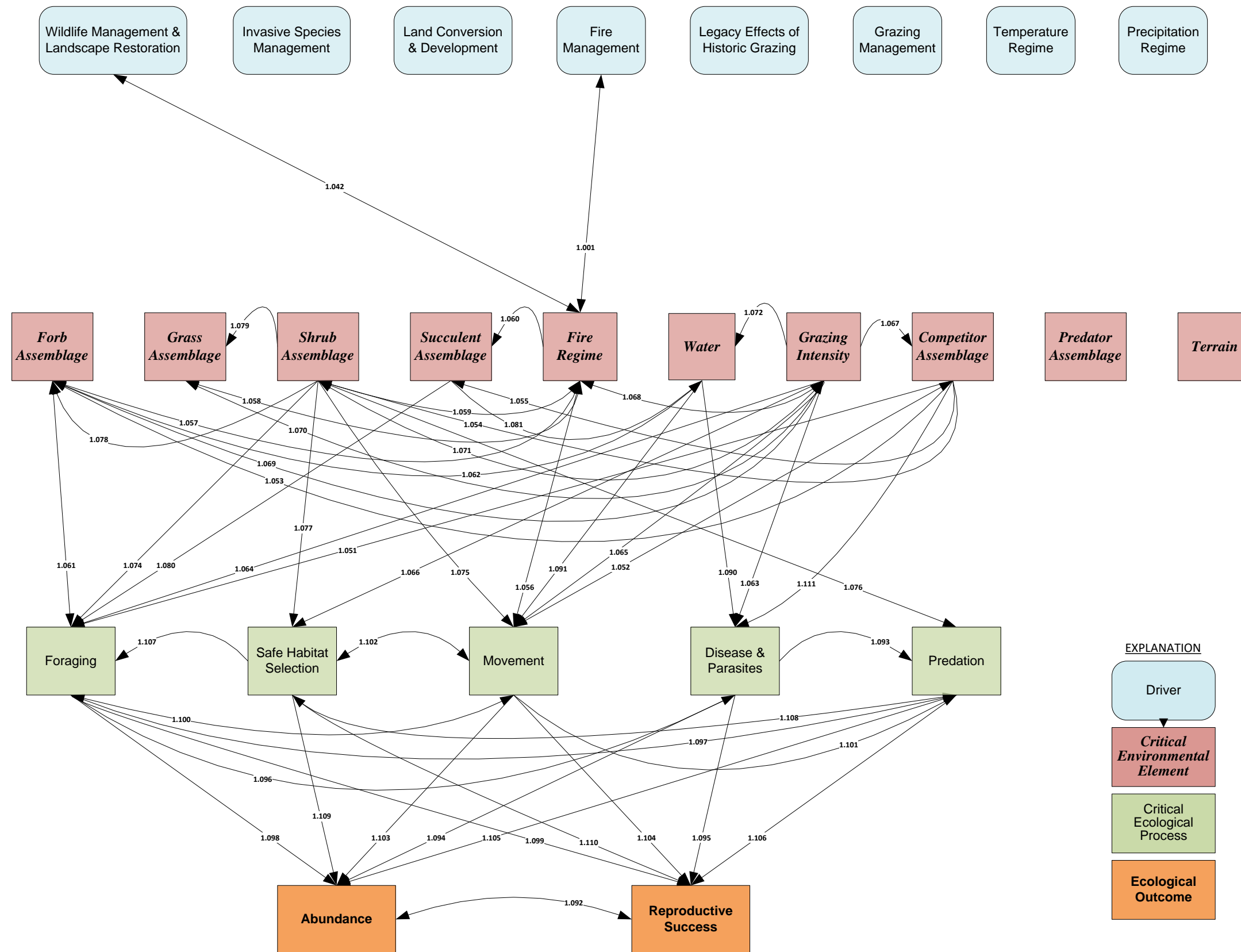
As shown in Figure 13-3, the precipitation regime is a key driver in how plants will respond following fire. Droughts following fire can significantly slow post-fire recovery of the grass and forb assemblages (Ladwig 2014), although one study found that grass abundance decreased and perennial forb abundance increased following fires in near drought conditions (Drewa and Havstad 2001). Most plant growth occurs during and following the summer monsoon and natural fires occur in early summer at the beginning of the monsoon. The temperature regime, as expressed by seasonal changes in temperature, is also a significant driver of the fire regime because the effects of cool season (fall, winter, early spring) fires will differ from warm season (late spring, summer) fires. In general, fire is less detrimental to dormant plants and many plants are dormant in the cool season compared to plants actively growing in the warm season. While in one study, there were no significant differences in fire effects recorded between spring, summer or fall burns, forb abundance increased most following summer burns (Ladwig 2014). Cool season fires before rains also may encourage annual and perennial forbs (Texas Parks and Wildlife Department 2010). In general, native grasses will be reduced in abundance by summer fires, and recovery is dependent on post-fire precipitation (Parmenter 2008). Recovery of the shrub assemblage is also dependent on post-fire precipitation (Rice et al. 2008), although shrubs have deeper roots and, therefore, more access to water (Molinar et al. 2002).

While the current abundance of the shrub assemblage is attributed to the lack of fire following historic grazing practices, the degree of mortality and the ability of shrubs to recover from fire are highly variable. Fire can be used to control creosote bush which is intolerant of fire. However, mesquite is more difficult to control, and may require mechanical, herbicide or repeated burns for significant reduction. The ability to apply repeated burns depends on the timing of and amount of accumulation of fine fuels (Lyon et al. 2000). The height and cover of the shrub assemblage will be reduced more by summer than cool season fires (Rice et al. 2008). Therefore, effective control of the shrub assemblage to favor the forb and grass assemblages will require summer fires. If conditions are dry and windy, as may be typical during that period, prescribed burns may be difficult to manage, so fire may be used during the cool season and, hence less effective in shrub control (Brown and Ockenfels 2007). Managers will need to balance the use of fire to reduce the shrub assemblage with potential effects on the grass and forb assemblages incorporating the potential limits on post fire recovery posed by the precipitation regime (Parmenter 2008).

The increase in abundance of nonnative grasses represents another biological change in these systems. For example the diversity of herbaceous species declines with increased abundance of lovegrass (Steidl et al. 2013). These species have created systems that are different from pre-settlement and different from those studied in the 1950s through the mid-1970s, when most of the research on fire ecology was conducted (McPherson 2006). These grasses produce more biomass than native grasses, especially during dry years. The biomass is also highly lignified and decomposes more slowly than native grasses,

so the fuel is available for a longer period of the year than where native grasses dominate. This increases the potential for fire spread and for higher intensity fires. Since these nonnative grasses respond favorably to fire, fire facilitates their increased abundance and the concomitant accumulation of fine fuels (McPherson 2006, Paysen et al. 2000, Rice et al. 2008).

Figure 13-3. Mule deer stressor model: Potential impacts of uncharacteristic wildfire.



13.3.3 Invasive Species

Chapters 2-3 discuss the ecological consequences of non-native species across the U.S. portion of the ecoregion in general and Chapters 5-10 discuss the consequences of invasive species introductions specifically across the terrestrial and riparian ecological systems of the region in which mule deer live and/or through which they move. Figure 13-4 presents the stressor model for mule deer in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by non-native species and their management. Appendix 1 presents the rationale and citations for each causal link shown in Figure 13-4.

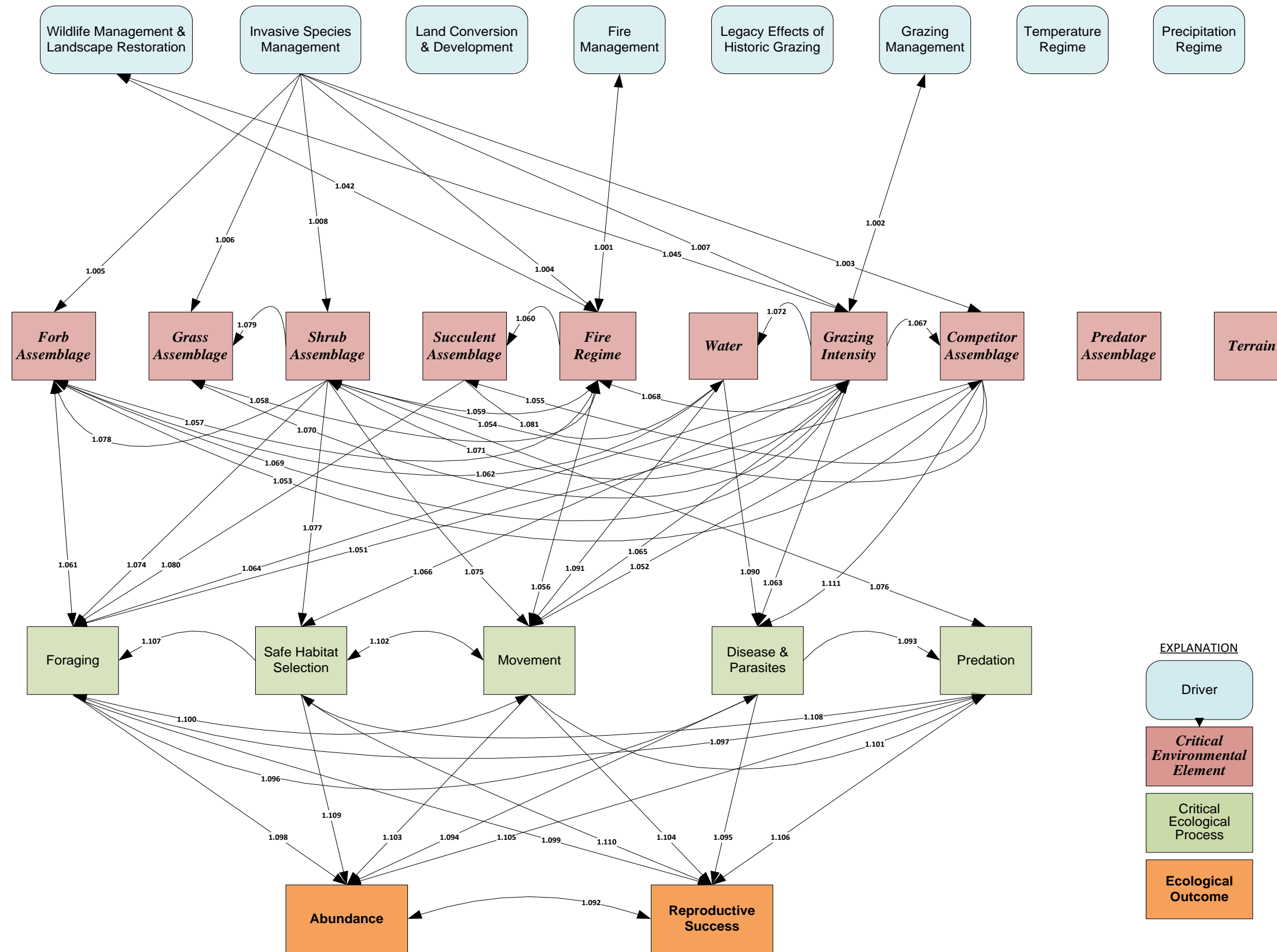
Few studies on the effects of invasive plants on mule deer and other large herbivores have been undertaken (Steidl et al. 2013). Buffelgrass (*Pennisetum ciliare*), Lehmann's Lovegrass (*Eragrostis lehmanniana*), and red brome (*Bromus rubens*) are three of the most significant invaders in southwest deserts. These and other grasses were originally seeded to address the loss of grasses for grazing by cattle and the problems of erosion caused by overgrazing (Cox et al. 1984, Heffelfinger et al. 2006, see Chapters 2-3 and 5-7). They have been shown to reduce the biomass, richness and diversity of native plants (McPherson 2006, Steidl et al. 2013). However, the geographic extent of these and other invasive plants in the Chihuahuan Desert is not well known (Rogstad et al. 2009). Other species of concern include giant reed (*Arundo donax*), Bermuda grass (*Cynodon dactylon*), Johnson grass (*Sorghum halepense*) and salt cedar (*Tamarix* spp.) (Rogstad et al. 2009).

One study of mule deer diet in the Sonoran Desert found that deer rarely consumed buffelgrass, though it was the most important forage for cattle (Alcala-Galvan and Krausman 2012). The reduced abundance and diversity of forbs and key shrub species would likely cause deer to avoid areas where these nonnative grasses have become dominant (Heffelfinger et al. 2006). However, one study indicated that mule deer did use habitat that had been invaded by buffelgrass, but selected sites based on high thermal cover from shrubs and proximity to water sources (Alcala-Galvan and Krausman 2013).

Non-native plants also affect mule deer indirectly, by altering wildfire dynamics (see above, this chapter, and Chapters 2, 3, and 5-7). Buffelgrass, Lehmann's Lovegrass and red brome are highly flammable and create more fine fuels than grasses native to the Chihuahuan Desert. Because they increase fine fuel loads they can significantly alter the fire regime by increasing the potential for wildfire. They also recover more quickly after fire than native grasses (McPherson 2006, Steidl et al. 2013). Some characteristics that help make grasses like lovegrass fire tolerant include deep roots and fire stimulated seed production. By contrast, members of the native grass and forb assemblages may take many years post-fire to recover and that recovery is slowed if precipitation is below normal (Ladwig 2014, Parmenter 2008). Non-native salt cedar may dominate many riparian areas today; however, the literature reviewed for this REA did not identify salt cedar dominance as a factor in mule deer use of riparian areas.

Non-native ungulates including cattle, domestic sheep, aoudad or Barbary sheep (*Ammotragus lervia*), feral hogs (*Sus scrofa*), Oryx or gemsbok (*Oryx gazella gazella*) and Persian (Bezoar) Ibex (*Capra aegagrus*) are part of the competitor assemblage using many of the food resources needed by deer, altering grazing intensity and thereby altering the shrub, grass and forb assemblages (Schwertner 2002, Texas Parks and Wildlife Department 2012). Competition from cattle and sheep is discussed in Livestock Grazing below.

Figure 13-4. Mule deer stressor model: Potential impacts of invasive species and associated management.



Oryx occupy all of White Sand Missile Test Range and have extended their range beyond. As oryx have increased, mule deer have decreased though there are no studies linking cause and effect (Dye 1998). Studies of oryx diets have shown a variation in food preferences ranging from primarily grasses, a relatively small proportion of shrubs and almost no forbs (Smith et al. 1998a), to primarily grasses with smaller proportions of forbs and shrubs (Dye 1998), to an almost even mixture of the three types of forage (Fletcher 2000). Any dietary overlap could affect deer abundance and reproductive success, particularly during drought periods when abundance of the forb and shrub assemblages is reduced. Oryx are also carriers of malignant catarrhal fever (MCF), which is fatal to mule deer though no studies have shown transmission from oryx to mule deer (Bender et al. 2003, Heffelfinger 2006).

The density of feral hogs is relatively low in the Texas portion of the Chihuahuan Desert so that their impacts on mule deer habitat are limited (Adkins and Harveston 2007). Aoudad sheep forage on many of the same plants as mule deer, and were found to have a dietary overlap of 74% in a study in Texas (Mungall 2000). However, aoudad have more flexibility in their diet giving them some competitive advantage. While evidence of impacts of these invasive animals on mule deer abundance are sparse at present, increased competition for limited resources will likely reduce deer abundance (Schwertner 2002).

Management of invasive plants that affect mule deer is a complex set of activities that will likely require the use of multiple techniques including mechanical removal, fire management, herbicide treatment and possibly biological controls. Reseeding with native species will also be needed but success is dependent on precipitation. In addition, a system of early detection and response is important to protect intact areas and avoid focusing all resources on invaded areas (Sheley et al. 2011). At the same time, if applied incorrectly, invasive species management could reduce native species. Deer feed on both shrubs and forbs, using different species in different seasons. Forbs and new shrub growth is particularly important for doe and fawn nutrition. Methods to reduce invasive species, particularly the application of herbicides, could affect native species abundance as well.

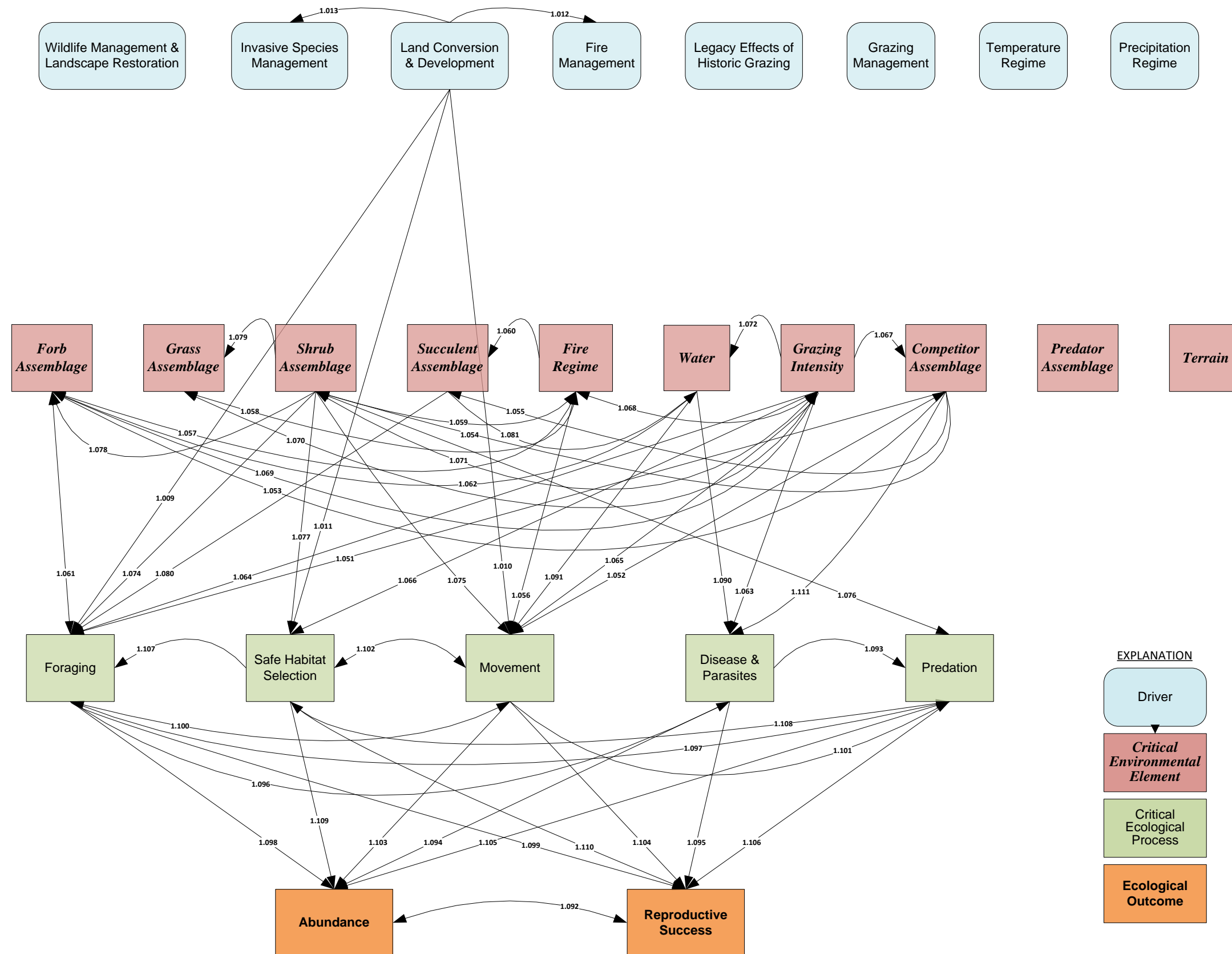
13.3.4 Development

Chapters 2 and 3 discuss the history and consequences of land development across the ecoregion in general, and Chapters 5-7 discuss these consequences specifically across the terrestrial ecological systems of the region, in which mule deer live. Figure 13-5 presents the stressor model for mule deer in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by land development. See Appendix 1 for a full presentation of the stressor model.

In general, land development affects mule deer by directly eliminating habitat for feeding, hiding, and moving among habitat patches, and by confining local populations within limited landscape fragments. Land development can also affect mule deer indirectly, through impacts to wildfire patterns and the spread of non-native species.

Land conversion and development includes the construction of residential and commercial development, roads, and utility rights-of-way, mining, energy extraction and agricultural land uses. All of these uses destroy and fragment habitat. Some development may provide additional water resources for deer (Tull and Krausman 2007). However, if deer are attracted to development, predators also will, creating greater conflicts between humans and wildlife.

Figure 13-5. Mule deer stressor model: Potential impacts of development.



Agricultural land uses may or may not provide foraging opportunities for mule deer, and farmers and ranchers may actively discourage deer foraging in their fields. Recreational uses, particularly off-road-vehicular use, can fragment habitat or disrupt mule deer movement patterns.

Phase II of this REA will assess the extent and pace of development in the U.S. portion of the ecoregion. Ruhlman et al. (2012) report very little change in the amount of land in development in the U.S. portion of the Chihuahuan Desert from 1973 to 2000 yet Pool et al. (2014) report an approximate six percent annual loss of grassland to agriculture in a portion of the Chihuahuan Desert in Mexico. The United States - Mexico borderlands are a center of increased road building and traffic, causing additional habitat loss and fragmentation. This development is also increasing illegal poaching. Much of the habitat degradation is tied to energy extraction within the area that occurs on public as well as private lands (New Mexico Department of Game and Fish 2006). Further information on the extent of land conversion is critically needed because all of these types of habitat loss and land use conversion take place over long time periods so the effects are difficult to quantify (Bergman et al. 2015).

Development may result in reduction in abundance due to vehicular collisions with deer as well as barriers to movement. Further study is needed on the number of vehicular deer collisions in the Chihuahuan Desert. However, past estimates for all deer in the United States ranged from 500,000 to 1.5 million deer killed annually by vehicles (Heffelfinger et al. 2006).

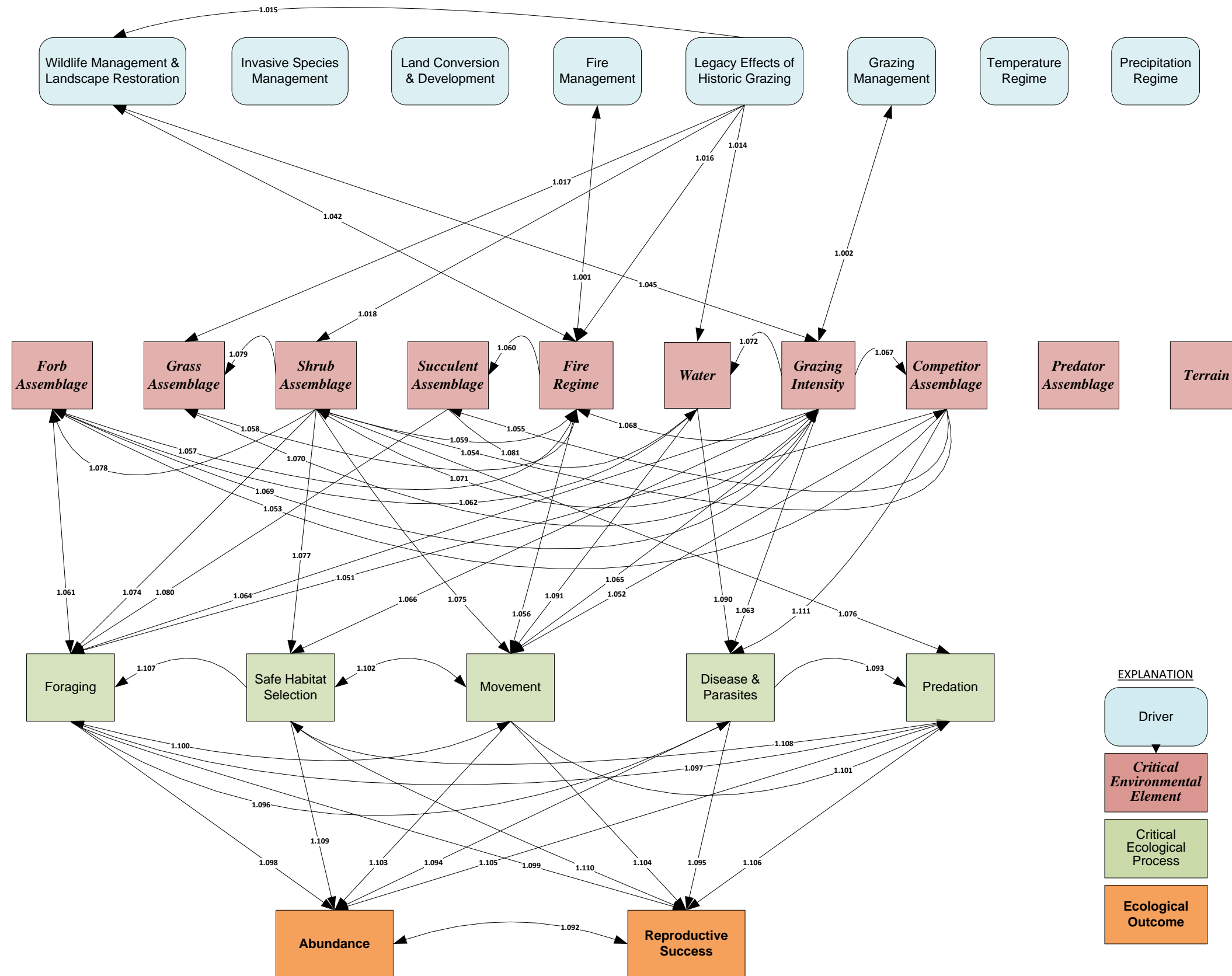
As development continues and habitat becomes more fragmented, deer movement and safe habitat selection will become increasingly limited. Deer will have fewer opportunities for foraging and for finding water in an already limited habitat where movement in response to changing conditions is essential. Populations may become physically and genetically isolated (Heffelfinger 2006; New Mexico Department of Game and Fish 2006). Fragmentation, road building and maintenance and off road vehicular use will foster the spread and establishment of invasive species (Brooks and Pyke 2001).

Land development potentially also can affect mule deer through its effects on fire management. Land development potentially can limit the use of prescribed fire near human developments for safety and health reasons including smoke impacts. Increasing development could also lead to the need for other types of fuel reduction treatments that could alter habitat. These changes in the fire regime would then influence the forb, grass, shrub and succulent assemblages. Increased land development also can contribute to the expansion of off-road vehicular use, which in turn can foster the spread and establishment of invasive species (Brooks and Pyke 2001).

13.3.5 Excessive Domestic Grazing

Chapters 2-3 discuss the ecological consequences of excessive domestic grazing across the U.S. portion of the ecoregion in general and Chapters 5-7 discuss the history and consequences of livestock grazing specifically across the terrestrial ecological systems of the region in which mule deer live. Figure 13-6 presents the stressor model for mule deer in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by non-native species and their management. Appendix 1 presents the rationale and citations for each causal link shown in Figure 13-6.

Figure 13-6. Mule deer stressor model: Potential impacts of excessive domestic grazing.



In general, excessive domestic grazing affects mule deer by altering vegetation, both directly through the grazing itself and indirectly through secondary effects on wildfire and the spread of non-native plants. Livestock can also compete with mule deer for space, food, and water, and transmit pathogens that also affect mule deer.

The legacy effects of historic grazing in the 1800s and early 1900s resulted in significant reductions in the abundance of the grass assemblage, soil loss and erosion, and stream incision, which in turn lowered water tables and reduced the availability of natural watering sites (Bahre 1995; 1991, Kerley and Whitford 2000, Parker 2002, Van Auken 2000). Historic grazing practices also affected the fire regime by reducing fine fuels from the grass assemblage and reducing fire frequency, intensity and area affected, thereby allowing for greater abundance in the shrub assemblage, as discussed above, this chapter, and in Chapters 2, 3, and 5-7. This altered vegetation assemblage is also enhanced by cattle eating seeds of some shrubs, particularly mesquite, they pass through their gut and thereby spread across a much wider area (Bahre 1995; 1991, McPherson 2006). Other human activities to expand agriculture and development resulted in further reductions in groundwater and surface water flow in streams and rivers. Changes in climate including reductions in summer rainfall and increases in winter rainfall likely favored increases in shrub abundance in the period from 1930 to the 1950s. Overgrazing enhanced this by accelerating soil erosion and reducing abundance of species in the grass assemblage (Neilson 1986). These interactions have created a landscape dramatically different from what existed prior to European settlement.

Grazing intensity is generally measured based on forage use, qualitative categories (light, moderate, severe) and quantitative measures of vegetation abundance (Holechek and Galt 2000). Grazing can alter plant community composition, reduce abundance of forage plants, and alter vegetation structure. One study found that cattle consumed 20 browse species, 22 forbs, 14 grasses and one succulent, with grasses constituting 57% of the diet. The same study found that mule deer consumed 45 browse species, 33 forb, 12 grasses, and 6 succulents with grasses less than 9% of the diet and the majority browse and forbs. Deer and cattle thus did consume the same species, but based on a similarity index, a significant biological overlap occurred only in the spring (Alcala-Galvan and Krausman 2012). However, the large size of cattle and the degree of grazing intensity can result in a significant reduction in forbs and browse (Heffelfinger et al. 2006).

Mule deer tend to avoid areas with large numbers of cattle. Both deer and cattle use washes and other xeroriparian areas that provide surface water and a diversity of forbs and preferred shrubs. If such areas are grazed heavily by cattle, deer will not be able to secure sufficient forage (Heffelfinger et al. 2006). In a study done during relatively high precipitation, deer preferred ungrazed pastures to grazed pastures (Ragotzkie and Bailey 1991).

Sheep have similar diets to deer and can reduce forage. Sheep ranching is still practiced on Native American lands (Heffelfinger et al. 2006).

A literature review of grazing impacts on arid western systems indicates soil porosity, litter cover and vegetation cover are negatively affected. The reduction in forb assemblage, along with grass and shrub

assemblages, reduces forage quality and the nutritional value of the plant community (Jones 2000). However, grazing impacts can be highly variable. In one study, ungrazed areas had lower cover of the forb assemblage, but light to moderate grazing resulted in higher cover of the forb assemblage (Holechek 1991). On the other hand, removing cattle from a northern Arizona grassland for five years did not significantly improve either cover of the forb assemblage nor cover that provided concealment from predators (Loeser et al. 2005). A study of an area where cattle had been excluded for 16 years found an increase the cover of annual forbs, but that increase was statistically significant for only a few species, possibly because annual forbs are highly dependent on precipitation (Kelt and Valone 1995).

The sustainability of grazing intensity depends on precipitation. Grasses and forbs need time to recover from grazing. Range condition with low intensity grazing increased in years with sufficient precipitation and decreased during drought (Molinar et al. 2011, Navarro et al. 2002). Light stocking is recommended following years of below average precipitation (Holechek et al. 2003; 1994, Khumalo et al. 2007). Short term, high intensity grazing is sometimes practiced and may decrease water infiltration as cattle in large numbers compact the soil (Heffelfinger et al. 2006). Deer often occupy riparian and xeroriparian habitat that are also used for livestock grazing and often are overused (Heffelfinger et al. 2006). Grazing and fire regimes and wildlife management and landscape restoration programs focusing on vegetation disturbances need to be implemented to promote seral stages beneficial to mule deer (Cantu and Richardson 1997, Zwartjes et al. 2005).

Within desert systems that have low productivity, reductions in plants needed for nutrition will be detrimental to deer populations as well as to raising juveniles. Deer feed on both shrubs and forbs primarily, so any significant reduction in the forb assemblage or shrub assemblages, would result in reductions in forage quality (Jones 2000).

13.3.6 Landscape Restoration

Chapters 3 and 5-10 discusses the types of landscape restoration projects taking place in the U.S. portion of the ecoregion, some of which currently or in the future could affect mule deer; and mule deer themselves are managed as a game species in the U.S. portion of the ecoregion. Figure 13-7 presents the stressor model for mule deer in the U.S. portion of the ecoregion, simplified to show only those causal relationships that address landscape restoration and mule deer management. Appendix 1 presents the rationale and citations for each causal link shown in Figure 13-7.

Management of mule deer involves three activities: (1) regulating hunting to provide for recreational hunting while sustaining population numbers, (2) controlling predators to reduce predation on deer, and (3) habitat restoration to enhance deer and other wildlife habitat.

Mule deer hunting in the U.S. portion of the ecoregion is regulated by the Texas Parks and Wildlife Department, New Mexico Department of Game and Fish, and Arizona Game and Fish Department. Each of these agencies has hunting regulations specifying the number of deer that can be taken by each hunter. Harvest rates are generally determined based on estimated buck: doe ratios (Cantu and Richardson 1997). Hunting is therefore a source of adult deer mortality, primarily males, though the control of the number of issued hunting permits limits hunting as a significant pressure on deer

population (Cantu and Richardson 1997). Wildlife biologists use surveys including spotlight surveys after dark or daylight helicopter surveys to determine the status of the population and set permit numbers for upcoming hunting seasons (Cantu and Richardson 1997). In Mexico, hunting is regulated by the Ministry of Environment and Natural Resources or SEMARNAT). Mule deer translocation has been successfully practiced in Mexico to reintroduce deer (Ortega-Sanchez 2013).

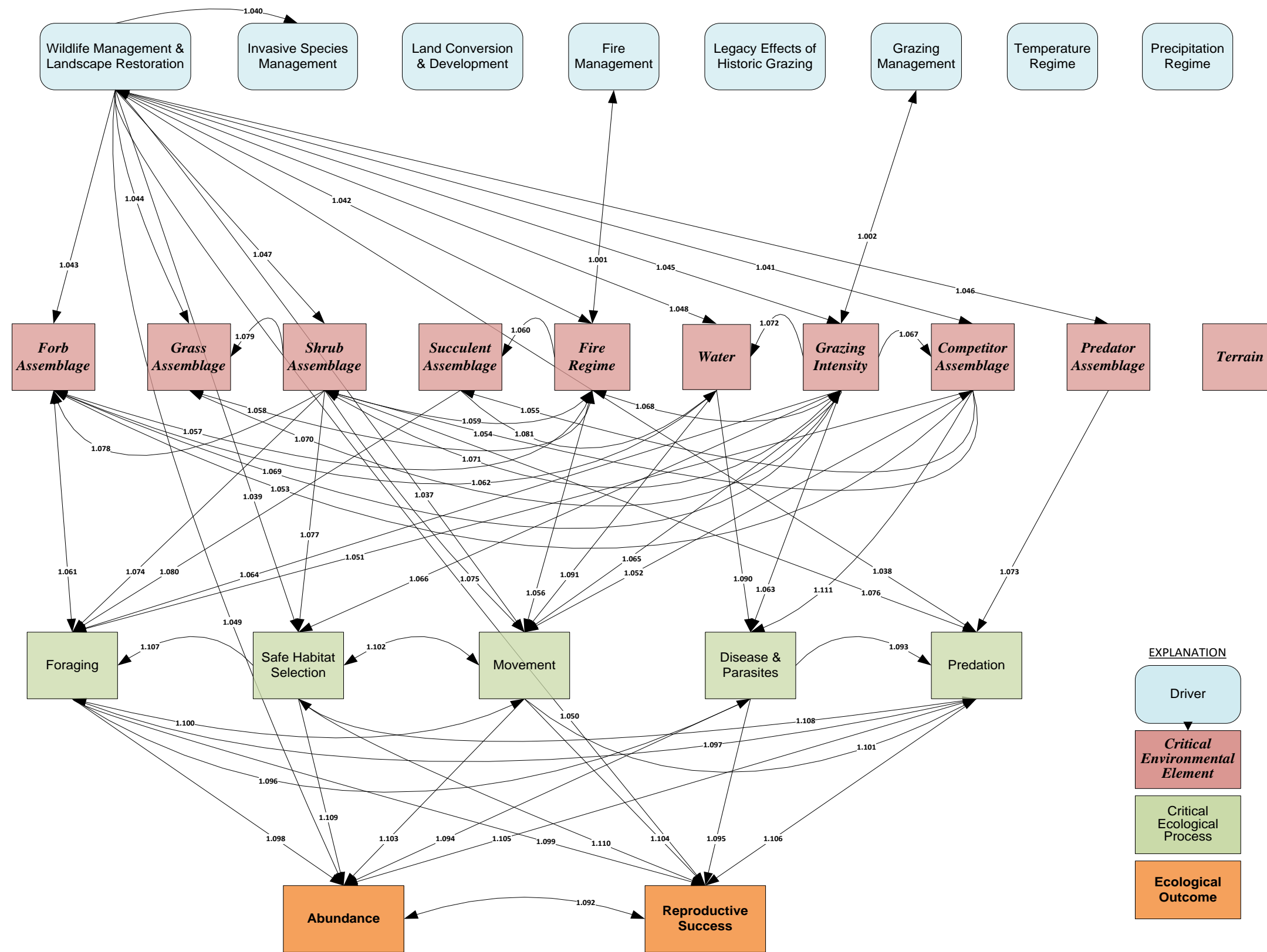
Predation causes mortality in adult deer but has a greater effect on fawns as they are particularly vulnerable in the first few months following birth. Predation was found to be a factor along with rainfall and forage abundance in fawn survival in the Trans-Pecos region (Lawrence et al. 2004). As discussed above, males typically have larger home ranges than do females, they tend to use more difficult terrain, and they are more solitary. Therefore they may be more prone to predation as well as injury from accidents (Cantu and Richardson 1997). Control of the predator assemblage has been shown to increase deer survival, particularly fawns where coyotes have been reduced. However, mortality is more closely correlated with the health and nutrition of the deer, with malnourished deer more prone to predation (Bergman et al. 2015, Smith and Lecount 1979). In fact, control of predators may also have unintended consequences. For example, the litter size of coyote increases following local population control, and juveniles become sexually mature earlier, likely as a result of reduced competition. Coyote population control may be more successful if completed after the establishment of dominance and territorial patterns for the coming breeding season and prior to whelping to prevent establishment of new breeding pairs (Connolly 1978). In any case, control of the predator assemblage may only benefit mule deer management where forage or cover are limited or where other environmental conditions limit recruitment. If the mule deer population is near carrying capacity, predator control is unlikely to be useful (Lawrence et al. 2004).

Water is a limiting factor for mule deer abundance and reproduction, unless the deer can get sufficient water from forage or natural water sources. The provision of artificial water sources can be important to supporting populations in the desert. At the same time, predators may be attracted to water sources due to deer numbers and may kill deer near these artificial water sources (Heffelfinger 2006). Artificial water sources may also significantly alter seasonal movement patterns of deer as well as attracting females in need of water during lactation (Rautenstrauch and Krausman 1989, Relyea et al. 2000). Water sources may become contaminated or augment the transmission of disease and parasites (Heffelfinger et al. 2006).

Efforts to improve range condition following historic overgrazing go back to before World War II (Cox et al. 1984). Methods included livestock management to reduce grazing impacts, site manipulation to reduce erosion, reseeding with native and nonnative grasses, shrub removal, and small mammal control to reduce seed predation (Herrick et al. 2006). Restoration management for deer can involve a number of actions from modifying grazing intensity to providing foraging opportunities, the use of herbicides, mechanical and fire treatments to reduce shrub growth, and seeding plant species to enhance forb and shrub assemblages (Bender 2012, Cantu and Richardson 1997, Heffelfinger et al. 2006). Herbicides have been effective in reducing the extent of the shrub assemblage but can also reduce abundance of the forb assemblage (Harveson et al. 2012). At Boracho Peak Ranch in Texas, over 21,600 ha were treated with Spike 20P herbicide (Tebuthiuron). The control of creosote and tarbush was effective and grasses

increased in abundance, but forb richness was reduced to half that found in untreated areas (Gage et al. 2011). Mechanical removal of shrubs has resulted in mixed results. In some cases, shrub removal has resulted in increases in the abundance of the grass or forb assemblages and in others no increase was found, though the latter occurred where shrub density was low and forbs and grasses had less competition from shrubs (Buonopane et al. 2005, Mata-Gonzalez et al. 2007).

Figure 13-7. Mule deer stressor submodel for wildlife management and landscape restoration (See Appendix 1 for link details).



Habitat management to benefit mule deer needs to balance the requirements for forage, movement and safe habitat selection. For example, mule deer feed on both shrubs and forbs, using different species in different seasons. Forbs and new shrub growth are particularly important for doe and fawn nutrition. However, an overabundance of shrubs can impede movement and can reduce forb growth. A total lack of shrubs would result in a loss of a significant food resource.

Restoration of riparian areas may be undertaken to improve habitat for both terrestrial and aquatic species. These areas also are often used by cattle as they may have standing water and higher plant species abundance. These habitats have become degraded due to overgrazing, periodic dumping of sewage into the river, and gravel extraction (Cornell et al. 2008). Restoration can involve reducing grazing intensity, reducing invasive species abundance and replanting with native species (Heffelfinger et al. 2006). Replanting techniques range from reseeding to planting live plants or using cut stems. Replanting is often undertaken to reduce erosion and stabilize the channel (Dreesen et al. 2002). At Bosque del Apache National Wildlife Refuge, restoration involved removal of salt cedar (*Tamarix* spp.) using mechanical means and herbicides and controlled flooding to encourage natural regeneration of native species (Dreesen et al. 2002). However, it is not known whether such efforts affect mule deer use of riparian areas.

In the Trans-Pecos, white-tailed deer (*Odocoileus virginianus*) are sympatric with mule deer and represent a member of the competitor assemblage. Mule deer prefer lower shrub cover than white-tailed deer (Avey et al. 2003). Private landowners may wish to manage for one or both species as a source of income from hunters. This could lead to either increases or reductions in habitat for mule deer depending on which species brings the landowner the greater return on investment.

13.4 Mule Deer Key Ecological Attributes

All ecological outcomes and critical ecological processes in the mule deer stressor model constitute the key ecological attributes for the CE. The list below identifies 7 key ecological attributes for mule deer based on these criteria. *Fully characterizing the present condition of the CE will require data on indicators for its key ecological attributes.* Indicators are determined during Phase II of the REA process. The definitions for the key ecological attributes are the same as the definitions for these model components presented above.

- **Ecological Outcomes**
 - Abundance
 - Reproductive Success
- **Critical Ecological Processes**
 - Disease and Parasites
 - Foraging
 - Movement
 - Predation
 - Safe Habitat Selection

14 Banner-tailed Kangaroo Rat Conceptual Model

This chapter presents the conceptual ecological model for the banner-tailed kangaroo rat, *Dipodomys spectabilis* (Merriam 1890). The banner-tailed kangaroo rat is a nocturnal, granivorous heteromyid rodent found throughout the grasslands of the Chihuahuan Desert. The species ranges from northeastern Arizona (Hoffmeister 1986) into southern Arizona and across most of New Mexico and western Texas south into Mexico (Findley et al. 1975, Best 1988; 1999, Fitzgerald et al. 1994). In New Mexico and western Texas, the species is found chiefly on mesa tops and the foothills of desert ranges (Bailey 1932, Schmidt-Nielsen 1964). It is also found in open creosote deserts (Hoffmeister 1986). The presentation of the banner-tailed kangaroo rat conceptual model follows the structure described in Chapter 4, with sections on sources of information, a species overview, the stressor model, and key ecological attributes. As noted in Chapter 4, the conceptual models for species and species assemblages do not include a separate control model.

14.1 Sources of Information

The banner-tailed kangaroo rat overview and stressor model integrate information from several sources, including summaries of the ecology of the species both in general across its entire range and specifically within the Chihuahuan desert (e.g., Vorhies 1922, Bailey 1932, Holdenried 1957, Schmidt-Nielsen 1964, Rosenzweig and Winakur 1969, Findley et al. 1975, Schroder 1979, Moroka et al. 1982, Jones 1984; 1986, Randall 1987, Brown et al. 1988, Moorhead et al. 1988, Hawkins and Nicoletto 1992, Valone et al. 1995, Guo 1996, Hawkins 1996, Brown et al. 1997, Whitford 1997, Anderson and Kay 1999, Eve et al. 1999, Ayarbe and Kieft 2000, Curtin et al. 2000, McPherson and Weltzin 2000, Curtin et al. 2002, Krogh et al. 2002, Waser and Ayers 2003, Skyvarla et al. 2004, Waser et al. 2006, Davidson and Lightfoot 2007, Davidson et al. 2008, Linzey and Timm 2008, Cosentino et al. 2014).

14.2 Banner-tailed Kangaroo Rat Overview

The Chihuahuan desert is an important ecoregion that is home to a large number of rodent species. The banner-tailed kangaroo rat is one of these species. While this species can be locally common (Bailey 1932, Findley et al. 1975) it is listed as “Near Threatened” by the International Union for Conservation of Nature (IUCN) due to widespread degradation of its desert grassland habitat throughout much of its range (Linzey and Timm 2008).

14.2.1 Distribution

The banner-tailed kangaroo rat is a mound-building rodent, and is considered an ecological engineer or keystone species that dramatically affects the community structure of both plants and animals (Brown and Heske 1990, Kerley et al. 1997, Curtin et al. 2000, Davidson et al. 2008, Davidson and Lightfoot 2007, Moorhead et al. 1988, Moroka et al. 1982). Banner-tailed kangaroo rat mounds increase soil heterogeneity, provide novel microhabitats for a number of rare plants and animals (Ayarbe and Kieft

2000, Guo 1996; Moorhead et al. 1988, Whitford 1997), and create expansive burrow networks that are used by many other organisms for shelter (Davidson et al. 2008, Davidson and Lightfoot 2007, Hawkins 1996, Hawkins and Nicoletto 1992). Banner-tailed kangaroo rats prefer heavier soils with well-developed short-grass cover (Anderson and Kay 1999, Findley et al. 1975, Schmidt-Nielsen 1964) as only the heavier soils may be compatible with the extensive and complex network of tunnels and chambers dug by the species (Findley et al. 1975).

The population of this species has declined from historical numbers (Waser and Ayers 2003, Whitford 1997). Investigators have attributed much of this decline to increased encroachment of woody vegetation into grassland habitats (Brown et al. 1997, Krogh et al. 2002, Valone et al. 1995, Whitford 1997). Woody vegetation density or, more correctly stated, grassland habitat area of cover, is the key indicator for the viability of this species. Banner-tailed kangaroo rats do not occur where there is more than 20% shrub cover (Anderson and Kay 1999, Krogh et al. 2002, Moroka et al. 1982, Waser and Ayers 2003). Future conversion of the Chihuahuan grasslands to shrublands, or to bare ground, will have severe negative consequences on this species.

Other challenges facing the banner-tailed kangaroo rat include the ongoing invasion of exotic and native invasive species, managing cattle grazing to avoid the mistakes of the past, altered fire regimes, climate change, increased human use, urban & industrial growth, and landscape restoration. Each of these is discussed below. These issues facing the banner-tailed kangaroo rat are complex, but the need for action is evident. At one time Vorhies (1922) described the banner-tailed kangaroo rat as so numerous as to be a conspicuous feature of the landscape. Presently, Eve et al. (1999) show that 27.5% of Chihuahuan Desert grasslands have already been severely degraded and suggest that banner-tailed kangaroo rats could become extinct sometime within the next 100 years.

14.2.2 Habitat

Banner-tailed kangaroo rats favor open desert grasslands within the Chihuahuan Desert and avoid areas with tall or dense grass cover or shrub cover greater than 20% (Anderson and Kay 1999, Krogh et al. 2002, Moroka et al. 1982, Rosenzweig and Winakur 1969, Schroder 1979, Waser and Ayers 2003). Unfortunately, many of these desert grasslands have been converted to shrublands during the past 150 years (Bahre and Shelton 1993, Brown et al. 1997, Buffington and Herbel 1965, Eve et al. 1999, Roundy and Biedenbender 1995). The term shrubland is used rather generically to include all woody vegetation, especially mesquite (*Prosopis* spp.) and creosote bush (*Larrea tridentata*). The cause of this shift to shrubland from grassland is not well understood and may be a matter of changing climate (Brown et al. 1997), cattle grazing (Curtin et al. 2002), fire suppression (McPherson and Weltzin 2000), or, most likely, interactions of all of these factors. This conversion of grasslands to shrublands is a major driver in the decline of this species. Nevertheless, banner-tailed kangaroo rats have been able to persist in the small patches of open grasslands that still remain (Waser and Ayers 2003).

Banner-tailed kangaroo rats are mostly solitary animals except, for periods when their young remain in their natal home range until they can secure their own mound (Jones 1986; 1984). Banner-tailed kangaroo rat runways do not appear to connect to adjacent mounds (Schroder 1979) and (Holdenried 1957) concluded the banner-tailed kangaroo rat confine their activities to small local areas of less than

152 meters distance from their mound. This small home area may limit dispersal and may be an important issue for increasing colonization into newly restored habitats (Cosentino et al. 2014, Skyvarla et al. 2004, Waser et al. 2006). More research is needed on this question. Banner-tailed kangaroo rats are active all year, they are nocturnal and they do not hibernate or estivate (Vorhies 1922). Males go to the home area of an estrous female to mate and when more than one male is present, competition for access to an estrous female occurs, and females may mate with more than one male (Randall 1987).

Several species prey upon banner-tailed kangaroo rats including badgers (*Taxidea taxus*), kit foxes (*Vulpes macrotis*), bobcats (*Lynx rufus*), coyotes (*Canis latrans*) and great horned owls (*Bubo virginianus*), as well as a number of snakes and other raptors (Brown et al. 1988; Vorhies 1922). Several predators including burrowing owls (*Athene cunicularia*) and Mohave rattlesnakes (*Crotalus scutulatus*) have also declined simultaneously with banner-tailed kangaroo rats, perhaps because they depend on banner-tailed kangaroo rat burrows for nesting sites (Brown et al. 1997).

14.3 Banner-tailed Kangaroo Rat Stressor Model

Table 14-1 presents and defines the drivers, critical environmental elements, critical ecological processes, and ecological outcomes represented in the banner-tailed kangaroo rat stressor model. The stressor model follows the methodology for species described in Chapter 4.

Table 14-1. Banner-tailed kangaroo rat definitions of stressor model components.

Model Component	Definition
Drivers	
Fire Management	Human activities to control the size, timing, intensity, or frequency of fires. Fire management applies to the control and prevention of wildfire as well as the use of prescribed fires to achieve management goals and objectives.
Grazing Management	Human activities to manage and control where and how domestic cattle are grazed within the Chihuahuan Desert. This includes the number of cattle grazed per hectare or other measures of grazing intensity as well as any grazing rotation strategies.
Human Uses and Urban & Industrial Growth	Human activity that converts native habitats (shrubland or grassland) into developed areas and includes barriers to movement, collisions with motor vehicles, attraction of domesticated predators, etc. Developed areas include urban, suburban, industrial, and agricultural development. Also refers to areas of intense recreational, military, borderland security, and other anthropogenic activities.
Invasive Species Management	Human activities to reduce the abundance of invasive species in the Chihuahuan Desert. This includes two primary foci: exotic grasses and native and exotic shrubs. This management may include mechanical removal, herbicide application, and native grass plantings.
Legacy Effects of Historic Grazing	Ongoing impacts to vegetation community composition and structure, especially increasing conversion of grasslands to shrublands, resulting from the legacy effects of unsustainable domestic cattle grazing in the 19th and early 20th centuries.
Precipitation Regime	The pattern, timing, intensity, and average annual rate of precipitation in the Chihuahuan Desert.
Restoration Management	Human activities including mechanical shrub removal, herbicide application and native grass plantings, which maintain or increase the percent cover of native grasses and/or reduce the percent of shrub cover in a specific region. This driver does not include invasive species management, fire management, or grazing management, which are treated separately in this framework.

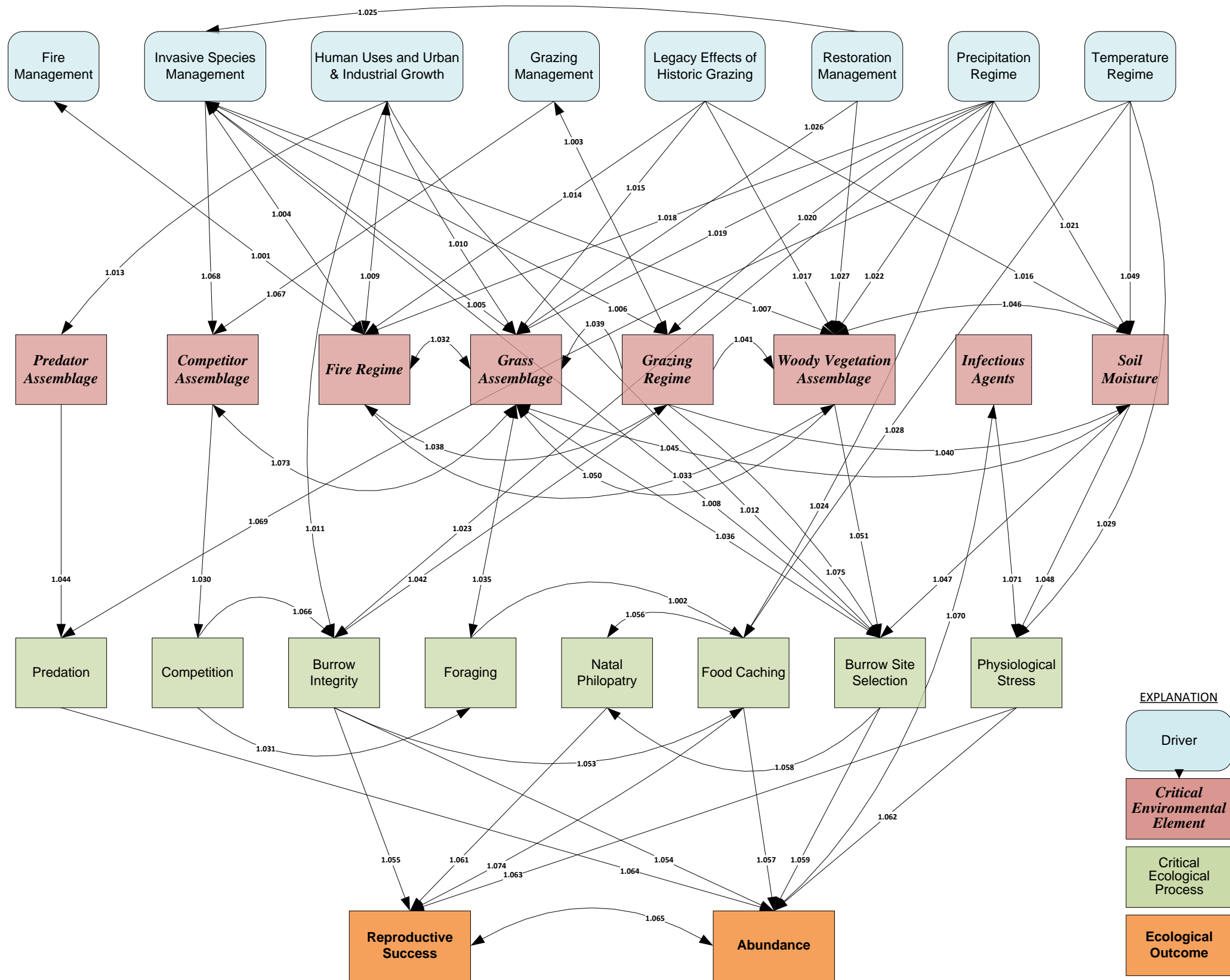
Model Component	Definition
Temperature Regime	The patterns of air temperature in the Chihuahuan Desert within and across all seasons.
Critical Environmental Elements	
Competitor Assemblage	The composition, abundance, and activity level of species that compete with banner-tailed kangaroo rats for food, habitat, or other resources.
Fire Regime	The frequency, intensity, severity, and seasonality of both wildfire and prescribed fire.
Grass Assemblage	The composition and abundance of grass species.
Grazing Regime	The number of cattle grazed per acre and the frequency of grazing rotations. In the literature, grazing intensity, indicated by greater numbers of cattle and higher frequencies of grazing, is generally characterized as light, moderate, or heavy.
Infectious Agents	The presence and abundance of pathogens and parasites that can weaken or kill banner-tailed kangaroo rats.
Predator Assemblage	The composition, abundance, and activity level of species that prey upon banner-tailed kangaroo rat.
Soil Moisture	The average soil moisture in an area across all seasons.
Woody Vegetation Assemblage	The composition and abundance of woody species, including shrubs as well as trees. Increased abundance is associated with desertification.
Critical Ecological Processes	
Burrow Integrity	The frequency with which banner-tailed kangaroo rat burrows maintain structural stability and shelter, regulate humidity and temperature, and store food reserves.
Burrow Site Selection	The frequency and extent to which the landscape provides a suitable amount and spatial arrangement of grassland, shrubland, and grass-shrub composite present in the Chihuahuan Desert for banner-tailed kangaroo rat to successfully secure burrows that provide adequate shelter and opportunity for reproduction.
Competition	The intensity with which banner-tailed kangaroo rats contend with other species for territory, food, mounds, and other resources.
Food Caching	The rate at which banner-tailed kangaroo rats are able to successfully store food inside their burrows.
Foraging	The rate of success of banner-tailed kangaroo rats to actively find and consume food resources of suitable quality within their habitats.
Natal Philopatry	The rate at which banner-tailed kangaroo rats engage in prolonged retention of offspring in natal home ranges.
Physiological Stress	The frequency and intensity of factors that negatively affect the health of banner-tailed kangaroo rats.
Predation	The rate of predation on banner-tailed kangaroo rats by species such as snakes, raptors, bobcats, foxes, coyotes, domestic animals, etc.
Ecological Outcomes	
Abundance	Numbers of banner-tailed kangaroo rats.
Reproductive Success	The rate with which banner-tailed kangaroo rats successfully give birth and raise young to self-sufficiency.

Figure 14-1 shows the full banner-tailed kangaroo rat stressor model, displaying the model components listed in Table 14-1 along with their causal relationships. Appendix 1 describes and presents the rationale for including every causal relationship in the stressor model, with citations for each causal relationship, providing a comprehensive presentation of the stressor model.

Figure 14-1 indicates the presence or absence of causal relationships between the system model components but does not indicate the potential magnitude or other characteristics of these relationships, as explained in Chapter 4. Figure 14-1 identifies the causal relationships that have affected

how the system likely has changed in the past, in response to changes in its drivers. The diagram provides a means for articulating how the condition of the system will likely change in response to changes in its drivers. As discussed and illustrated below, the stressor model makes it clear: (a) which critical environmental elements would likely be affected by a change in one or more particular drivers, including change agents; (b) which critical ecological processes would likely be affected by the cascading effects of these changes in critical environmental elements; and (c) which system characteristics (ecological outcomes) would likely be affected by the cascading effects of these changes in critical environmental elements and ecological processes. By doing so, in turn, the stressor model also highlights those components of the model—drivers, critical environmental elements, critical ecological processes, and ecological outcomes—that demand indicator data.

Figure 14-1. Banner-tailed kangaroo rat stressor model.



As defined earlier, a causal relationship exists when a change in one component of the system results in a change in some other component. Change in the first component is said to “cause a change in the second component. Each chain of causation, from driver to outcome, describes how the condition of the system likely has changed in the past, or likely would change in the future, in response to changes in its drivers.

The following paragraphs discuss how each Change Agent affects the banner-tailed kangaroo rat in the U.S. portion of the ecoregion, as articulated in the stressor model for this CE. A “sub-model” diagram for each Change Agent presents a simplified version of the master stressor model diagram, showing only the direct and indirect effects of the Change Agent of interest. The text accompanying each sub-model diagram summarizes information from Appendix 1 concerning the impacts of each Change Agent, including selected citations. For each Change Agent, the paragraphs summarize present understanding of: (a) which critical environmental elements would likely be affected—directly or indirectly—by each Change Agent and (b) which critical ecological processes would likely be affected – directly or indirectly—by the cascading effects of these changes in critical environmental elements.

14.3.1 Climate Change

The final report for the Chihuahuan Desert REA will include a discussion of current forecasts of the ways in which climate change will affect the ecoregion and its ecological resources. The present chapter presents only the conceptual model of the causal relationships and outcomes that potentially will be affected for the banner-tailed kangaroo rat. Figure 14-2 presents the banner-tailed kangaroo rat stressor model for the U.S. portion of the ecoregion, simplified to show only those causal relationships that potentially will be affected by changes in the air temperature and precipitation regimes. Appendix 1 presents the rationale and citations for every causal link shown in the diagram.

Climate change directly affects two significant drivers, the precipitation and temperature regimes. Changes in these regimes have profound impacts on nearly all of the critical environmental elements identified in the banner-tailed kangaroo rat stressor model, including grass assemblage, woody vegetation assemblage, soil moisture, grazing regime, infectious agents, and fire regime, as well as several critical ecological processes including predation, food caching, burrow integrity, and physiological stress.

Changes in temperature and precipitation patterns could have profound effects on two other drivers: grazing management and fire management. Navarro et al. (2002) suggests a moderate, controlled amount of livestock grazing is sustainable on the Chihuahuan Desert rangelands receiving 26-35 cm of precipitation annually. However, when overgrazing is coupled with drought in the Chihuahua Desert, woody vegetation increases (Brown et al. 1997). During a drought, the amount of forage available to cattle is reduced and may lead to increased intensity of grazing to support existing herds (Kerley and Whitford 2000). As a result, grazing can exacerbate the effects of drought and may lead to woody vegetation encroachment. Along with precipitation patterns, grazing practices need to be monitored and adjusted as needed to limit the invasion of woody vegetation (Beck et al. 2007) (see Chapters 5-7, above).

Prescribed burns followed by a drought also can result in further degradation of the grasslands (Ladwig 2014). Because precipitation is so difficult to predict, Ladwig (2014) suggests that fire managers should schedule prescribed burns only after precipitation events. In conditions where precipitation is higher, fire managers may use prescribed burns as a beneficial tool in maintaining the health of the grasslands. However, some people question the benefits of prescribed burns in these grasslands because fire-induced mortality of mesquite is low and fire has an adverse effect on black grama grasslands (Kilgore et al. 2009), as discussed below (see Uncharacteristic Wildfire, this chapter). Nevertheless, fire dynamics must be considered in context with precipitation patterns and managed based upon the amount and intensity of precipitation events. Further research may be needed to determine if prescribed fires actually help restore banner-tailed kangaroo habitat.

Most climate change models predict that over the next 100 years the Chihuahuan Desert will become hotter and drier, and experience more extreme weather events; but they also indicate that the response of arid lands to climate change will be strongly influenced by interactions with non-climatic factors (Archer and Predick 2008). The effects of a drier environment will most likely favor woody vegetation because of their deep roots and ability to reach moisture deeper in the soil (Backlund et al. 2008, White et al. 2011). This increase in woody vegetation will be at the expense of grasslands, and banner-tailed kangaroo rat habitat will be lost. As noted above, grass assemblage conditions affect burrow site selection, and banner-tailed kangaroo rats do not live in habitat where shrub cover is greater than 20%.

Changes in the grass assemblage could affect banner-tailed kangaroo rat foraging, and could reduce the banner-tailed kangaroo rat's ability to secure an adequate food cache. Drier conditions will entail lower primary productivity, which will lower food resource availability for granivores that feed primarily upon grass seeds (Kerley and Whitford 2000). Insufficient food caching would then affect abundance. Insufficient food caches can also affect natal philopatry as juvenile rats depend on parental food caches while maturing and prior to securing their own mounds. Any negative impacts on the young rats will affect reproductive success and ultimately abundance.

The ecological impacts of climate change on precipitation in the Chihuahuan Desert depend not only on how much precipitation falls, but also on the timing of the precipitation. Generally, most rain falls in the late summer and is associated with the summer monsoon with another peak of precipitation in the winter. The timing of these rains is important. If the phenology of the precipitation changes, even if the amount of precipitation does not change, there can be profound effects on the plant community as well as on the banner-tailed kangaroo rat. An increase in winter rains will favor the growth of woody invasive shrubs as well as annual plants that uptake soil nutrients and reduce nutrient availability for warm season grasses that start growing later in the year (Burgess 1995, Neilson 1986). Creosote bush and mesquite are C₃ species best adapted to wet winters, while black grama grass is a C₄ species adapted to wet summers. As a result, increases in winter precipitation tend to favor the woody vegetation assemblage (Neilson 1986). Summer rains are also often associated with intense rain events that can drop large amounts of precipitation in a short time. Brown et al. (1997) has suggested that since the 1970s there has been an abnormally high amount of winter precipitation and they suggest this is one of the major drivers for increased shrub growth. However, it is not known if shrub density will decrease if winter precipitation decreases with climate change.

Figure 14-2. Banner-tailed kangaroo rat stressor model: Potential impacts of climate change.



In addition to changes in the overall amount of precipitation, and possibly in the phenology of the precipitation, many climate models forecast more extreme precipitation events. These extreme precipitation events are often associated with the remnants of hurricanes or unusually severe thunderstorms and can have negative effects on the banner-tailed kangaroo rat. Extreme rainfall events have been associated with the collapse of mounds, resulting in the potential death of the banner-tailed kangaroo rat, the destruction of the banner-tailed kangaroo rat's food stores or both (Valone et al. 1995). Furthermore, these large precipitation events have the potential to soak the food cache in the mound and cause the seeds to sprout or show extreme fungal growth - perhaps even mycotoxin growth toxic to the banner-tailed kangaroo rat (Valone et al. 1995). In addition, heavy precipitation events allow water to penetrate deeply into the soil which promotes the survival and growth of long-lived shrub species, leading to increased loss of banner-tailed kangaroo rat habitat (Thibault et al. 2010).

The potential effects of increased temperature on the banner-tailed kangaroo rat will likely be complex and mainly determined by how these temperature changes affect the plant community and overall primary productivity (Throop et al. 2012). However, temperature can also directly affect banner-tailed kangaroo rats. Banner-tailed kangaroo rats are well adapted to hot, dry environments, but do not have a good means to dispel excess heat from their bodies. They cannot sweat or pant, and because they live in a water-stressed environment, they mainly rely on behavioral adaptations to maintain body temperature. Moses et al. (2012) showed that even small increases in body temperature for the banner-tailed kangaroo rat can be fatal. While it is hard to predict how banner-tailed kangaroo rats will adapt to increased temperatures, it will potentially add additional physiological stress to the animal.

Increased temperature may also have several indirect negative effects on the banner-tailed kangaroo rat. Increased temperature may change the phenology of banner-tailed kangaroo rat predators, such as snakes, increasing the time they are active and able to prey on the banner-tailed kangaroo rat. Elevated land surface temperature coupled with high aridity can cause plant material and seeds to desiccate and may reduce available water in forage and potentially leading to negative water balance and reduced fitness (Alpert 2000, Schmidt-Nielsen 1964, Waser and Jones 1991). In order to compensate for lost water, banner-tailed kangaroo rats must increase foraging in order to secure more seeds as they convert sugars in seeds to metabolic water (Moses et al. 2012). Lastly, the interaction of temperature with moisture is important for the maintenance of proper humidity in the burrow. Too little humidity causes an increase in the loss of water through respiration of the banner-tailed kangaroo rat, while too much humidity can lead to spoilage of the food reserves.

Moisture and temperature conditions are important drivers for disease among banner-tailed kangaroo rats (Kolivras and Comrie 2004). Outbreaks of several different infectious agents in the southwestern U.S., including mosquito borne viruses, sylvatic plague, hantavirus, and coccidioidomycosis, have been conclusively linked to climatic factors (Glass et al. 2002, Kolivras and Comrie 2004). A number of parasites are associated with the banner-tailed kangaroo rats, including protozoan parasites (Stout and Duszynski 1983) and a cestode (Guay and Senger 1962). Many ticks, mites, and fleas are also associated with banner-tailed kangaroo rats (Eads et al. 1952, Graves et al. 1974, Radford 1953, Vorhies 1922). Banner-tailed kangaroo rats are resistant to sylvatic plague infection (Holdenried and Quan 1956), which is often associated with black-tailed prairie dogs (see Chapter 15). In most cases, moisture does not

directly cause disease, but increases in precipitation increase the food sources of the small mammals and this in turn increases abundance (Parmenter et al. 1999). These increases in small mammal numbers provide a greater chance that disease will spread throughout populations, increasing the number of infected individuals. Future risks of these diseases due to climate change are very difficult to predict (Balbus and Wilson 2000). In addition, it is not well understood how decreased precipitation or changes in the phenology of precipitation might affect infectious diseases.

Changes in the temperature regime also affect food caching. Increased temperature can desiccate the seeds, reducing the water available in the seeds and requiring the banner-tailed kangaroo rats to work harder to secure adequate moisture for survival. Increased temperature can also reduce soil moisture, which in turn reduces humidity in the mound, causing increased loss of water through respiration. This increase in water loss can lead to increased physiological stress and result in decreased abundance. Increased temperature regimes may also directly affect the banner-tailed kangaroo rats by increasing body temperature beyond critical thresholds, causing physiological stress and decreased abundance. Increased temperature regimes may also increase ectothermic predator seasons of activity. This predation will increase mortality and further reduce banner-tailed kangaroo rat abundance.

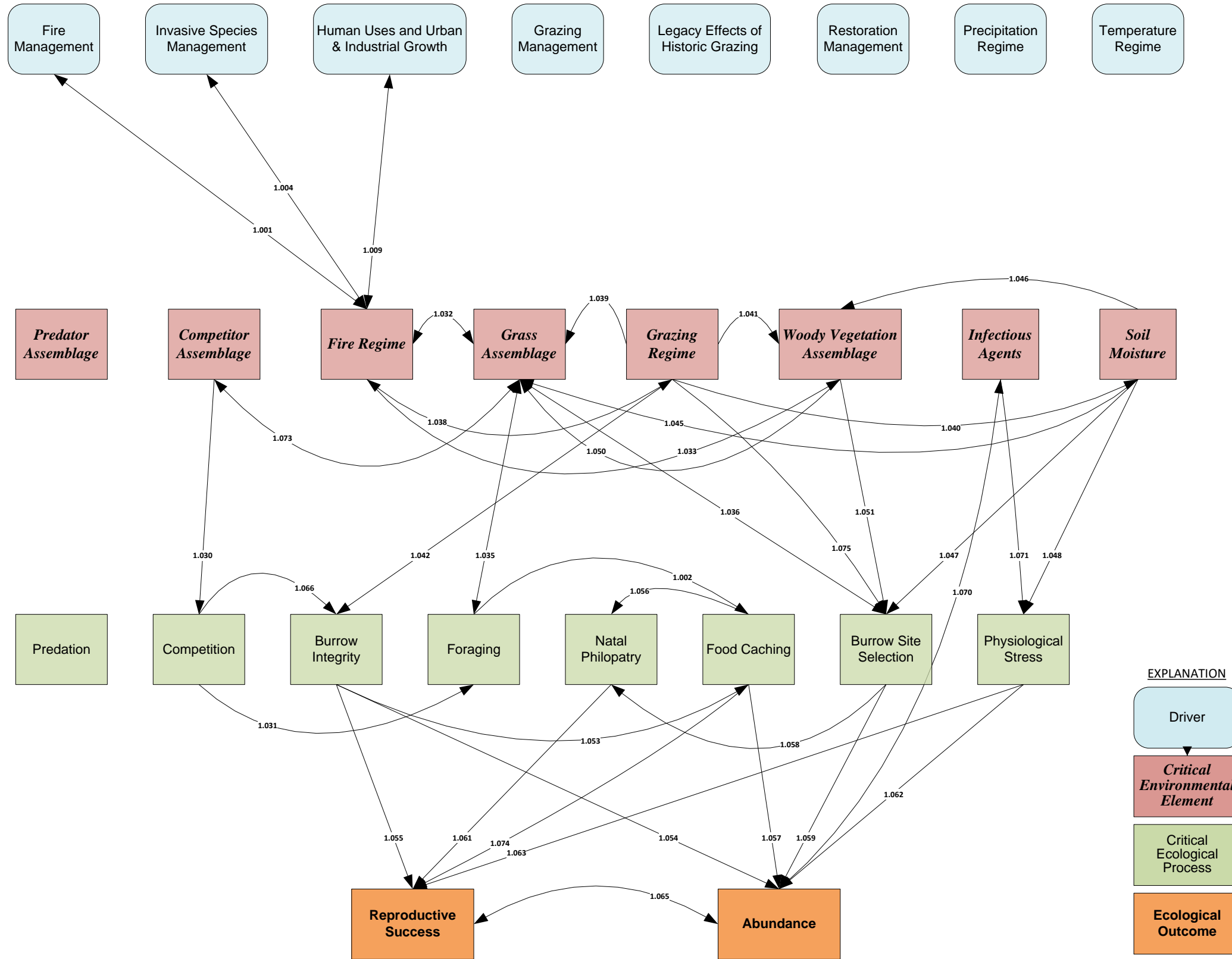
14.3.2 Uncharacteristic Wildfire

The fire regime of the U.S. portion of the Chihuahuan desert has changed as a result of the interaction of several drivers, as discussed in several previous chapters. Chapters 2 and 3 discuss the history, causes, and consequences of uncharacteristic wildfire across the ecoregion in general, and Chapters 4-6 discuss the causes and consequences of altered fire regimes specifically across the terrestrial systems of the region, in which banner-tailed kangaroo rats live. Figure 14-3 presents the stressor model for the banner-tailed kangaroo rat in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by uncharacteristic wildfire. See Appendix 1 for a full presentation of the stressor model.

The historical importance of fire in the Chihuahuan Desert grasslands is not well understood. Some authors have stated that fire appeared to be unimportant in desert grassland vegetation before settlement (Dick-Peddie and Alberico 1977, McClaran and Anable 1992), while others suggest fire was historically prevalent in most grasslands and woodlands in the borderlands region (Kilgore et al. 2009). However, most believe some of the observed changes in vegetation are undoubtedly associated with many decades of fire suppression (McPherson and Weltzin 2000, Webster and Bahre 2001).

Starting in the late 1800's overgrazing eliminated many of the grasses and other fine fuels required to support grassland fires, resulting in fewer fires (Bahre 1991). Later, fire suppression and changes in precipitation patterns led to the spread of mesquite and other woody plants which resulted in a shrub-dominated community over time (Brown et al. 1997). Ultimately, fires became less frequent regardless of grazing pressure (Curtin et al. 2002, Drewa and Havstad 2001). After the introduction of Lehmann lovegrass, areas where Lehmann lovegrass has become established are at greater risk of fire because lovegrass generates greater quantities of fine fuels than do native grasses (Cox et al. 1990, McClaran and Anable 1992). Grazing can reduce these fuel loads and decrease the risks of fires (Holechek et al. 1994, Wright 1974).

Figure 14-3. Banner-tailed kangaroo rat stressor model: Potential impacts of uncharacteristic wildfire.



It is unclear whether fire has any substantial direct negative effects on the banner-tailed kangaroo rat. Valone et al. (2002) found no effect of fire on banner-tailed kangaroo rat populations. Later studies show that the mosaic of burned and unburned patches contributed to the lack of a fire effect on the rodent community (Hoffmeister 1986, Kilgore et al. 2009, McPherson 1995). Some of this desert mosaic is the result of the presence of banner-tailed kangaroo rat mounds and clearing around the mounds. In spite of the seeming lack of direct impacts of fire on the banner-tailed kangaroo rat, it is agreed that grazing, introduction of exotic plants, fire suppression and climate have all interacted to change the historical fire regime. These interactions with fire lead to the familiar theme with the banner-tailed kangaroo rat: fire itself may have minimal direct impacts on the banner-tailed kangaroo rat but as the fire regime changes banner-tailed kangaroo rat habitat and abundance declines.

Prescribed burning has been suggested as a method to prevent shrub encroachment on desert grasslands (McGlone and Huenneke 2004a). However, this may not be an effective tool. Fire-induced mortality of mesquite is low (Kilgore et al. 2009) and Kilgore et al. (2009) found that fire has an adverse effect on black grama grasslands. Geiger and McPherson (2005) found that the reintroduction of fire and removal of livestock did not lead to an increase in native species diversity or a decrease in non-native grasses or mesquite. The timing and intensity of precipitation makes predicting the post-fire response of desert plant communities difficult. Grasses show a strong positive response to fire when accompanied by relatively high soil moisture and neutral response, or short-term decline, during drought (Curtin et al. 2002). Kilgore et al. (2009) concluded that there are few significant positive changes in the native plant community after prescribed burning and suggest that the utility of prescribed fire as a positive management tool in the northern Chihuahuan Desert is not supported. In addition, suburban development has increased the liability on such efforts, where even a small number of homes make fire management effectively impossible at the landscape level (Curtin et al. 2002).

14.3.3 Invasive Species

Chapters 2-3 discuss the ecological consequences of non-native species across the U.S. portion of the ecoregion in general and Chapters 5-7 discuss the consequences of invasive species introductions specifically across the terrestrial ecological systems of the region in which the banner-tailed kangaroo rat lives. Figure 14-4 presents the stressor model for banner-tailed kangaroo rat in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by non-native species and their management. Appendix 1 presents the rationale and citations for each causal link shown in Figure 14-4.

There are numerous introduced plant species in the Chihuahuan Desert as well as a number of native woody species that have increased dramatically in abundance, potentially affecting banner-tailed kangaroo rat habitat. These introduced species can be defined in two broad categories: grass assemblage and woody vegetation assemblage. Some of the more notable non-woody species include Lehmann lovegrass (*Eragrostis lehmanniana*), African buffelgrass (*Pennisetum ciliare*) and red brome (*Bromus rubens*) (Van Devender et al. 2013). Within the woody vegetation assemblage, the desert shrubs that seem to have the most impact on the banner-tailed kangaroo rat populations include, but are not limited to, creosote bush (*Larrea tridentata*), mesquite (*Prosopis* spp.), broom snakeweed

(*Gutierrezia sarothrae*), tarbush (*Flourensia cernua*), and whitethorn (*Acacia neovernicosa*) (Anderson and Kay 1999, Krogh et al. 2002).

As mentioned earlier (see Overview, above, this chapter), the relative proportion of grass to woody vegetation affects banner-tailed kangaroo rat survival. The species does not occur in areas with woody vegetation cover above 20%. The later section on Landscape Restoration (below, this chapter) discusses this subject further.

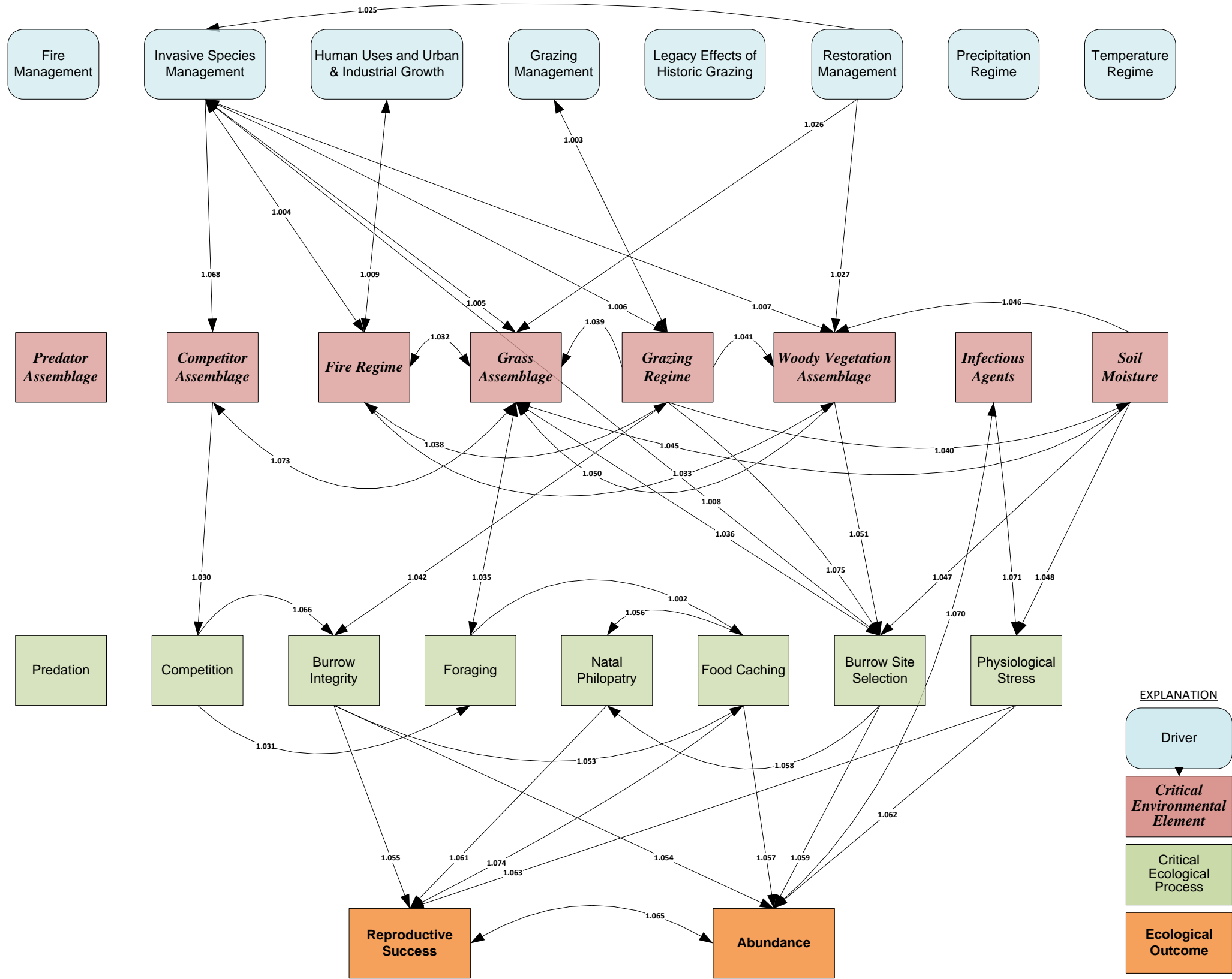
Banner-tailed kangaroo rats are typically found in desert grasslands historically dominated by black grama grass (*Bouteloua eriopoda*). In these grasslands, black grama cover can range from 44% in dry years to 75% in wet years (Paulsen and Ares 1962; Peters and Gibbens 2006). In 1858, good grass cover was present on more than 90% of the 58,492ha studied. By 1963, less than 25% of the area had good grass cover (Havstad and Beck 1996). One of the many factors leading to the decline of these grasslands was changes in the grass assemblage caused by the numerous exotic and invasive species that have adversely affected these black grama grasslands.

Lehmann lovegrass seems to be the most prevalent and most studied of these exotic species (McClaran and Anable 1992). As with many of the exotic grasses, Lehmann lovegrass was introduced and seeded from 1950-1980 to improve rangeland production and reduce erosion. While these exotic grasses may or may not have improved cattle forage, the lovegrass has reduced banner-tailed kangaroo rat burrow site selection options as it has displaced native grasses on large areas of the banner-tailed kangaroo rat range, especially where soils have been disturbed (Crawley and Gray 1987, Paulsen and Ares 1962). Bock et al. (1976) showed a serious decrease in most native plant species and native animals in the presence of Lehmann lovegrass. In addition, no banner-tailed kangaroo rats were found in the Lehmann lovegrass plots at the Jornada pasture site, but it was not clear if this was coincidence or a direct result of the Lehmann lovegrass (Hupy et al. 2004).

Lehmann lovegrass and other invasive species also likely affect the banner-tailed kangaroo rat indirectly, through their effects on the fire regime (Cable 1965, McClaran and Anable 1992, Brooks and Pyke 2001, Brooks et al. 2004, McGlone and Huenneke 2004a, McPherson 2006, Parmenter 2008, McDonald 2012, McGlone 2013, Ladwig 2014). Chapters 2-3 and 5-7 and the section on Uncharacteristic Wildfire, above, this chapter, provide detailed discussions of the impacts of fire on grass, shrub, and woody vegetation across the U.S. portion of the ecoregion.

Exotic species often flourish in disturbed areas. Human activities and grazing are often the primary sources for disturbance. Mack (1985) suggests that disturbance may be necessary only for initial establishment but not for subsequent spread of lovegrass. In established lovegrass populations, density tends to increase over time but does not seem to be affected by different grazing regimes. Grazing seems to disproportionately affect native grasses because native grass density decreases and lovegrass relative abundance increases with time and increased grazing intensity. This is most likely a result of preferential grazing by the cattle. Interestingly, lovegrass density and relative abundance did not differ between adjacent ungrazed and grazed areas in a study by McClaran and Anable (1992) and livestock grazing was not necessary for Lehmann lovegrass to spread.

Figure 14-4. Banner-tailed kangaroo rat stressor model: Potential impacts of invasive species and landscape restoration.



The effects on banner-tailed kangaroo rat abundance due to changes in the grass assemblage from native perennial grasses to exotic annual grasses are not well understood although it may be presumed that banner-tailed kangaroo rats favor native perennial grasses (Monson 1943). More research is needed on the effects of exotic species. One area of particular concern is foraging. In many cases the banner-tailed kangaroo rat may forage and use seeds from these exotic species as food, as up to 80 species of seeds have been found in banner-tailed kangaroo rat food caches (Hope and Parmenter 2007, Monson 1943). However, little is known about this. The impact of seeding of non-native grasses is unknown and may reduce food availability.

The most significant factor affecting the Chihuahuan grasslands and banner-tailed kangaroo rat abundance is the invasion of woody species into these grasslands. Many of these woody species are native species that have increased dramatically. Grasses have historically dominated the areas where banner-tailed kangaroo rats are found, even though shrubs and other woody plants may have been present – but in low densities (Johnston 1977, Van Auken 2000). During the past 150 years woody vegetation has substantially increased (Van Auken 2000). Grasslands that have suffered drought, livestock overgrazing and altered fire regimes have been invaded by woody vegetation (Gao and Reynolds 2003, Yanoff and Muldavin 2008). The once primarily dominant black grama grass is now on 1 % or less of the area, as found in one study at the Jornada Basin, and has been completely extirpated from many areas (Gibbens et al. 2005). One reason for the success of woody vegetation is its ability to outcompete grasses during periods of drought or extreme rain events (Gao and Reynolds 2003, Gibbens et al. 2005, Schlesinger et al. 1990). With increasing numbers of low-value shrubby plants on semi-desert grass-shrub ranges, the grasses inevitably decline and are unable to regain dominance of the site even when rainfall is not limiting (Paulsen and Ares 1962). Shrub encroachment into the desert grasslands of the Chihuahuan Desert is still occurring (Grover and Musick 1990) and shrub encroachment is recognized as the main feature of desertification in this region (Krogh et al. 2002).

Within this woody vegetation assemblage, the desert shrubs that seem to have the most impact on the banner-tailed kangaroo rat populations include, but are not limited to, creosote bush (*Larrea tridentata*), mesquite (*Prosopis* spp.), broom snakeweed (*Gutierrezia sarothrae*), tarbush (*Flourensia cernua*), and whitethorn (*Acacia neovernicosa*) (Anderson and Kay 1999, Krogh et al. 2002). The individual effects of each species on the banner-tailed kangaroo rat are not well known, and individually may or may not have an effect. For example, broom snakeweed may or may not depress banner-tailed kangaroo rat populations (Waser and Ayers 2003). However, as an aggregate, and in densities above 20%, shrubs seem to have a serious adverse effect on burrow site selection and banner-tailed kangaroo rat abundance as they completely displace banner-tailed kangaroo rat populations (Anderson and Kay 1999, Krogh et al. 2002, Moroka et al. 1982, Waser and Ayers 2003). Banner-tailed kangaroo rats are absent in most shrub filled areas and are eliminated when grassland degrades to mesquite coppice dune or eroded creosote bush communities (Whitford 1997). The best indicator of the presence or absence of the banner-tailed kangaroo rats is shrub cover and banner-tailed kangaroo rats do not occur where there is more than 20% shrub cover. Higher than 20% shrub cover results in local extinction (Krogh et al. 2002).

Furthermore, the cumulative effects of severe encroachment by woody species on an ecosystem can

drive the system into an alternative steady state that may be difficult or impossible to return to its historical community type (Schlesinger et al. 1990). These changes to the grassland communities have encouraged much effort be devoted to restoring these degraded grasslands. However, as with all invasive species, it is very hard to undo an introduction and subsequent invasion. The best approach going forward may be to contain the spread of the plants. Efforts to minimize soil disturbance from OHV's, cattle, and even cross-border traffic may mitigate the spread of some species. However, research on this is lacking. In addition, research on the effects of various grazing regimes as well as fire regimes may be warranted, as these factors may affect different species in different ways.

One additional area of research that may be warranted concerns invasive animal species such as new rodents or predators. The introduction of domesticated and feral cats and dogs associated with human activities are discussed in the section of human uses and urban and industrial growth. However, several rodents associated with human activity such as the black rat and the house mouse have been introduced into the Chihuahuan desert (Anderson 1972). Although they are mostly associated with human activities, they do enhance the chances for new infectious agent introductions and naturalization into the grasslands as well. Other rodents or snakes could also be accidentally introduced via escaped pets or other means, and these new animal introductions could change the competitor assemblage, increase competition, change the predator assemblage and/or increase predation, and may affect burrow integrity. In all cases these would have a negative influence on banner-tailed kangaroo rat reproductive success and banner-tailed kangaroo rat abundance. The impact of such an occurrence is hard to predict but warrants monitoring.

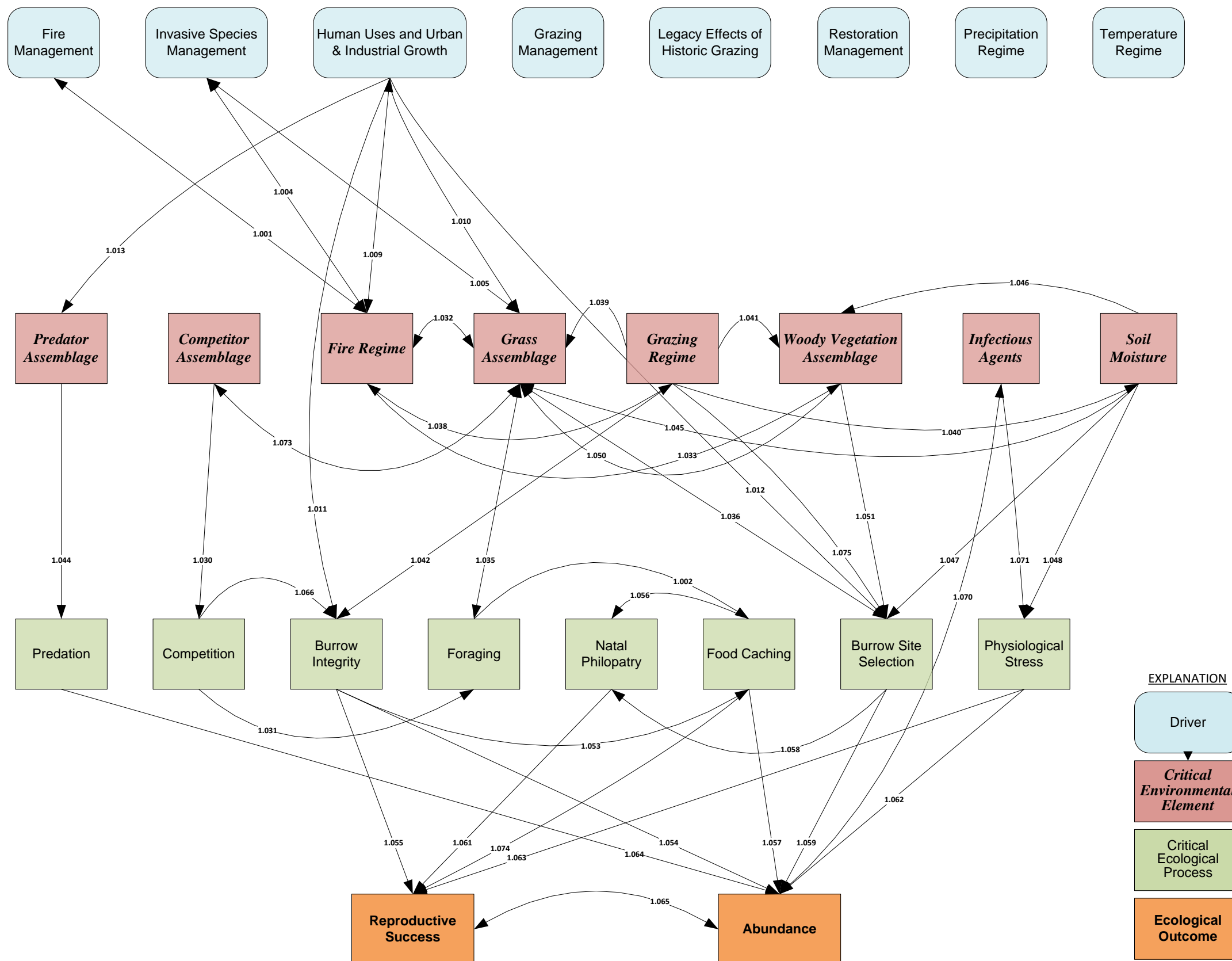
14.3.4 Development

Chapters 2 and 3 discuss the history and consequences of land development across the ecoregion in general, and Chapters 5-7 discuss these consequences specifically across the terrestrial ecological systems of the region, in which banner-tailed kangaroo rats live. Figure 14-5 presents the stressor model for the banner-tailed kangaroo rat in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by land development. See Appendix 1 for a full presentation of the stressor model.

Figure 14-5 identifies many different factors that typically lead to a loss of banner-tailed kangaroo rat habitat and/or abundance of the banner-tailed kangaroo rat. Land development, including expanding urban, suburban, industrial, energy-extraction, and agricultural developments, may have more severe and less repairable impacts on native species in the ecoregion than may other stressors (Havstad and Coffin Peters 1999). Although more limited spatially, and more recent than ranching, suburbanization is rapidly expanding and altering large areas in the Chihuahuan Desert (Archer and Predick 2008, Curtin et al. 2002). This suburbanization likely will continue to occur as land value for real estate development can be 4-100 times the value of ranching (Curtin et al. 2000) and development of grasslands can eliminate banner-tailed kangaroo rat habitat. Fortunately, a substantial portion of the U.S. portion of the ecoregion is owned by federal and state governments (see Chapters 2-3), and therefore experience lower direct threats from suburban development. However, other threats such as solar arrays, oil and gas developments, and mining can affect banner-tailed kangaroo rat habitat even on federal lands

(Whitford and Bixby 2006). Increased human presence brings changes to the fire regime. Increased human activity may lead to increased sources of ignition for wildfires (Hemstrom 2014). Further, it takes only a small number of exurban homes to render fire management effectively impossible at the landscape level (Curtin et al. 2002).

Figure 14-5. Banner-tailed kangaroo rat stressor submodel for human uses and urban & industrial growth.



Land development inevitably leads to increased human presence and human activities on lands surrounding developed areas. Maintenance of roads and rights-of-way as well as recreational OHV use will have the potential to seriously degrade banner-tailed kangaroo rat burrows and destroy food caches, thereby reducing reproductive success as well as killing banner-tailed kangaroo rats inside their mounds when they collapse. In the past few decades, increased immigration and border patrol activities along the Mexico-United States border have increased foot and vehicle traffic. Increased foot and vehicle traffic can negatively impact the soil and can lead to an increase in woody vegetation and other invasive plants via soil compaction and damage (Whitford and Bixby 2006). No studies have been conducted on the effect of these activities on banner-tailed kangaroo rats but it can be assumed there has been an increase in burrow destruction. More people living in close proximity to banner-tailed kangaroo rat habitat also means the addition of domesticated dogs and cats into the predator assemblage leading to increased predation and a subsequent drop in banner-tailed kangaroo rat abundance although this needs more research specific to the Chihuahuan desert.

The loss of habitat over time poses significant challenges for the banner-tailed kangaroo rat. Prime habitat does seem to be somewhat scarce, as there is competition for mounds and this competition has led to natal philopatry in this species (Jones 1984). Fortunately, banner-tailed kangaroo rats do not require large areas to maintain populations and can survive even on small patches of good habitat. However, as development occurs, roads and utility rights-of-way can fragment the landscape causing permanent changes to ecosystem structure, eventually resulting in a loss or reduction in abundance of many of the grassland animals (Whitford and Bixby 2006). The need for connectivity of banner-tailed kangaroo rats to other patches and the effect of this fragmentation is not well understood. Skyvarla et al. (2004) demonstrated that this species has a rather small home range and Cosentino et al. (2014) found that recolonization rates are very slow and they hypothesized it was due to a limited ability to disperse lack of connectivity to other banner-tailed kangaroo rat populations.

It is very difficult to undo suburbanization, agricultural development or other human-induced land conversion. Reichardt (1982) found very limited establishment of native perennial grasses in Colorado fields cultivated 40 years earlier. In the Sonoran Desert, Jackson et al. (1991) found virtually no plant cover on some fields even 25 years after abandonment. In most cases, prevention or management of development is the best strategy.

Land conversion is often thought of only as a negative process. One possible land use that may help the banner-tailed kangaroo rat maintain its abundance is the use of grassbanking agreements or similar conservation easements on large tracts of privately held land (Curtin et al. 2002). These “legal paper” land conversions, if designed correctly, may be able to maintain, or at least slow, the destruction of banner-tailed kangaroo rat habitat.

14.3.5 Excessive Domestic Grazing

Chapters 2-3 discuss the ecological consequences of excessive domestic grazing across the U.S. portion of the ecoregion in general and Chapters 5-7 discuss the history and consequences of livestock grazing specifically across the terrestrial ecological systems of the region in which banner-tailed kangaroo rats live. Figure 14-6 presents the stressor model for the banner-tailed kangaroo rat in the U.S. portion of the

ecoregion, simplified to show only those causal relationships affected by non-native species and their management. Appendix 1 presents the rationale and citations for each causal link shown in Figure 14-6.

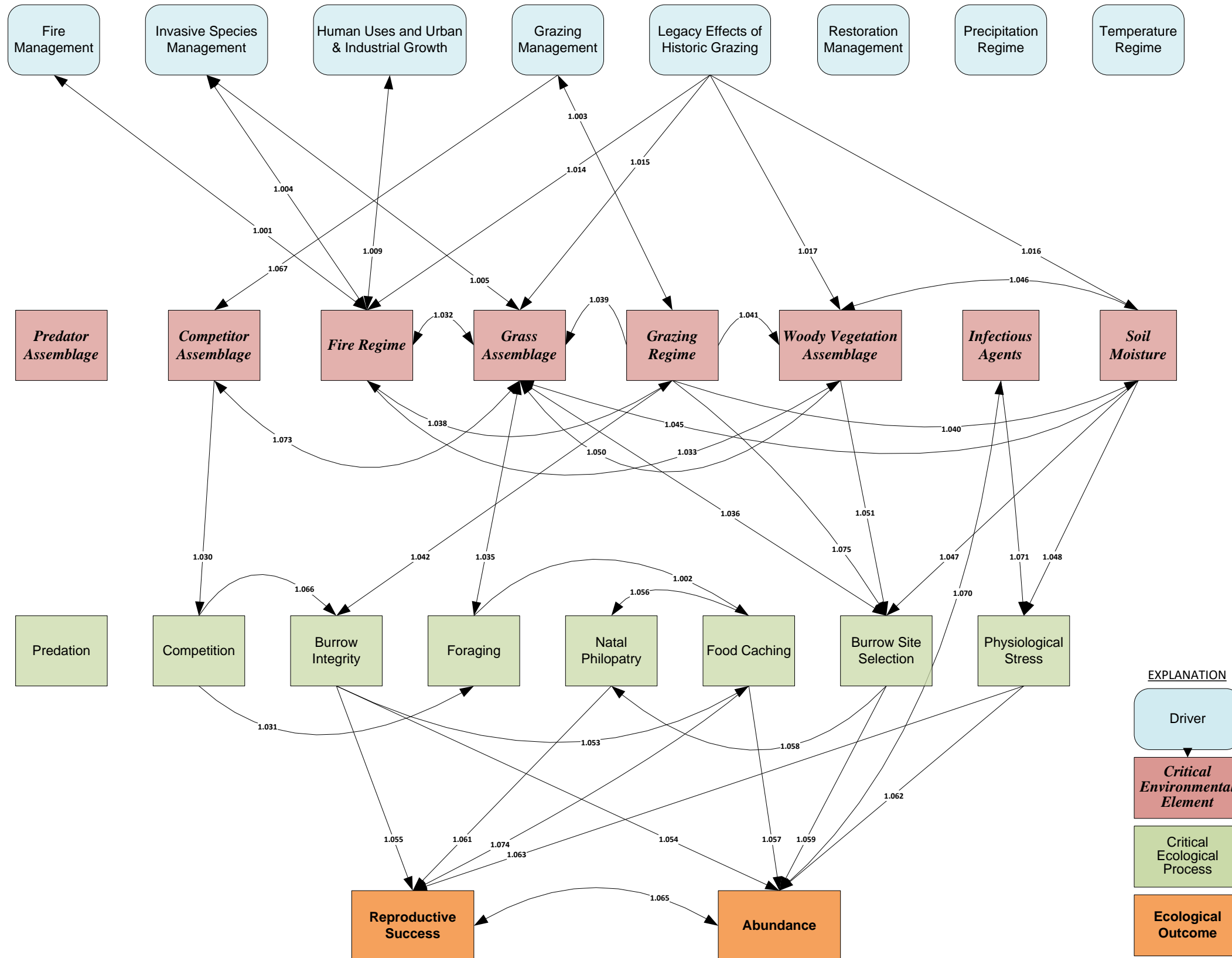
Many of the environmental impacts of excessive grazing occurred shortly after Euro-American settlement, aggravated by climate changes at the same time (Curtin et al. 2002, McPherson and Weltzin 2000) (see Chapters 2-3 and 5-7 for detailed discussions). One major manifestation of the damage caused by historic excessive grazing is in the desertification of the grasslands and associated invasion of woody plants (Bahre and Shelton 1993). This desertification then caused a cascade of interactive effects, including changing the fire regime, grass assemblage, woody vegetation assemblage, soil characteristics, and soil moisture holding abilities (Bahre and Shelton 1993). Overall, this overgrazing to the point of desertification has caused large-scale declines in banner-tailed kangaroo rat abundance (Krogh et al. 2002, Waser and Ayers 2003).

Modern grazing is managed based on forage use, qualitative categories (light, moderate, severe) and quantitative measures of vegetation abundance (Holechek and Galt 2000). While many advocate for removal of cattle from the grasslands, some authors think that reducing or eliminating livestock now will not cause the land to heal (Curtin et al. 2002). Excluding livestock grazing from most sites now will have little to no impact on the abundance of woody plants or non-native herbs during the next several decades (McPherson and Weltzin 2000). Curtin et al. (2002) feel that climate, substrate, evolutionary history, and other disturbance factors are often more important in determining current vegetation response than the number or presence of livestock. This is not to suggest that grazing is benign with respect to these grasslands.

Jones (2000) revealed that grazing is generally unfavorable for rodent communities in arid grasslands. Mata-González et al. (2007) found that in the absence of grazing, there was 50% higher grass cover and 35% higher total biomass. Heske and Campbell (1991) also found that cattle grazing had a modest negative effect on banner-tailed kangaroo rats. Livestock can negatively impact rodent populations by trampling burrows and compacting soil or harvesting seed heads and other plant parts while grazing, thus removing resources that otherwise would be available to rodents (Heske and Campbell 1991). In addition, Best (1972) has shown that mound formation, relocation, entrances and caches of food are affected by trampling by cattle. Trampling may also result in a reduction of moisture in the upper layers of soil, thus favoring the deep-rooted mesquite (Bahre and Shelton 1993). However, we know of no quantitative studies assessing the frequency of trampling or the amount of harm caused by trampling.

More complex interactive effects of grazing, exotic species, and fire are still ongoing. Historical efforts to improve forage or increase access to grazing lands included the introduction of exotic plants, like Lehmann lovegrass. As discussed elsewhere in this document, this grass seems to out-compete and displace the native black grama grass. In addition, it has the potential to change the fire regimes (Brooks et al. 2004; McClaran and Anable 1992). Cattle may also eat seeds of some shrubs, particularly mesquite, and spread them over a much wider area. Furthermore, livestock grazing has reduced grass fuel, which has lessened the ability of the rangelands to carry fire and has lowered fire temperatures and, hence, the ability of fires to kill mesquite (Bahre and Shelton 1993, Van Auken 2000). The cumulative effects of the grazing x fire x invasive species interactions result in a more shrub-dominated landscape and reduced banner-tailed kangaroo rat habitat.

Figure 14-6. Banner-tailed kangaroo rat stressor submodel for legacy effects of historic grazing and grazing management.



Grazing management can mitigate harm done to banner-tailed kangaroo rats by cattle and sheep, and while we may not be able to return to pre-grazing levels of grass, we can maintain the grasslands we do have. Low intensity grazing regimes may be sustainable, depending on precipitation. Grasses and forbs need time to recover from grazing. Range condition increased in years with sufficient precipitation and decreased during drought (Molinar et al. 2011, Navarro et al. 2002). In addition, adequate monitoring to assess burrow trampling, competition for perennial grasses, soil compaction, and shrub encroachment can guide land managers to mitigate loss of banner-tailed kangaroo rat habitat.

Figure 14-6 illustrates how the legacy effects of historic grazing and grazing management can affect banner-tailed kangaroo rat abundance. Grazing has been divided into two different drivers in order to separate the legacy effects of historic grazing and current grazing management. As can be seen in Figure 14-6, the legacy effects of historic grazing have strong links to the grass assemblage, woody vegetation assemblage, and soil moisture, all of which affect banner-tailed kangaroo rat burrow site selection and thus banner-tailed kangaroo rat abundance. As discussed previously, much of the degradation of these grasslands are a result of historic grazing affecting these three critical environmental elements. We also see continued influence of the legacy effects of historic grazing on the fire regime, which in turn affects current fire management.

Current grazing management links with the grazing regime. Land managers responsible for setting grazing regulations need to consider grazing regime links with burrow integrity, banner-tailed kangaroo rat burrow site selection, fire regime, soil moisture, grass assemblage, and woody vegetation assemblage. In addition, the cattle are part of the banner-tailed kangaroo rat competitor assemblage and compete in foraging for grass seeds. This complex mix of critical environmental elements, activities, and processes will require continuous monitoring and adjustment to balance the economics of grazing with assuring that banner-tailed kangaroo rats will have opportunities for burrow site selection and enhancing banner-tailed kangaroo rat abundance.

14.3.6 Landscape Restoration

Chapters 3 and 5-10 discusses the types of landscape restoration projects taking place in the U.S. portion of the ecoregion, some of which currently or in the future could affect banner-tailed kangaroo rat habitat; and the banner-tailed kangaroo rat itself is managed as a species of conservation concern in the U.S. portion of the ecoregion. Figure 14-4 presents the stressor model for the banner-tailed kangaroo rat in the U.S. portion of the ecoregion, simplified to show only those causal relationships that address invasive species, their management, and landscape restoration. Figure 14-4 addresses these two Change Agents together because they are so closely linked in their impacts on the banner-tailed kangaroo rat in the U.S. portion of the ecoregion. Appendix 1 presents the rationale and citations for each causal link shown in Figure 14-4.

Invasive species management requires consideration of both the fire and grazing regimes. As a result, four drivers in the banner-tailed kangaroo rat stressor model – and invasive species management, fire management, grazing management, and restoration management – are often interrelated, with a common goal of improving grassland habitat. Landscape restoration treatments have been applied to >200,000 ha of shrubland in the U.S. portion of the ecoregion Desert (Cosentino et al. 2014). Restoration

often really means revegetation and these revegetation projects typically focus on reestablishing grass for soil conservation and livestock forage rather than restoring banner-tailed kangaroo rat habitat. Fortunately, efforts to improve forage for cattle and reduce woody vegetation may be beneficial to banner-tailed kangaroo rat populations as well.

Restoration efforts that focus on returning shrub-invaded habitat back to grassland habitat may use fire, chaining, and herbicide to push the community succession. Treatments applied at intense levels severe enough to kill shrub roots may transition a shrub-dominated state back to a grassland state – although the grasses may not all be native grasses (Hemstrom 2014). Less severe treatments do not kill shrub roots and only set the plant community back to a grassland state with live shrub roots remaining that can resprout (Hemstrom 2014). This leads to a faster return of shrub cover and need for more frequent treatments. In many cases, the native grasses have not established as well as non-native grasses and the application of treatments sometimes disturbs the soil and inadvertently increases non-native or woody species (Hemstrom 2014, Roundy and Biedenbender 1995). It is not well understood how well banner-tailed kangaroo rats are able to use many of the exotic grasses as forage.

Attempts to improve plant community composition have not always been successful. External factors such as precipitation amounts and precipitation phenology may have more to do with the increase or decrease in densities of particular species of invasive shrubs than do any restoration activities (Roundy and Biedenbender 1995). Shrub-dominated ecosystems resist attempts to restore them back to grasslands (Kerley and Whitford 2000). Ultimately, the complexity and cost of landscape-level restoration may limit true grassland restoration to small areas that mainly serve educational and research purposes (Fitzgerald et al. 2001, Roundy and Biedenbender 1995). These small-scale efforts may not be substantial enough to improve banner-tailed kangaroo rat abundance in any significant way. Eve et al. (1999) suggests that the probability of grassland restoration in present-day monoculture creosote bush ecosystems or mesquite coppice dune ecosystems is virtually zero. Cosentino et al. (2014) also suggest that, managers selecting areas for restoration should adopt a landscape mosaic approach and try to maximize connectivity, because banner-tailed kangaroo rat dispersers prefer to renovate vacant mounds rather than construct new mounds (Best 1972, Jones 1984, Waser et al. 2006). This indicates that emigration to restoration areas is likely limited by the spatial distribution of vacant mounds.

As stated previously, banner-tailed kangaroo rats are ecological engineers that dramatically affect the community structure of both plants and animals (Curtin et al. 2000, Davidson et al. 2008, Davidson and Lightfoot 2007, Moorhead et al. 1988, Moroka et al. 1982) and are considered a keystone species (Brown and Heske 1990, Kerley et al. 1997). Banner-tailed kangaroo rat mounds increase soil heterogeneity, provide novel microhabitats for a number of rare plants and animals (Ayarbe and Kieft 2000, Guo 1996, Moorhead et al. 1988, Whitford 1997), and create expansive burrow networks that are used by many other organisms for shelter (Davidson et al. 2008, Hawkins 1996, Hawkins and Nicoletto 1992).

Brown and Heske (1990) found that, when kangaroo rats are excluded from grasslands, the densities of both tall perennial grasses and annual grasses increase approximately threefold and the composition of

the small rodent composition shifts. They attribute much of this response to two species: Lehmann lovegrass, which increased more than 20-fold, and the annual, *Aristida adscensionis*, which increased approximately threefold (Brown and Heske 1990). These responses may be consequences of increased availability of seeds, reduced competition, decreased soil disturbance, or selective seed predation (Brown and Heske 1990). When only the banner-tailed kangaroo rat is removed, the densities of the two congeneric species, *Dipodomys merriami* and *Dipodomys ordii*, increased and there were no significant changes in the densities of other rodents (Brown and Munger 1985).

These study findings suggest that if the exclusion of banner-tailed kangaroo rats results in dramatic changes in the grasslands, then reintroduction of banner-tailed kangaroo rats may also have dramatic effects on the grasslands. Reintroduced banner-tailed kangaroo rats may help maintain a shrub-free open grassland. Unfortunately, no research has been conducted on this specific question. Cosentino et al. (2014) investigated restoring populations of banner-tailed kangaroo rats into areas that have undergone habitat restoration—in this case shrub removal. This only answers part of the management question. Cosentino et al. (2014) did not investigate physically reintroducing banner-tailed kangaroo rats. Such reintroduction would face the same issue of limited mounds site as discussed above. However, Cosentino et al. (2013) did find that that lizard community composition was sensitive to the density of banner-tailed kangaroo rat—particularly on areas treated with herbicide—and that there was a strong, positive effect of banner-tailed kangaroo rats density on the abundance of the desert grassland whiptail, *Aspidoscelis uniparens*, in particular. As mentioned previously, kangaroo rats may provide needed soil heterogeneity and novel microhabitats for rare plants and animals and may create the expansive burrow networks used by many other organisms for shelter. Cosentino et al. (2013) felt their results indicate that burrowing rodents may play a key role in controlling the response of wildlife species to habitat restoration, most likely through their engineering effects. However, it may take many decades to see any substantive effects (Cosentino et al. 2014; 2013).

14.4 Banner-tailed Kangaroo Rat Key Ecological Attributes

All ecological outcomes and critical ecological processes in the banner-tailed kangaroo rat stressor model constitute the key ecological attributes for the CE. The list below identifies ten key ecological attributes for the banner-tailed kangaroo rat based on these criteria. *Fully characterizing the present condition of the CE will require data on indicators for its key ecological attributes.* Indicators are determined during Phase II of the REA process. The definitions for the key ecological attributes are the same as the definitions for these model components presented above.

- **Ecological Outcomes**
 - Abundance
 - Reproductive Success
- **Critical Ecological Processes**
 - Burrow Integrity
 - Burrow Site Selection
 - Competition
 - Food Caching
 - Foraging
 - Natal Philopatry
 - Physiological Stress
 - Predation

15 Black-tailed Prairie Dog Conceptual Model

This chapter presents the conceptual ecological model for the black-tailed prairie dog (*Cynomys ludovicianus*), a colonial, burrowing rodent that inhabits several types of open grassland habitats from the Great Plains south through the deserts of northern Mexico. The presentation of the banner-tailed kangaroo rat conceptual model follows the structure described in Chapter 4, with sections on sources of information, a species overview, the stressor model, and key ecological attributes. As noted in Chapter 4, the conceptual models for species and species assemblages do not include a separate control model.

15.1 Sources of Information

The black-tailed prairie dog overview and stressor model integrate information from several sources, including Garrett and Franklin (1988), Miller et al. (1994), Kotliar et al. (1999), Basurto and Hadley (2006), Milne-Laux and Sweitzer (2006), Magle and Crooks (2009), and Avila-Flores et al. (2010).

15.2 Black-tailed Prairie Dog Overview

The Chihuahuan desert is an important region for conservation of black-tailed prairie dog. The ecoregion contains the largest intact colony complex of the species, the Janos-Casas Grande Complex of over 20,000 hectares, located in Janos, Mexico, just south of the border with the United States (Basurto and Hadley 2006).

15.2.1 Distribution

The area covered by black-tailed prairie dog colonies in 1900 was estimated at 40 to 100 million hectares but declined to about 600,000 hectares by 1960 (Miller et al. 1994). Black-tailed prairie dogs are ecological engineers that alter the structure and composition of the grasslands they occupy and are considered a keystone species in grassland systems (Kotliar et al. 1999). Many species are dependent on black-tailed prairie dog grazing to maintain suitable habitat and multiple predator species directly prey on the rodents. Predators include several species of high conservation concern, including ferruginous hawk (*Buteo regalis*), swift fox (*Vulpes velox*), mountain plover (*Charadrius montanus*), and the federally endangered black-footed ferret (*Mustela nigripes*), the last of which is an obligate predator of black-tailed prairie dog (and two other species of prairie dogs in the western United States).

Black-tailed prairie dog have been extensively killed by shooting, trapping, and poisoning since the arrival of European settlers, largely because of their perceived conflict with cattle and horse grazing. The colonial behavior and patchy distribution of black-tailed prairie dog facilitate eradication: Estimates suggest that more than 90% of the historic population of the species has been lost, with intentional extermination programs considered to be the largest driver of this decline (Miller et al. 1994). Legal protections are now in place to limit or halt eradication efforts in parts of its range, but public eradication programs have continued and conservation initiatives have struggled to strike a balance with agricultural interests (Basurto and Hadley 2006). Aside from direct mortality, hunting and poisoning can depress the ability of colonies to recover from other threats and increase inter-colony distance when entire colonies are eradicated.

15.2.2 Habitat

The black-tailed prairie dog's large historical range encompasses two major habitats: shortgrass and tallgrass prairie in the cooler, wetter, more northerly portions of its range and desert grasslands in the hotter, drier, more southerly portions of its range. This report focuses on black-tailed prairie dog's usage of desert grassland in the Chihuahuan Desert. Avila-Flores et al. (2010) examined a variety of habitat variables and how they correlated with the siting of black-tailed prairie dog colonies in the Chihuahuan Desert. Colony siting was positively correlated with soil moisture level, local percent forb cover, local percent cover of unpalatable vegetation, local percent cover of bare ground, and the amount of other colonies within 1 km. The importance of sufficient local forbs supports the hypothesis that the abundance of black-tailed prairie dog in desert grasslands is primarily limited by food availability (Avila-Flores et al. 2010). In shortgrass prairie systems, the black-tailed prairie dog typically occurs in areas with vegetation cover between 25-90% and do not occur in areas with less than 15% percent cover of vegetation. In contrast, none of the desert grassland sites examined by Avila-Flores et al. (2010) had percent cover of vegetation higher than 15%. The positive correlation between colony siting and both bare ground and unpalatable vegetation may be a by-product of colony presence, rather than a habitat preference of black-tailed prairie dog. Foraging activities and the construction of burrows likely increases the amount of bare ground in a colony, while selective foraging and the ability of unpalatable and invasive plants to colonize areas disturbed by digging may influence the local cover of unpalatable vegetation within colonies. Colony siting was negatively correlated with altitude, local percent shrub cover, local average vegetation height, and the amount of hostile habitat within 1 km. The two largest correlations were shrub cover and vegetation height. While both of these variables may be influenced by active clipping by black-tailed prairie dog within and around their colonies, their significance to colony siting nonetheless underscores the importance of visibility for the species to detect predators. Topography and elevation appear to be important, as most colonies were located at elevations below 1,600 meters, even when seemingly adequate habitat exists at higher elevations.

Avila-Flores et al. (2010) examined several other variables and found no others with strong correlation to colony siting, including the presence of humans and soil type. While colonies located within and near human development face unique threats from this proximity, development itself does not seem to preclude black-tailed prairie dog from utilizing otherwise adequate habitat. Unlike colonies in shortgrass prairie, where soil type strongly influences colony siting, black-tailed prairie dog appear to use all soil types in the Chihuahuan Desert, proportional to their availability, but may avoid the rockiest areas (Avila-Flores et al. 2010). Hostile habitat includes any landcover type that cannot be colonized by black-tailed prairie dog and effectively acts as a barrier to their movement and dispersal. Experiments have shown that vegetation height exceeding 40 cm prevents most or all movement by black-tailed prairie dog (Milne-Laux and Sweitzer 2006, Avila-Flores et al. 2010). The mobility and dispersal ability of black-tailed prairie dog is important for establishing new colonies and recolonizing areas after local extinction events. Habitats may remain unutilized if the distance between colonies is too large, as the maximum dispersal distance for individual animals is estimated to be 8 km (Garrett and Franklin 1988). Habitat connectivity is especially important in urban areas with considerable barriers to black-tailed prairie dog movement (Magle and Crooks 2009). For these reasons, metapopulation structure is considered a critical component of black-tailed prairie dog colony persistence.

Predators of black-tailed prairie dog include coyote (*Canis latrans*), golden eagle (*Aquila chrysaetos*), ferruginous hawk (*Buteo regalis*), and black-footed ferret (*Mustela nigripes*) (Basurto and Hadley 2006). Almost no study has been directed at how variation in predator density and rates of predation on black-tailed prairie dog may be affecting this species in the Chihuahuan Desert. Understanding how these interactions may be changing remains a key data gap in the management of black-tailed prairie dog.

15.3 Black-tailed Prairie Dog Stressor Model

Table 15-1 presents and defines the drivers, critical environmental elements, critical ecological processes, and ecological outcomes represented in the black-tailed prairie dog stressor model. The stressor model follows the methodology for species described in Chapter 4.

Table 15-1. Black-tailed prairie dog definitions of stressor model components.

Model Component	Definition
Drivers	
Fire Management	Refers to any human activities to control the size, timing, intensity, or frequency of fires. Fire management applies to the control and prevention of wildfire as well as the management of prescribed burns.
Grazing Management	Refers to human activities to manage and control where and how domestic cattle are grazed within the Chihuahuan Desert. This includes grazing intensity as well as any grazing rotation strategies.
Hunting/Poisoning	Refers to human activities that intentionally cause mortality in black-tailed prairie dogs. This includes shooting, poisoning, and the use of lethal traps.
Land Conversion	Refers to any human activity that converts native habitats (shrubland or grassland) into developed areas. Developed areas include urban, industrial, and suburban development as well as agricultural croplands. Croplands do not provide significant habitat for black-tailed prairie dogs.
Precipitation Regime	Refers to the pattern, timing, intensity, and average annual rate of precipitation in the Chihuahuan Desert.
Temperature Regime	Refers to the patterns of air temperature in the Chihuahuan Desert across all seasons.
Urban Threats	Refers to a myriad of potential threats facing black-tailed prairie dogs that live in close proximity to human development. These include collisions with motor vehicles, flooding from street run-off, exposure to environmental toxins, injury from cats and dogs, physical barriers to movement, and disturbance from the presence of humans.
Critical Environmental Elements	
Bare Ground	Refers to the average percent cover of bare ground within a specific region.
Distance between Colonies	Refers to the average distance separating discrete colonies of black-tailed prairie dogs.
Fire Frequency/Intensity	Refers to the specific intensity of burns (characterized as low, medium, or high intensity) and the frequency with which these burns occur in a specific area. This element can be applied to both wildfire and prescribed burns.
Forb Cover	Refers to the average percent cover of forbs within black-tailed prairie dog colonies.
Grazing Intensity	Refers to the amount and frequency of plant material removal compared with the plant material remaining for use (e.g., habitat, food, cover) by black-tailed prairie dogs. In the literature, intensity is generally characterized as light, moderate, or heavy.
Hostile Habitat	Refers to the extent of hostile habitat surrounding black-tailed prairie dog colonies. Hostile habitat refers to any land cover that inhibits the movement of BTPD. Examples of hostile habitat include areas with extensive shrub cover, water features, or fences.

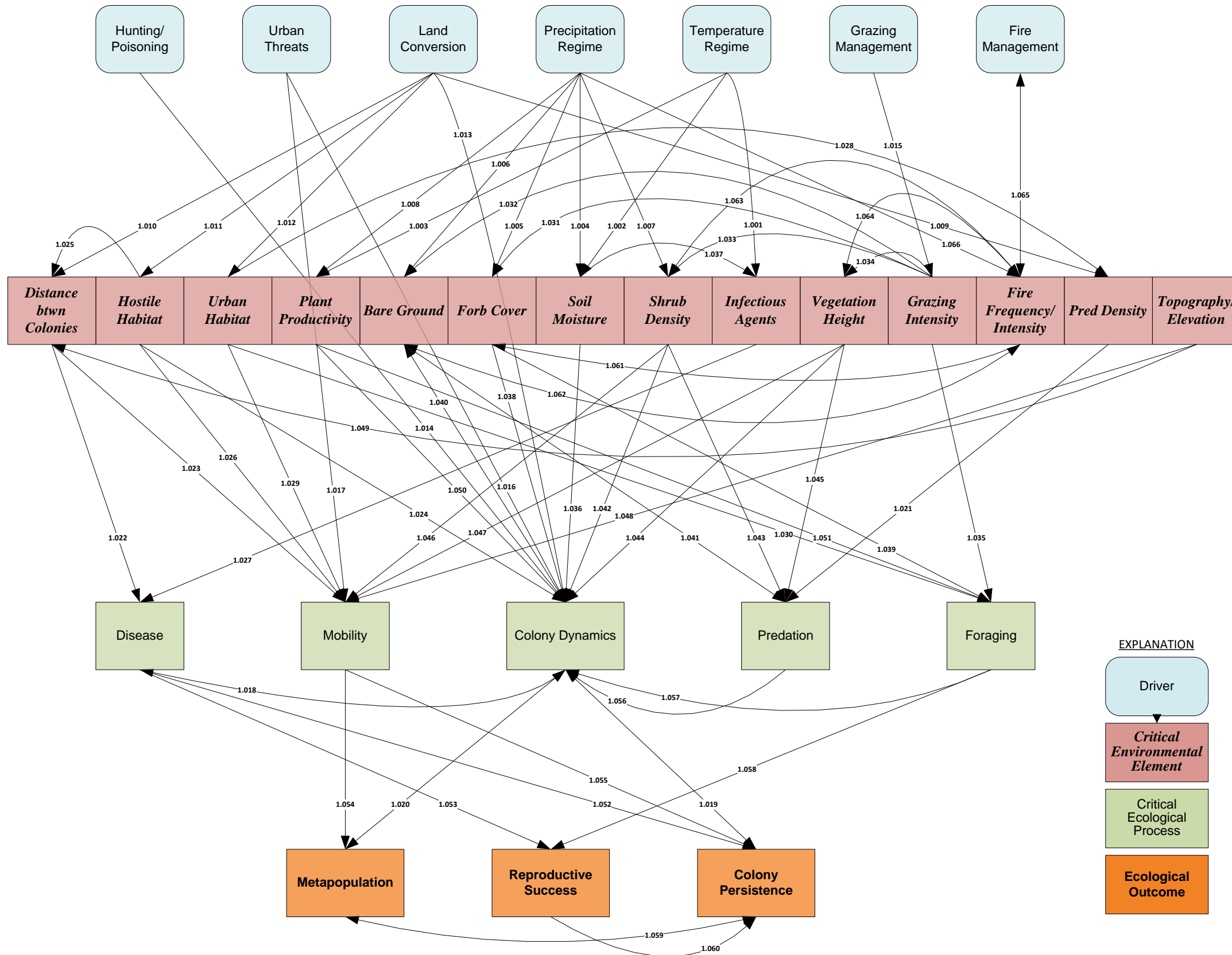
Model Component	Definition
Infectious Agents	Refers to the presence and abundance of infectious agents that can cause disease and mortality in black-tailed prairie dog. One of the most important infectious agents is plague (<i>Yersinia pestis</i>).
Plant Productivity	Refers to the amount of growth and seed production of plants in a given area.
Predator Density	Refers to the density and type of predators that prey on black-tailed prairie dogs.
Shrub Density	Refers to the density of woody shrub species within a given area.
Soil Moisture	Refers to the average soil moisture in an area across all seasons.
Topography/Elevation	Refers to landscape features including rock formations, cliffs, and slope, as well as the elevation of specific areas.
Urban Habitat	Refers to the extent human development in a given area.
Vegetation Height	Refers to the average height of all types of vegetation, including shrubs, forbs, and grasses, in a given area.
Critical Ecological Processes	
Colony Dynamics	Refers to number of black-tailed prairie dogs in each colony, the rates these sizes are changing, and the rate of formation and abandonment of colonies.
Disease	Refers to any condition affecting the health of individual black-tailed prairie dog, including internal and external parasites, bacterial, fungal and viral infections, and environmentally-based toxicity.
Foraging	Refers to the ability of black-tailed prairie dog to actively find and consume food resources within their habitats
Mobility	Refers to the ability of black-tailed prairie dogs to physically disperse beyond the boundary of their colony, either to reach other colonies or to colonize unoccupied habitat.
Predation	Refers to mortality that black-tailed prairie dog face from predators.
Ecological Outcomes	
Colony Persistence	Refers to the ability of individual colonies to persist across time.
Metapopulation	Refers to the structure of metapopulations of black-tailed prairie dog colonies in a given area. Metapopulation structure includes the number of colonies, the individual size of colonies, the distances between colonies, and the degree of mobility and interaction between the colonies.
Reproductive Success	Refers to the rate that black-tailed prairie dog are successfully able to birth and raise young to self-sufficiency.

Figure 15-1 shows the full black-tailed prairie dog stressor model, displaying the model components listed in Table 15-1 along with their causal relationships. Appendix 1 describes and presents the rationale for including every causal relationship in the stressor model, with citations for each causal relationship, providing a comprehensive presentation of the stressor model. A causal relationship exists when a change in one component of the system results in a change in some other component. Change in the first component is said to “cause a change in the second component. Each chain of causation, from driver to outcome, describes how the condition of the system likely has changed in the past, or likely would change in the future, in response to changes in its drivers.

Figure 15-1 indicates the presence or absence of causal relationships between the system model components but does not indicate the potential magnitude or other characteristics of these relationships, as explained in Chapter 4. Figure 15-1 identifies the causal relationships that have affected how the system likely has changed in the past, in response to changes in its drivers. The diagram provides a means for articulating how the condition of the system will likely change in response to

changes in its drivers. As discussed and illustrated below, the stressor model makes it clear: (a) which critical environmental elements would likely be affected by a change in one or more particular drivers, including change agents; (b) which critical ecological processes would likely be affected by the cascading effects of these changes in critical environmental elements; and (c) which system characteristics (ecological outcomes) would likely be affected by the cascading effects of these changes in critical environmental elements and ecological processes. By doing so, in turn, the stressor model also highlights those components of the model—drivers, critical environmental elements, critical ecological processes, and ecological outcomes—that demand indicator data.

Figure 15-1. Black-tailed prairie dog stressor model.



The following paragraphs discuss how each Change Agent affects black-tailed prairie dog in the U.S. portion of the ecoregion, as articulated in the stressor model for this CE. A “sub-model” diagram for each Change Agent presents a simplified version of the master stressor model diagram, showing only the direct and indirect effects of the Change Agent of interest. The text accompanying each sub-model diagram summarizes information from Appendix 1 concerning the impacts of each Change Agent, including selected citations. For each Change Agent, the paragraphs summarize present understanding of: (a) which critical environmental elements would likely be affected—directly or indirectly—by each Change Agent and (b) which critical ecological processes would likely be affected—directly or indirectly—by the cascading effects of these changes in critical environmental elements.

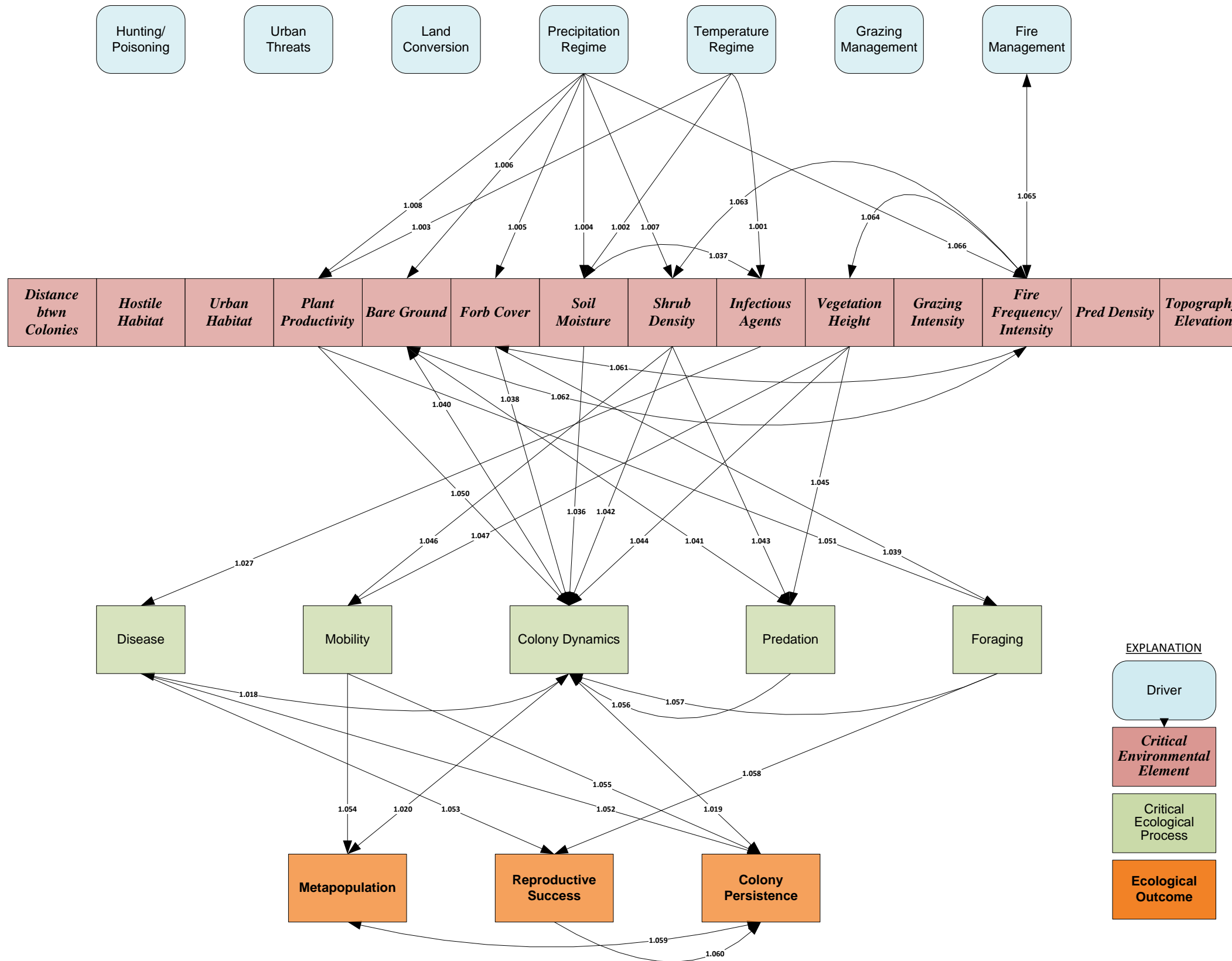
15.3.1 Climate Change and Uncharacteristic Wildfire

The final report for the Chihuahuan Desert REA will include a discussion of current forecasts of the ways in which climate change will affect the ecoregion and its ecological resources. The present chapter presents only the conceptual model of the causal relationships and outcomes that potentially will be affected for the black-tailed prairie dog. Figure 15-2 presents the black-tailed prairie dog stressor model for the U.S. portion of the ecoregion, simplified to show only those causal relationships that potentially will be affected by changes in the air temperature and precipitation regimes. Appendix 1 presents the rationale and citations for every causal link shown in the diagram.

The climate of the Chihuahuan Desert is expected to become more arid in the coming decades with conditions similar to past droughts (e.g. the Dust Bowl, the 1950s, and the current drought in the region) becoming the baseline climate for the region (Backlund et al. 2008, Seager et al. 2007). Drought conditions have been shown to favor shrub species over grasses, likely due to the deeper root systems of the former (White et al. 2011, Backlund et al. 2008). Decreased precipitation is also linked to lower plant productivity in all habitat types, which in turn lowers food resource availability for animals in these habitats, including black-tailed prairie dog. Lower productivity may also lead to increased grazing intensity from domestic cattle, because both more extensive and intensive grazing is required to support existing herds, which may increase the competition between cattle and black-tailed prairie dog for forage resources and the potential for conflict with humans. Plant productivity is especially important to black-tailed prairie dog in the Chihuahuan Desert, where it is believed to be the primary limiting factor to their abundance (Avila-Flores et al. 2010). Drought-caused mortality in grasses is also linked to shrub encroachment into grasslands and the overall decline of habitats dominated by grasses. Facka et al. (2010) found that all black-tailed prairie dog colonies they studied in the Chihuahuan Desert declined by at least 68% during drought years and reproductive success during droughts was near 0% for most colonies. If predictions of sustained drought conditions in the region are accurate, this new climate will likely both increase the rate of loss of grassland habitat and decrease the amount forage resources and thereby lower the region’s carrying capacity for black-tailed prairie dog.

Climate change could also affect the incidence of sylvatic plague among black-tailed prairie dog colonies in the ecoregion. The next section of this chapter, on Invasive Species, discusses this possibility.

Figure 15-2. Black-tailed prairie dog stressor model: Potential impacts of climate change and uncharacteristic wildfire.



Changes to the fire regime of the Chihuahuan Desert caused by either human fire management or shifts to the region's climate have the potential to affect the habitat of black-tailed prairie dog. This interaction has been studied in shortgrass systems, where fire has been shown to facilitate colony expansion and colonization of new areas by reducing vegetation height and increasing bare ground, improving both black-tailed prairie dog mobility and habitat (Augustine et al. 2007). However, the magnitude of this benefit from fire may vary considerably with annual precipitation and plant productivity (Augustine et al. 2007). In the Chihuahuan Desert, where overall vegetation cover and height are much lower, vegetation barriers to mobility are probably less important to black-tailed prairie dog dispersal. However, experimental studies in the Chihuahuan Desert have shown that colony expansion is positively linked with recent fire (Northcott et al. 2008). Fire can also alter the structure of plant communities, especially the relative abundance of woody shrubs, grasses, and bare ground. Low intensity burns, which are more frequent in the Chihuahuan Desert due to relatively low fuel loads, may select for larger shrubs that aren't consumed by fire, but many questions remain about how this process may affect black-tailed prairie dog.

15.3.2 Invasive Species and Landscape Restoration

Chapters 2-3 discuss the ecological consequences of non-native species across the U.S. portion of the ecoregion in general and Chapters 5-7 discuss the consequences of invasive species introductions specifically across the terrestrial ecological systems of the region in which the black-tailed prairie dog lives. Figure 15-1, the stressor model for black-tailed prairie dog in the U.S. portion of the ecoregion, does not include a driver concerning non-native species and their management, because little is known about this subject. The stressor model does identify infectious agents as a critical environmental element, with particular emphasis on sylvatic plague (*Yersinia pestis*), a bacterial infection of Eurasian origin that can infect a variety of mammal species, including the black-tailed prairie dog.

The close proximity of individual animals within colonies makes black-tailed prairie dog particularly susceptible to outbreaks of sylvatic plague, which has a mortality rate of nearly 100% in the species and can reduce reproductive success to nearly 0% in infected colonies. More severe outbreaks can cause local extinctions. Sylvatic plague typically is blamed as one of the primary threats to black-tailed prairie dog survival, but the disease is a more significant threat in the shortgrass habitat in the north, compared with the drier desert grasslands in the south. Savage et al. (2011) examined how environmental conditions, including temperature and precipitation regimes, affect the persistence of both plague bacteria and their primary vector, fleas. In general, incidence of plague outbreaks was higher in years with warmer winters and cooler, wetter summers, likely because these conditions are more conducive to large flea populations. Wetter summers increase soil moisture, which likely results in lower desiccation levels for larval fleas. It is not clear precisely how a drier, warmer climate in the Chihuahuan Desert will affect flea populations and plague outbreaks, but it seems probable that less summer rainfall and hotter days may further reduce the impact of the disease on black-tailed prairie dog.

Savage et al. (2011) also found correlations between colony dynamics, specifically colony size and the distance between colonies, and the incidence of plague outbreaks. However, their study did report some outbreaks in colonies that were seemingly isolated by large distances from other colonies and did

not examine outbreaks in the context of metapopulation structure and long-term colony persistence. Outbreaks in isolated colonies may be explained by other mammals serving as vectors for infected fleas, but this aspect of the disease's ecology has not been quantified. Northern grasshopper mice (*Onychomys leucogaster*), which have been suggested as a vector to carry plague between black-tailed prairie dog burrows (Kraft and Stapp 2013), along with other rodent species are potential vectors. The structure of black-tailed prairie dog metapopulations may be a better indicator of long-term resilience and colony persistence in the face of local plague outbreaks and extinctions caused by other threats. George et al. (2013) characterized a healthy metapopulation as one with small enough distances between colonies to facilitate dispersal and recolonization, as well as at least some colonies that have large populations. This is based on the assumption that plague outbreaks and local extinctions are inevitable events that occur in all populations, so maintaining the ability of colonies to recover through sheer numbers and high mobility may be essential to maintaining colonies over the long-term. Even though plague may be less common in black-tailed prairie dog populations in the Chihuahuan Desert, this focus on metapopulation health is useful when dealing with any change agent that may depress the population of colonies or cause local extinctions.

However, the impacts of this disease are not even across the geographic distribution of the black-tailed prairie dog. Environmental conditions in the desert grasslands of the Chihuahuan Desert, specifically the extreme summer heat and aridity, may be less conducive to the spread and persistence of sylvatic plague, compared to the conditions further north in shortgrass prairie habitat. Climate change in the Chihuahuan Desert is expected to further the extreme parameters that hinder the plague in these habitats. For these reasons, disease remains an important factor to consider in the Chihuahuan Desert but may not be a key limiting factor of survival and recovery of black-tailed prairie dog in this ecoregion. Although, further study and management of the disease is critical at the species level because the plague remains one of the most significant contributors to individual mortality, low reproductive success, and local extinctions across much of the overall distribution of the black-tailed prairie dog.

Few studies have examined the interaction between black-tailed prairie dog and invasive plant species, including plant species that replace native forage. Black-tailed prairie dogs consume some invasive species and may potentially facilitate their dispersal and expansion, but more work is needed to examine this relationship (Magle and Angeloni 2011). The impacts of invasive plant species on black-tailed prairie dog colony size, reproductive success, metapopulations, and other ecological outcomes have not been examined. Landscape restoration can be a valuable tool for increasing the local population of black-tailed prairie dogs or reintroducing them to areas where they have been extirpated, especially if the type of ground cover and vegetation height are outside of the species' habitat parameters. Dispersal of black-tailed prairie dog can be significantly increased through activities that reduce grass height, including burns, mowing, and mechanical removal of shrubs (Sackett et al. 2012). As climate change continues to alter habitat in the Chihuahuan Desert, restoration activities may become more essential to restoring and maintaining black-tailed prairie dog metapopulations.

15.3.3 Development

Chapters 2 and 3 discuss the history and consequences of land development across the ecoregion in

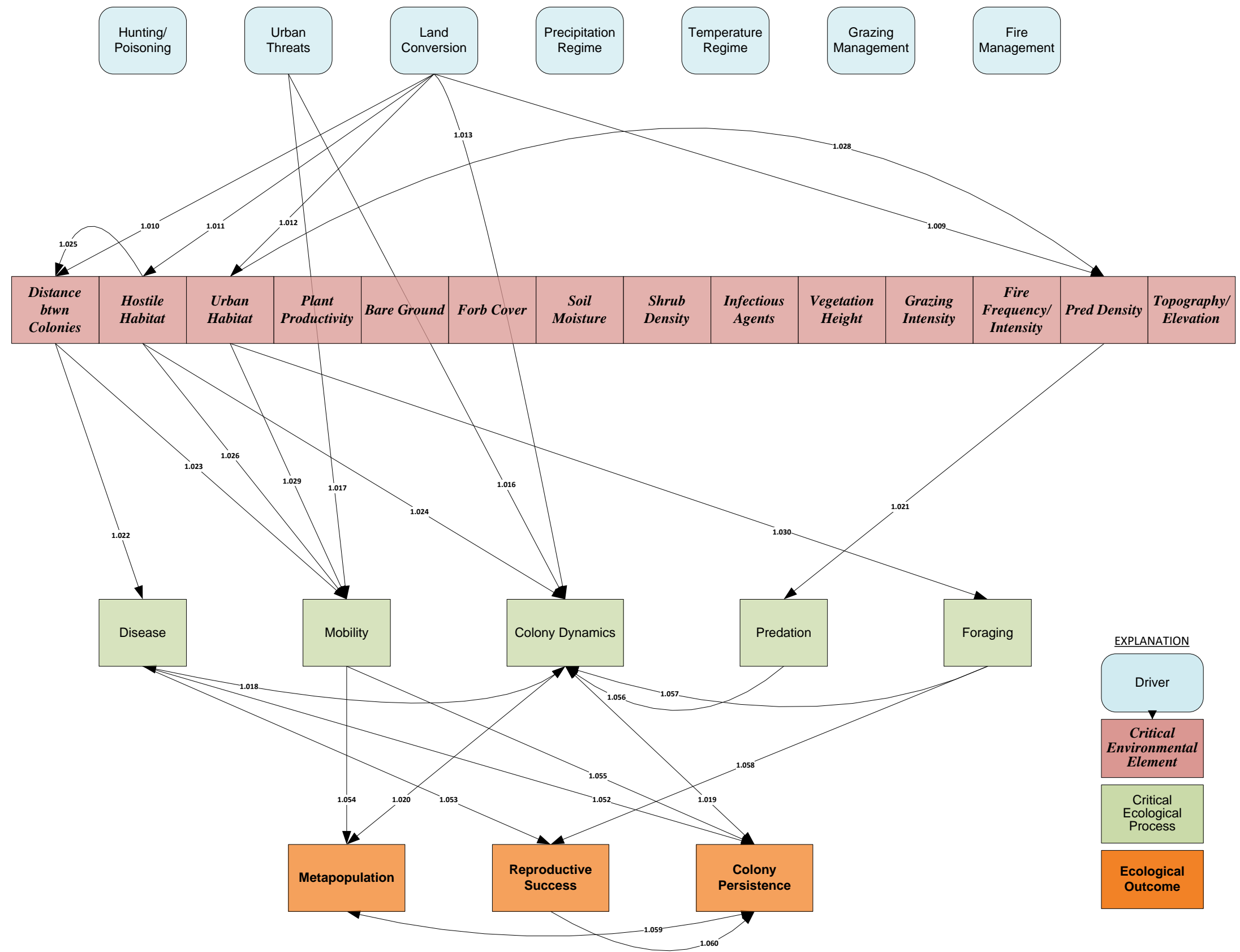
general, and Chapters 5-7 discuss these consequences specifically across the terrestrial ecological systems of the region, in which the black-tailed prairie dog lives. Figure 15-3 presents the stressor model for the black-tailed prairie dog in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by land development. See Appendix 1 for a full presentation of the stressor model.

Land conversion is a key driver in habitat loss for black-tailed prairie dog, as grassland is converted to both agricultural cropland and exurban and urban development. Across the species' range, native grasslands have been reduced by more than 33% from these land-use changes (Basurto and Hadley 2006). Human development can negatively impact black-tailed prairie dog in many ways. The most obvious threat it poses is the net loss of habitat, including the increased fragmentation of intact native grassland.

Development and fragmentation can also affect the mobility of black-tailed prairie dog, an essential element of healthy metapopulations in any colonial species. Even if black-tailed prairie dogs are able to move through developed areas, their mobility and dispersal may be limited if fragmentation separates habitat patches by more than the maximum dispersal distance of individual animals. If dispersal is limited, areas that experience local extinctions may not be repopulated by individuals from nearby colonies, reducing the effective carrying capacity of an area. Black-tailed prairie dogs are resilient in their ability to move through many types of developed area. Sackett et al. (2012) showed that the species moved moderately well through lightly developed urban habitat and over roads and very well through agricultural areas, but denser development and increasing distances between colonies pose a threat to long-term colony persistence and metapopulations.

Black-tailed prairie dogs also face other direct urban threats related to human activity, some of which may also influence their mobility. Development likely increases mortality from domestic animals, especially cats and dogs, inadvertent exposure to environmental contaminants through run-off and other vectors, and collisions with motor vehicles. Human activities that intentionally kill black-tailed prairie dog and likely increase with development, including hunting and poisoning, are discussed below under the 'Hunting and Poisoning' heading. Even if human activities do not directly injure or kill black-tailed prairie dog, the presence of humans causes disturbances that impact the rodent's behavior. Few studies have examined these impacts, with most focused on changes to daily time budgets. Colonies in urban areas appear to spend more time on vigilance than those in more rural areas, although observations over multiple years have shown that the fraction of daily time spent on vigilance decreases in urban populations overtime as they adapt to higher levels of stimuli (Magle and Angeloni 2011, Ramirez and Keller 2010). To the best of our understanding, black-tailed prairie dog persistence is not necessarily incompatible with human driven land conversion but strategic planning that minimizes both direct threats and barriers to mobility will help ensure the long-term viability of the species in human altered landscapes.

Figure 15-3. Black-tailed prairie dog stressor model: Potential impacts of development.



15.3.4 Excessive Domestic Grazing

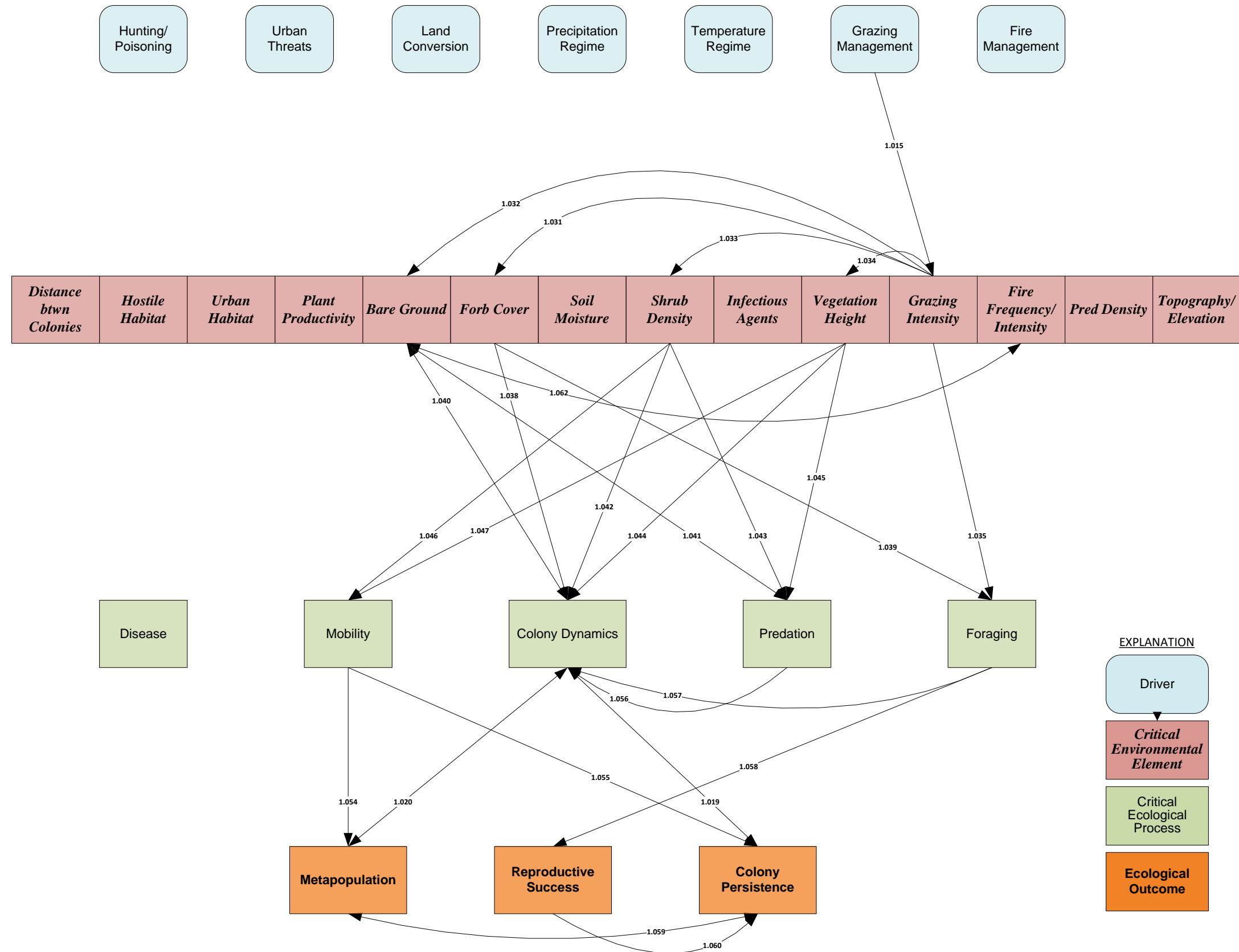
Chapters 2-3 discuss the ecological consequences of excessive domestic grazing across the U.S. portion of the ecoregion in general and Chapters 5-7 discuss the history and consequences of livestock grazing specifically across the terrestrial ecological systems of the region in which black-tailed prairie dogs live. Figure 15-4 presents the stressor model for the black-tailed prairie dog in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by excessive domestic grazing and its management. Appendix 1 presents the rationale and citations for each causal link shown in Figure 15-4.

Historically, black-tailed prairie dogs have been considered a nuisance by livestock owners for their perceived competition with cattle for forage. This conflict has driven widespread extermination efforts that led to extirpation of black-tailed prairie dog in many parts of its range and a substantial decline in its total population. More recent studies have shown that the relationship between cattle and black-tailed prairie dog is complex and that the rodents' net impact is dependent on other environmental factors, especially precipitation. In the shortgrass habitat of the northern part of black-tailed prairie dog's range, cattle are primarily limited by their digestion rate and not by the amount of forage available (Augustine and Springer 2013). Further, the presence of black-tailed prairie dog may actually benefit cattle in a positive feedback loop in these shortgrass systems. The selective browsing of black-tailed prairie dog helps increase the forage diversity and potential for cattle, which then selectively graze within black-tailed prairie dog colonies, helping to control vegetation height within the colony and the surrounding area, a benefit for maintaining visibility and increasing colony size (Basurto and Hadley 2006, Miller et al. 2007). Miller et al. (2007) also showed no significant difference in cattle weight gain when they were grazed in shortgrass systems with and without black-tailed prairie dog, but did not examine the relationship in desert grasslands.

Desert grassland systems, such as those in the Chihuahuan Desert, experience less precipitation than shortgrass systems and have less overall plant productivity available for forage. In these more arid regions, it has been suggested that cattle are limited by forage ability, rather than digestion rate, and may compete with black-tailed prairie dog for the same limited vegetation (Basurto and Hadley 2006, Cheng and Ritchie 2006). However, this suggestion has not found empirical support. Avila-Flores et al. (2010) found no correlation between the location of black-tailed prairie dog colonies and the presence of cattle grazing in the region. Davidson et al. (2010) experimentally manipulated the presence of both cattle and black-tailed prairie dog in the Chihuahuan Desert to examine how each species' presence affected the local abundance of the other species. They found that black-tailed prairie dog abundance in trials with cattle was double that of trials without cattle, suggesting that the two species may have a synergistic relationship even when forage opportunities are lower. Other work in the Chihuahuan Desert found that black-tailed prairie dog benefit from the large grazing ranges of cattle, which can reduce vegetation height near burrows and open up new areas for colonization (Basurto and Hadley 2006). Nevertheless, the ecological interactions of black-tailed prairie dog and cattle in the Chihuahuan Desert

remain incompletely understood, especially the relative impacts of differing grazing intensity and the interactions between grazing and plant productivity that could lead to foraging competition between these species (Cheng and Ritchie 2006). Further study is essential to inform grazing management strategies that have the potential to benefit both species and minimize conflict between stakeholders in the region.

Figure 15-4. Black-tailed prairie dog stressor model: Potential impacts of excessive domestic grazing.



15.4 Black-tailed Prairie Dog Key Ecological Attributes

All ecological outcomes and critical ecological processes in the black-tailed prairie dog stressor model constitute the key ecological attributes for the CE. The list below identifies eight key ecological attributes for the black-tailed prairie dog based on these criteria. *Fully characterizing the present condition of the CE will require data on indicators for its key ecological attributes.* Indicators are determined during Phase II of the REA process. The definitions for the key ecological attributes are the same as the definitions for these model components presented above.

- **Ecological Outcomes**
 - Colony Persistence
 - Metapopulation
 - Reproductive Success
- **Critical Ecological Processes**
 - Colony Dynamics
 - Disease
 - Foraging
 - Mobility
 - Predation

16 Grassland Bird Assemblage Conceptual Model

Numerous bird species endemic or near-endemic to the grasslands of the Chihuahuan desert for at least part of their life cycle have experienced steep population declines throughout the 20th century (Pool et al. 2012, Sauer et al. 2014). These Chihuahuan desert grassland obligate bird species have similar ecological requirements. The grassland bird assemblage CE for the Chihuahuan Desert REA includes five of these declining species: Arizona grasshopper sparrow (*Ammodramus savannarum ammolegus*), Baird's sparrow (*Ammodramus bairdii*), Cassin's sparrow (*Aimophila cassinii*), chestnut-collared longspur (*Calcarius ornatus*), and scaled quail (*Callipepla squamata*). These five species are appropriate targets for conservation focus due to their limited range and declining populations. Additionally, they are diverse enough in their life history and habitat requirements to represent the ecological requirements of the grassland bird assemblage as a whole within the U.S. portion of the ecoregion.

This chapter presents the conceptual ecological model for the grassland bird assemblage CE for the Chihuahuan Desert REA. The presentation follows the structure described in Chapter 4, with sections on sources of information, an overview of the assemblage, the stressor model, and key ecological attributes. As noted in Chapter 4, the conceptual models for species and species assemblages do not include a separate control model.

16.1 Sources of Information

The grassland bird assemblage overview and stressor model integrate information from numerous sources, including those identified in the Grassland Bird Assemblage Overview, below.

16.2 Grassland Bird Assemblage Overview

The Chihuahuan Desert is an important ecoregion that provides both breeding and wintering habitat for numerous bird species. Because grasslands are one of the most endangered habitats in North America, the habitats of this region are of particular conservation interest (Askins et al. 2007, Pidgeon et al. 2001).

16.2.1 Distribution

The guild of grassland birds is experiencing the steepest population decline of any group of North American avifauna (Sauer et al. 2014). Declining members of this grassland guild include loggerhead shrike (*Lanius ludovicianus*), Sprague's pipit (*Anthus spragueii*), Cassin's sparrow (*Aimophila cassinii*), Baird's sparrow (*Ammodramus bairdii*), grasshopper sparrow (*Ammodramus savannarum*), savannah sparrow (*Passerculus sandwichensis*), lark bunting (*Calamospiza melanocorys*), chestnut-collared longspur (*Calcarius ornatus*), western meadowlark (*Sturnella neglecta*), eastern meadowlark (*Sturnella magna*), and bobolink (*Dolichonyx oryzivorus*) (Sauer et al. 2014).

The five grassland obligate bird species and subspecies that make up the grassland bird assemblage CE for the Chihuahuan Desert REA all show trends of declining abundance, as noted above. Their average annual trends in abundance during the breeding season from 1966 to 2012 both range wide and within the Chihuahuan Desert, if applicable, are as follows (Sauer et al. 2014) {percent change range wide,

percent change in Chihuahuan Desert ecoregion):

- Arizona grasshopper sparrow (no sub-specific data) {-2.86%, not available}.
- Baird's {-2.96%, not available}.
- Cassin's sparrow {-1.56%, -1.4%}.
- Chestnut-collared longspur {-4.23%, not available}.
- Scaled quail {-3.14%, -3.17%}.

Management actions that benefit these species should have positive impacts on other members of the grassland bird guild (Pool et al. 2012).

Two of the species that make up the grassland bird assemblage CE for the Chihuahuan Desert REA, chestnut-collared longspur and Baird's sparrow, are winter residents of the ecoregion, while the other two species and the single subspecies, Arizona grasshopper sparrow, Cassin's sparrow, and scaled quail, are year-round residents in the ecoregion. Baird's sparrow and chestnut-collared longspur are both migratory species that breed in the northern Great Plains and migrate to the southwestern United States and Mexico to winter (Green et al. 2002, Hill and Gould 1997). Baird's sparrow is nearly endemic to the Chihuahuan Desert during the winter. Its limited winter range extends from northern Mexico into southeastern Arizona and southwestern New Mexico (Green et al. 2002). Chestnut-collared longspur has a larger wintering range that extends beyond the Chihuahuan Desert into central Texas and western Oklahoma, but the Chihuahuan Desert is recognized as hosting a significant portion of the species' wintering population (DeBano 1999, Sauer et al. 2014).

The other two species and the single subspecies that make up the grassland bird assemblage CE for the Chihuahuan Desert REA, Arizona grasshopper sparrow, Cassin's sparrow, and scaled quail, are year-round residents in the ecoregion and rely on Chihuahuan Desert habitats for nesting during the summer months. The Arizona grasshopper sparrow is an understudied subspecies that occurs almost exclusively in the grasslands of southern Arizona (Phillips et al. 1978, Vickery 1996) and uses only a small portion of the Chihuahuan desert. This subspecies is believed to be primarily non-migratory and uses the region for both breeding and wintering, although more study is needed (Vickery 1996). In the northern portion of the Chihuahuan Desert (New Mexico and portions of Arizona and Texas), Cassin's sparrow is primarily a summer breeding resident with populations breeding as far north as Nebraska. Nearly the entire population of Cassin's sparrow migrates to the desert Southwest and Mexico for the winter months. The species is present year round in parts of southeast Arizona, northern Mexico and western Texas while its wintering range extends through much of the Chihuahuan Desert in Mexico (Dunning Jr et al. 1999). Scaled quail is a non-migratory species that ranges from the northern Chihuahuan Desert, east to northern Texas and western Oklahoma, and south throughout northern Mexico (Schemnitz 1994). Despite its large year-round range, relative to the other focal species, scaled quail is an important focal species due to severe range contractions and near extirpation in the northern and eastern portions of its range during recent decades, as well as its recreational value as a game bird (Bristow and Ockenfels 2006, Rollins and Carroll 2001, Saiwana et al. 1998, Schemnitz 1994). The breeding biology and the factors that influence breeding success for these two species and single subspecies have received little study in the Chihuahuan Desert and remain a significant data gap for the conservation and management

of grassland obligate birds.

All five species of interest occur primarily within grassland habitats in the Chihuahuan Desert. While the other major habitat type within the Chihuahuan Desert, shrubland, also contains bird species that are experiencing population declines, we have chosen not to focus on those species for several reasons. First, shrublands and desert scrub habitats are not considered a habitat under serious threat in much of North America and the total acreage covered by this land cover type has expanded over the last 150 years in the Chihuahuan Desert (Coffman et al. 2014, Pidgeon et al. 2001). Understanding and monitoring this habitat succession is important because of the strong association grassland obligate birds show for habitat dominated by grasses with minimal shrubs. Second, many shrubland bird species readily colonize grasslands with low to moderate levels of shrub encroachment. Some shrubland bird species, including the loggerhead shrike, actually exhibit peak abundance in grasslands with very low levels of shrub cover (~5%), which corresponds to peak or near peak abundances for many species of grassland obligates including several of the species included in the grassland bird assemblage CE for the Chihuahuan Desert REA (Naranjo and Raitt 1993, Pool et al. 2012). Third, while some shrubland bird species are experiencing declines in abundance, that trend is not uniform across the guild and the magnitude of most declines in abundance are not as large as those of grassland obligate birds (Sauer et al. 2014). Many of these shrubland bird species do not exhibit the same degree of habitat specialization and occur over a larger geographic range and in a larger diversity of land cover types beyond the Chihuahuan Desert.

16.2.2 Habitat

Woody shrubs and grasses are the two dominant types of vegetation present in the Chihuahuan Desert, with many species of each present in varying abundances across the region. While some areas are dominated primarily by shrubs or grasses, much of the total habitat in the region is a composite of both vegetation types. Since the widespread introduction of cattle and horses and other large-scale impacts by American settlers began in the late 1800s, it is estimated that 25-50% of habitat historically dominated by grasses has been replaced with shrublands or composite habitats (Dinerstein et al. 2000). In total, about 20% of the total area of the Chihuahuan Desert is currently dominated by grasses, with the remaining 80% consisting of grass-shrub composite and shrub-dominated landscape (Dinerstein et al. 2000). The United States portion of the Chihuahuan Desert has the largest sections of contiguous grassland remaining, while grassland habitat in Mexico is largely fragmented (Askins et al. 2007).

Overall, composite habitats co-dominated by grasses and shrubs have higher species richness and diversity than shrublands, while grasslands have the lowest overall richness and diversity (Coffman et al. 2014; Naranjo and Raitt 1993, Pidgeon et al. 2001, Whitford et al. 1998). However, composite and shrubland habitats host a different suite of species than grassland habitats. Grassland obligates, including the focal species for this report, are very sensitive to shrub cover/height and demonstrate significant declines in abundance when shrub cover reaches 5-10% or higher of the total area in a habitat (Block and Morrison 2010, Macías-Duarte et al. 2009, Pool et al. 2012). Much of the avian diversity in composite habitats comes from shrubland species (e.g. Brewer's sparrow and savannah sparrow) that can readily colonize intermediate areas, in addition to widespread habitat generalists (e.g.

mourning dove) (Block and Morrison 2010). These composite habitats typically lack grassland obligates or only contain them at very low densities (Pool et al. 2012). Several studies have examined the correlation between grass cover/height and shrub cover/height on the abundance of desert grassland birds in the Chihuahuan Grasslands.

Overall, the focal species are found at the highest abundance when shrub cover is minimal (<5%), grass cover is extensive (>80%), and grass height is moderate (30-40 cm), with observed abundance declining rapidly as variables move away from these optimal parameters (Block and Morrison 2010, Macías-Duarte et al. 2009, Pool et al. 2012). Chestnut-collared longspur is the most sensitive of the focal species to shrub cover and height, with 75% of detections in one winter survey coming from plots with <1% shrub cover (Macías-Duarte et al. 2009) and other studies showing that the species exhibits a strong negative correlation with average shrub height of 2-3 m and a maximum density when average shrub height is <20 cm (Block and Morrison 2010, Pool et al. 2012). The requirements and tolerance for bare ground is understudied for most of the focal species in the Chihuahuan Desert, with the exception of scaled quail. The relationship between scaled quail abundance and the relative percent cover of bare ground has been studied but has yielded conflicting results. Chestnut-collared longspur will readily use open ground within prairie dog colonies but more study is needed to understand how its abundance varies as a function of bare ground percentage. The species seems to not occur in habitats without bare ground and prefers at least a minimal (3-5%) percent cover of bare ground, but studies have shown contradictory results as to whether there is a positive or negative correlation with abundance when bare ground percent cover exceeds 5% (Block and Morrison 2010, Bristow and Ockenfels 2006, Saiwana et al. 1998, White et al. 2011).

Managing grassland habitat for the parameters described above (<5% shrub cover, shrub height <2 m, >80% grass cover, grass height 30-40 cm) should benefit not only the focal species but most other avian grassland obligates in the Chihuahuan Desert. Cassin's sparrow is a partial exception, with a slightly different optimum for shrub cover during the breeding season. In one study during the breeding season, 75% of Cassin's sparrow detections occurred at sites with 10-14% shrub cover (Macías-Duarte et al. 2009). The species does show a significant negative correlation with shrub height of 2-3 m (Block and Morrison 2010). This preference for minimal shrub cover in the summer is likely related to the species' ability to nest in both low shrubs and grasses (Dunning, Jr. et al. 1999) and demonstrates the need for maintaining a mosaic habitat with different levels of shrub cover. Given the extensive loss of grasslands to shrub encroachment in recent centuries, restoration management to reduce shrub cover and height and increase grass cover is an important tool for grassland obligate bird conservation. Habitat restoration activities include mechanical shrub removal, herbicide applications, and native grass plantings. Because most grassland obligate birds exhibit non-linear responses to changes in the habitat parameters described above, restoration activities that target areas which have parameter values near optimum should provide the largest increase in grassland obligate bird carrying capacity. Several other changes drivers, including grazing management, fire management, and invasive species management, also impact habitat composition and are discussed below.

The ability of a habitat to provide food resources is a major factor in the abundance and density of grassland birds that it can support. All of the focal species and most grassland obligate birds are at least

partially granivorous, especially during winter when the focal species rely almost entirely on seeds, making seed abundance an important element of habitat quality (Dunning Jr. et al. 1999, Green et al. 2002, Hill and Gould 1997, Schemnitz 1994, Vickery 1996). Arthropods are the other primary food resource for the focal species and most grassland obligate birds. Insect abundance is especially important for breeding grassland obligates, including Cassin’s sparrow, Arizona grasshopper sparrow, and scaled quail, as these species all rely on insects to feed their young. Insects can also make up as much as 50% of the diet of adult birds during the summer months (Dunning Jr et al. 1999, Schemnitz 1994, Vickery 1996). While the composition of diets for the focal species has received some study, no work has been conducted to examine how differences in the abundance of food resources in Chihuahuan Desert habitats affects the occurrence, abundance, or breeding success of grassland obligate birds. Food abundance may correlate with other measures of habitat quality (e.g. grass cover, seral stage, annual precipitation), but examining it independently may shed some light on differential patterns of avian abundance and density across the Chihuahuan Desert, especially on a year-to-year basis.

Three of the focal species (Baird’s sparrow, Cassin’s sparrow, and grasshopper sparrow) have been documented as being relatively sedentary during winter, making patch size important to their survival. Gordon (2000a) suggested that small patches of habitat and large distance between patches may negatively impact these species due to their lack of mobility to transition between these patches and access sufficient resources. However, this hypothesis has not been tested for its impacts on survivorship or abundance in these species. Macías-Duarte et al. (2009) also found that the geographic location of specific habitat patches in relation to migratory routes can impact abundance for focal species. Physical barriers, such as mountains, which are avoided by grassland birds during migration, may lead to differential occupancy rates across the region, boosting abundance at lower quality but easier to access sites when compared with higher quality sites obstructed by migration barriers.

16.3 Grassland Bird Assemblage Stressor Model

Table 16-1 presents and defines the drivers, critical environmental elements, critical ecological processes, and ecological outcomes represented in the grassland bird assemblage stressor model. The stressor model follows the methodology for species described in Chapter 4.

Table 16-1. Grassland bird assemblage definitions of stressor model components.

Model Component	Definition
Drivers	
Fire Management	Refers to any human activities to control the size, timing, intensity, or frequency of fires. Fire management applies to the control and prevention of wildfire as well as the management of prescribed burns.
Grazing Management	Refers to human activities to manage and control where and how domestic cattle are grazed within the Chihuahuan Desert. This includes the number of cattle grazed per hectare, grazing rotation strategies, and grazing intensity.

Model Component	Definition
Invasive Species Management	Refers to any human activities to reduce the abundance of invasive species in the Chihuahuan Desert, with a primary focus on invasive grasses. This management may include mechanical removal, herbicide application, and native grass plantings. Invasive Species Management may overlap with Fire Management, Grazing Management, and Habitat Management when any of those activities also impact invasive species.
Land Conversion	Refers to any human activity that converts native habitats (shrubland or grassland) into developed areas. Developed areas include urban, industrial, and suburban development as well as agricultural croplands, as the croplands support low densities of grassland birds and does not provide significant habitat for any of the focal species.
Legacy Effects of Historic Grazing	Refers to any ongoing impacts to vegetation community composition and structure, especially increasing conversion of grasslands to shrublands, resulting from the legacy effects of unsustainable domestic cattle grazing in the 19th and early 20th centuries.
Legacy Effects of Prairie Dog Eradication	Refers to impacts on habitat composition and structure in the Chihuahuan Desert in the absence of grazing pressure from prairie dogs, a species that was historically present throughout most of the region but has been extirpated from all but a few areas.
Other Anthropogenic Disturbances	Refers to impacts from human activities not included in the other drivers in this model. These impacts include, but are not limited to, pollution and noise from development and the impact of recreational activities in and near grassland bird habitat.
Precipitation Regime	Refers to the pattern, timing, intensity, and average annual rate of precipitation in the Chihuahuan Desert.
Restoration Management	Refers to human activities, including mechanical shrub removal, herbicide application and native grass plantings, which maintain or increase the percent cover of native grasses and/or reduce the percent cover of shrubs in a specific region. This driver does not include invasive species management, fire management, or grazing management, which are treated separately in this framework.
Temperature Regime	Refers to the patterns of air temperature in the Chihuahuan Desert across all seasons.
Critical Environmental Elements	
Bare Ground	Refers to the average percent cover of bare ground within a specific region.
Boundary Length	Refers to the sum total length of boundaries between different habitat types (shrubland, grassland, and mixed) as well as between native habitats and human development (including croplands). Specific subtypes of these habitat boundaries may correlate with and impact grassland birds in differing ways but further study is needed to parse out these effects.
Distance btwn Patches	Refers to the average distance between patches of grassland habitat (<5% shrub cover and >80% grass cover) that are separated by other habitat types or development.
Fire Frequency/Intensity	Refers to the specific intensity of burns (characterized as low, medium, or high intensity) and the frequency with which these burns occur in a specific area. This element can be applied to both wildfire and prescribed burns.
Grass Cover/Height	Refers to the average percent cover and average height of native grasses within a specific region.
Habitat Composition/Structure	A grouping of three, related critical environmental elements (Grass Cover/Height, Shrub Cover/Height, Bare Ground) that refers to the basic vegetation composition of a specific habitat, in addition to the average heights of different vegetation classes within a habitat. Since forbs represent a low and minimally variable percentage of total cover in the Chihuahuan Desert, they are not included in this measure.
Habitat Spatial Elements	A grouping of three, related critical environmental elements (Patch Size, Distance btwn Patches, and Boundary Length). This grouping represents the physical size and spatial distribution of grassland habitat within broader landscape mosaics.
Insect Abundance	Refers to the amount of insect prey for grassland birds available in a given habitat.
Invasive Grass Cover	Refers to the percent cover of any invasive grass species within a specific region.

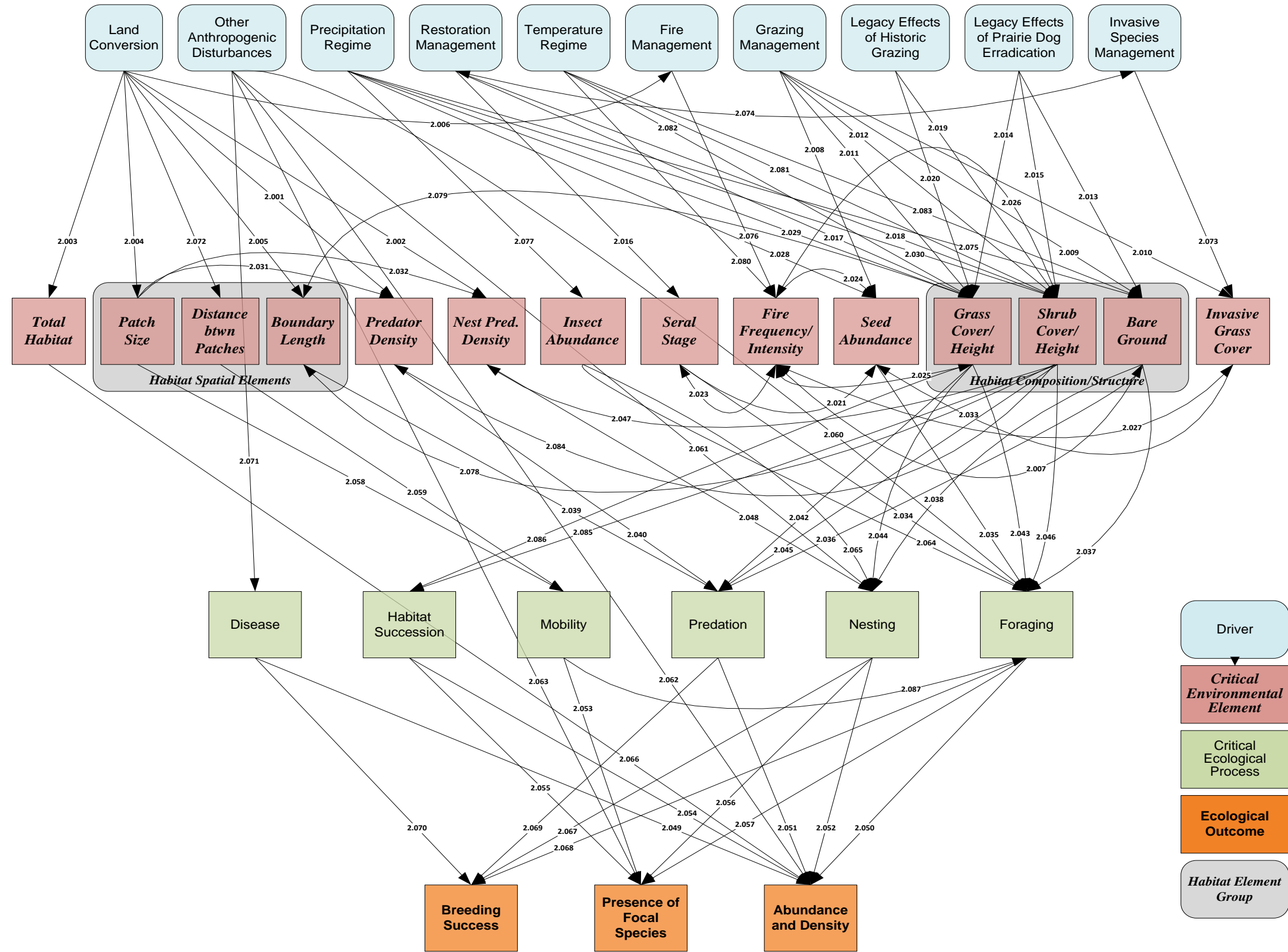
Model Component	Definition
Nest Pred. Density	Refers to the density and type of predators that prey on the eggs and nestlings of grassland birds, as well as the brood parasite Brown-headed Cowbird.
Patch Size	Refers to the average physical size of patches of grassland habitat (<5% shrub cover and >80% grass cover) within a specific region.
Predator Density	Refers to the density and type of predators that prey on either adult grassland birds or juveniles that have fledged.
Seral Stage	Refers to the maturity of grassland communities and is usually characterized as early, mid, or mature (late). As seral stage increases (in the absence of grazing pressure), both average grass height and grass density increase.
Seed Abundance	Refers to the amount of seeds, primarily from grasses, which are available in a given habitat. Seed abundance is closely tied to the productivity and extent of grasses.
Shrub Cover/Height	Refers to the average percent cover of woody shrubs and their average height within a specific region.
Total Habitat	Refers to the total amount of grassland, shrubland, and grass-shrub composite present in the Chihuahuan Desert. It does not include land that has been developed, either for agriculture or urban uses, as these areas provide unsuitable habitat for grassland birds.
Critical Ecological Processes	
Disease	Refers to any condition affecting the health of individuals of the grassland bird assemblage, including internal and external parasites, bacterial, fungal and viral infections, and environmentally-based toxicity.
Foraging	Refers to the ability of members of the grassland bird assemblage to actively find and consume food resources within their habitats.
Habitat Succession	Refers to the natural process of Chihuahuan Desert habitat shifting from areas dominated by grass to areas dominated by woody shrubs.
Mobility	Refers to the ability of birds to both physically access habitat segments during migration and to move between patches of habit within breeding or wintering seasons.
Nesting	Refers to the activities of selecting nest sites, constructing nests, laying eggs, and rearing young in the three grassland bird assemblage species that nest in the Chihuahuan Desert (Cassin's sparrow, Arizona grasshopper sparrow, scaled quail).
Predation	Refers to mortality that grassland birds face from predators. This process includes predation pressures on birds of all life stages (eggs, juveniles, adults) and the effects of brood parasites (Brown-headed Cowbird).
Ecological Outcomes	
Abundance and Density	Refers to basic measurements of population size, abundance, and distribution for the grassland bird assemblage at both at the local and ecoregional level.
Breeding Success	Refers to the rate that the three members of the grassland bird assemblage that breed in the Chihuahuan Desert (Arizona grasshopper sparrow, Cassin's sparrow, and scaled quail) are able to successfully raise offspring to self-sufficiency.
Presence of Focal Species	Refers to the presence (detected) or absence (undetected) of the five species of the grassland bird assemblage (Baird's sparrow, Arizona grasshopper sparrow, Cassin's sparrow, chestnut-collared longspur, and scaled quail) in a given habitat.

Figure 16-1 shows the full grassland bird assemblage stressor model, displaying the model components listed in Table 16-1 along with their causal relationships. Appendix 1 describes and presents the rationale for including every causal relationship in the stressor model, with citations for each causal relationship, providing a comprehensive presentation of the stressor model. A causal relationship exists when a change in one component of the system results in a change in some other component. Change in the first component is said to “cause a change in the second component. Each chain of causation,

from driver to outcome, describes how the condition of the system likely has changed in the past, or likely would change in the future, in response to changes in its drivers.

Figure 16-1 indicates the presence or absence of causal relationships between the system model components but does not indicate the potential magnitude or other characteristics of these relationships, as explained in Chapter 4. Figure 16-1 identifies the causal relationships that have affected how the system likely has changed in the past, in response to changes in its drivers. The diagram provides a means for articulating how the condition of the system will likely change in response to changes in its drivers. As discussed and illustrated below, the stressor model makes it clear: (a) which critical environmental elements would likely be affected by a change in one or more particular drivers, including change agents; (b) which critical ecological processes would likely be affected by the cascading effects of these changes in critical environmental elements; and (c) which system characteristics (ecological outcomes) would likely be affected by the cascading effects of these changes in critical environmental elements and ecological processes. By doing so, in turn, the stressor model also highlights those components of the model—drivers, critical environmental elements, critical ecological processes, and ecological outcomes—that demand indicator data.

Figure 16-1. Grassland bird assemblage stressor model.



The following paragraphs discuss how each Change Agent affects the banner-tailed kangaroo rat in the U.S. portion of the ecoregion, as articulated in the stressor model for this CE. A “sub-model” diagram for each Change Agent presents a simplified version of the master stressor model diagram, showing only the direct and indirect effects of the Change Agent of interest. The text accompanying each sub-model diagram summarizes information from Appendix 1 concerning the impacts of each Change Agent, including selected citations. For each Change Agent, the paragraphs summarize present understanding of: (a) which critical environmental elements would likely be affected—directly or indirectly—by each Change Agent and (b) which critical ecological processes would likely be affected—directly or indirectly—by the cascading effects of these changes in critical environmental elements.

16.3.1 Climate Change

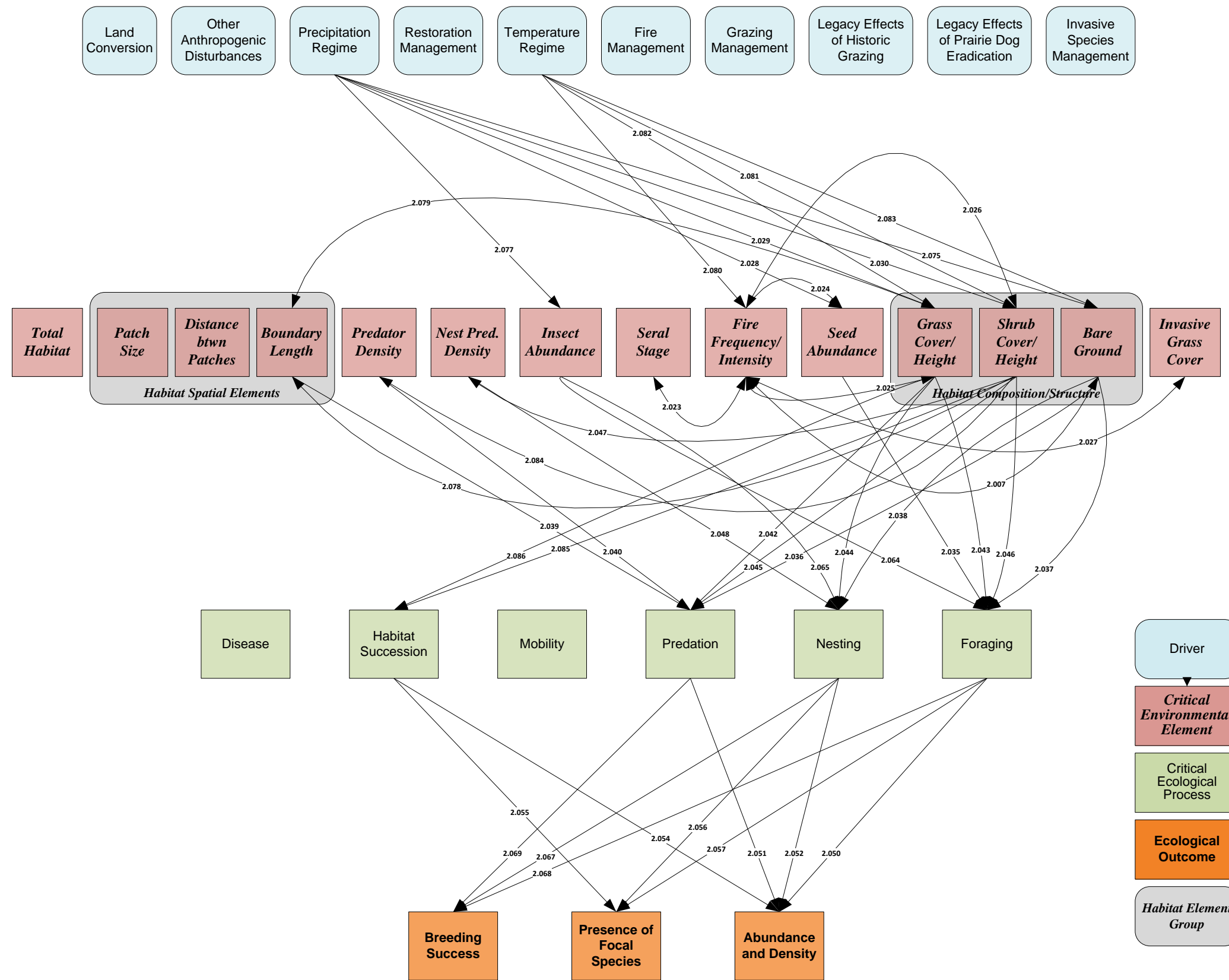
The final report for the Chihuahuan Desert REA will include a discussion of current forecasts of the ways in which climate change will affect the ecoregion and its ecological resources. The present chapter presents only the conceptual model of the causal relationships and outcomes that potentially will be affected for the grassland bird assemblage. Figure 16-2 presents the grassland bird assemblage stressor model for the U.S. portion of the ecoregion, simplified to show only those causal relationships that potentially will be affected by changes in the air temperature and precipitation regimes. Appendix 1 presents the rationale and citations for every causal link shown in the diagram.

The climate of the Chihuahuan Desert is expected to become more arid in the coming decades with conditions similar to past droughts (e.g. the Dust Bowl, the 1950s, and the current drought in the region) becoming the baseline climate for the region (Backlund et al. 2008, Seager et al. 2007). Because the desert is a hot, water-stressed environment, it is not likely that shifts to a more arid local climate will cause loss of overall arid lands to other habitat types (e.g. forest or riparian). However, drought conditions have been shown to favor shrub species over grasses, likely due to the deeper root systems of the former (Backlund et al. 2008, White et al. 2011). Drought-caused mortality in grasses is also linked to shrub encroachment into grasslands and the overall decline of habitats dominated by grasses (White et al. 2011). If predictions of sustained drought conditions in the region are accurate, this new climate will likely increase the rate of loss of grassland habitat and lower the region’s carrying capacity for grassland obligate birds.

Decreased precipitation is also linked to lower productivity in all habitat types, which in turn lowers food resource availability for animals in these habitats, especially granivores that feed primarily on grass seeds (Kerley and Whitford 2000). Lower productivity may also lead to increased grazing intensity from domestic cattle, as both more extensive and intensive grazing is required to support existing herds (Kerley and Whitford 2000). Higher grass mortality from water stress may also increase the percentage of bare ground within habitats, which will facilitate increased wind- and water-driven erosion (Backlund et al. 2008). Given that existing grassland habitat is mostly surrounded and subdivided by shrubland, a more arid climate with increased shrub encroachment is likely to further fragment grassland habitat.

Further studies are needed to better understand how patch size, distance between patches, and boundary length between land cover types affect the abundance and survival of grassland obligate birds. One study in the Texas portion of the Chihuahuan Desert demonstrated a positive correlation between the length of various land cover type boundaries and the abundance of shrub specialist and habitat generalist birds in grasslands (Gutzwiller and Barrow 2008).

Figure 16-2. Grassland bird assemblage stressor model: Potential impacts of climate change.



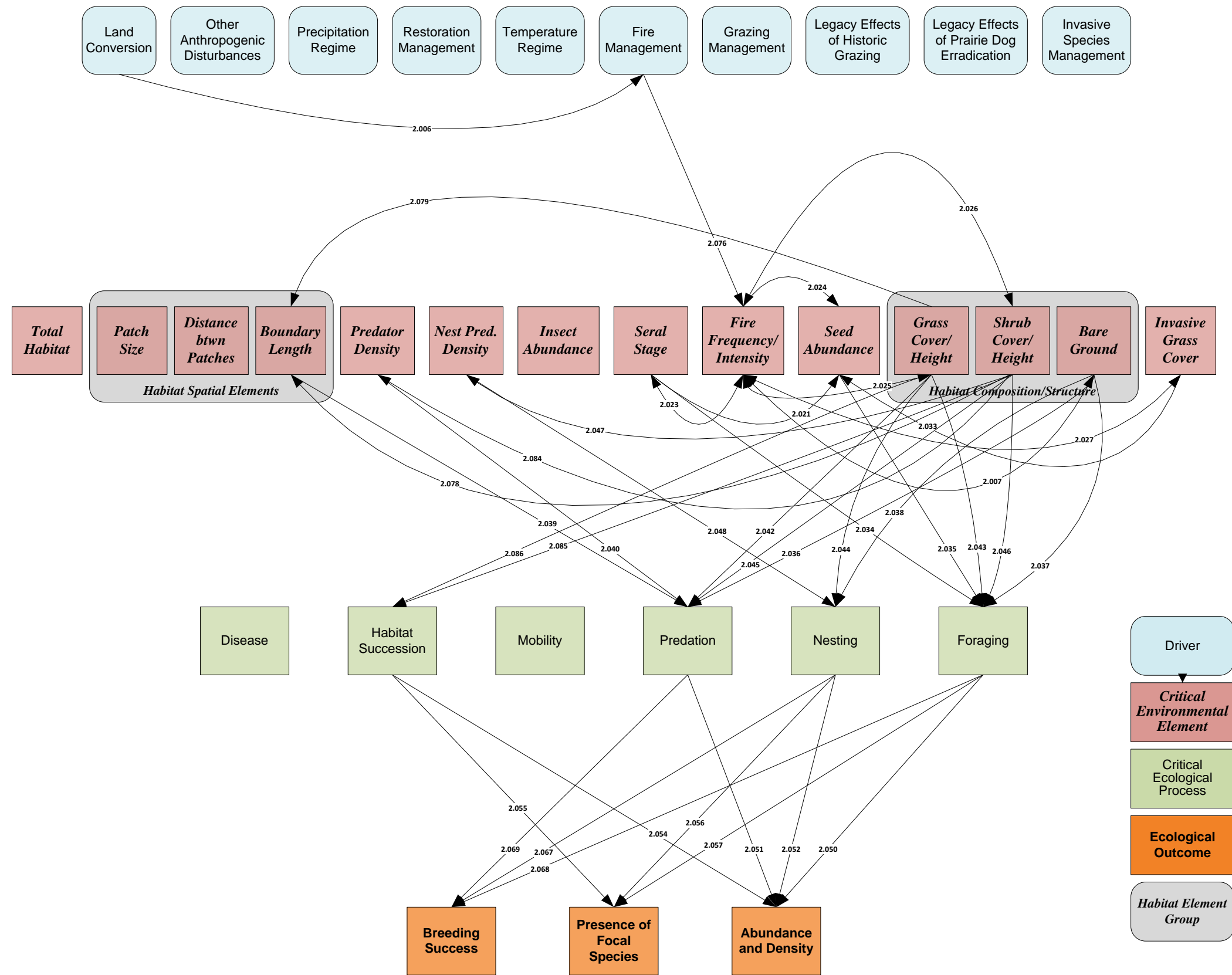
Overall, disturbance and land use will be the biggest change agents controlling the Chihuahuan Desert's response to changes in climate. As many of the organisms in the region are pushed closer to their physiological limits, they will likely become increasingly sensitive and susceptible to outside stressors (Backlund et al. 2008).

16.3.2 Uncharacteristic Wildfire

The fire regime of the U.S. portion of the Chihuahuan desert has changed as a result of the interaction of several drivers, as discussed in several previous chapters. Chapters 2 and 3 discuss the history, causes, and consequences of uncharacteristic wildfire across the ecoregion in general, and Chapters 5-7 discuss the causes and consequences of altered fire regimes specifically across the terrestrial systems of the region, in which the species members of the grassland bird assemblage live. Figure 16-3 presents the stressor model for the grassland bird assemblage in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by uncharacteristic wildfire. See Appendix 1 for a full presentation of the stressor model.

Due to their potential impacts to human property and lands used for domestic cattle grazing, natural fires in the arid west are generally controlled or suppressed by fire management activity. Prescribed fire has been used in some areas to prevent the accumulation of high fuel loads and to assist in habitat management. However, high fuel loads do not typically occur naturally in the Chihuahuan Desert, so management actions to reduce them may not be necessary in most areas (Askins et al. 2007). Fire is often described as a management tool to help restore habitats that have experienced encroachment by woody shrubs or invasive species by allowing the system to 'start over'. However, there is little data supporting the use of fire to restore grassland bird habitat and several studies have suggested that it may actually benefit shrubs (Askins et al. 2007, McGlone 2013). Prescribed fire has also been shown to have no impact on controlling or promoting the spread of either invasive Lehman lovegrass or native grasses in the Chihuahuan Desert (McGlone 2013). High intensity fires are sufficient to consume nearly all vegetation in a burn area, including large shrubs, but fuel loads to support fires this hot are generally not present in the Chihuahuan Desert and likely did not occur historically (Askins et al. 2007, Van Auken 2000). Low and medium intensity fires can consume grasses, forbs and smaller shrubs but may leave some larger shrubs intact (McGlone 2013). These lower intensity fires may provide a selective force for larger shrubs and against grasses, promoting the encroachment of shrubs into grasslands and lowering the habitat quality for grassland birds but this process is not well understood. More study is needed to evaluate how changes to fire frequency/intensity impact habitat and abundance of grassland birds in the Chihuahuan Desert.

Figure 16-3. Grassland bird assemblage stressor model: Potential impacts of uncharacteristic wildfire.



16.3.3 Invasive Species and Landscape Restoration

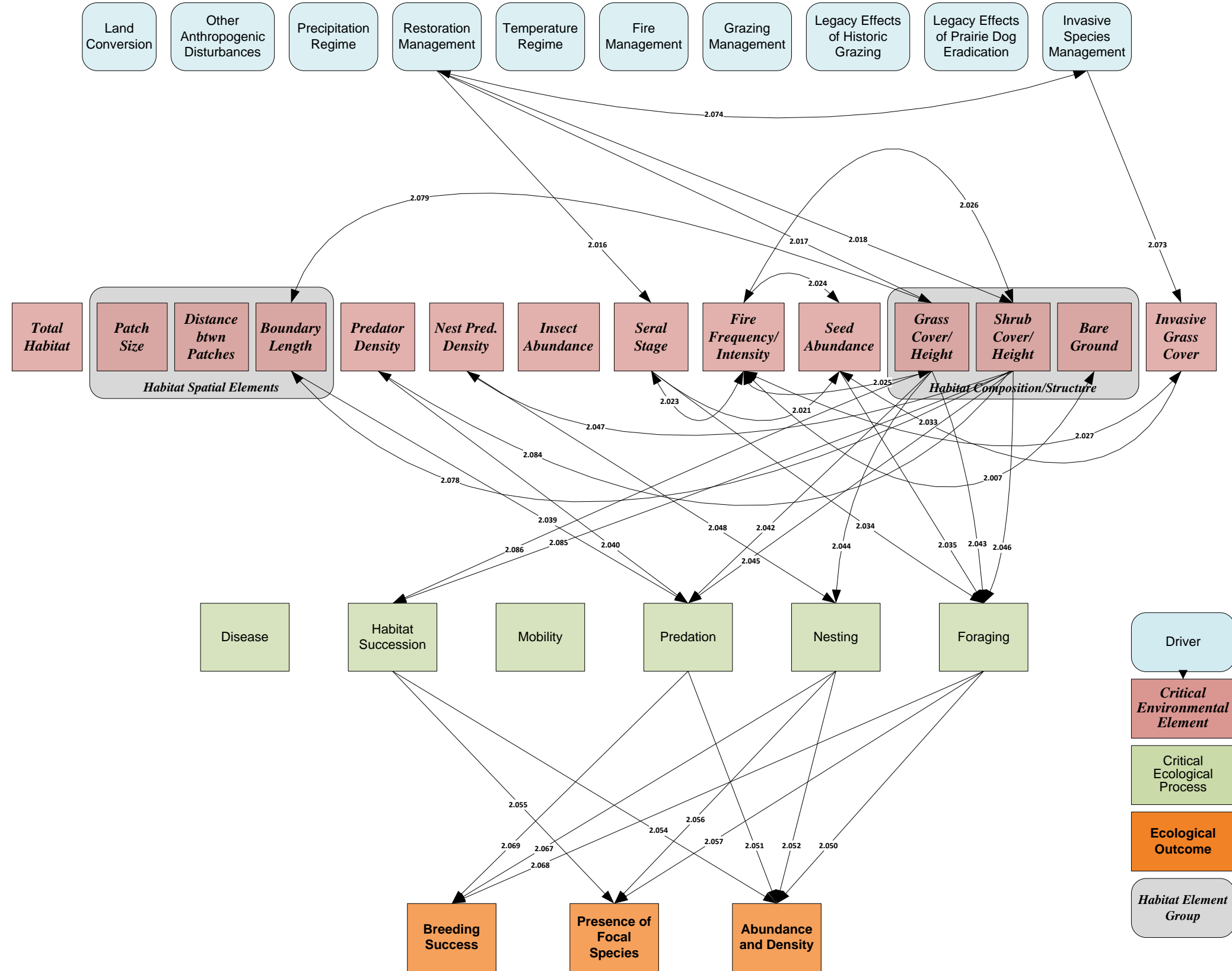
Chapters 2-3 discuss the ecological consequences of non-native species across the U.S. portion of the ecoregion in general and Chapters 5-7 discuss the consequences of invasive species introductions specifically across the terrestrial ecological systems of the region in which the species members of the grassland bird assemblage live. Figure 16-4 presents the stressor model for grassland bird assemblage in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by non-native species and their management. Figure 16-4 also includes those causal relationships affected by landscape restoration that in turn affect the grassland bird assemblage in the U.S. portion of the ecoregion. As discussed in Chapters 2-3 and 5-7, much if not most landscape restoration in the U.S. portion of the ecoregion is carried out to remove or control invasive plant species (see also Chapters 8-15). Appendix 1 presents the rationale and citations for each causal link shown in Figure 16-4.

Multiple species of invasive plants are present in the Chihuahuan Desert, with most introduced as supplemental livestock feed in the latter half of the 20th century. Lehman lovegrass is the most widespread invasive grass in the Desert and is the only species that has been directly evaluated for its impacts on grassland birds. Bock and Bock (1992b) found that plots with high invasive grass cover had less and different birds, when compared to plots dominated by native species. Among the grassland obligate species, only Baird's sparrow was found to be more numerous in areas dominated by the exotic species. More study is needed to better understand the impacts of invasive grass species on the abundance of grassland obligate birds. Given the very specific habitat needs of grassland obligates, it seems likely that significant encroachment of any exotic plants into native grasslands will have a negative impact on the avian carrying capacity in these habitats. Methods of invasive species management include mechanical removal, herbicide application, and native grass plantings.

The ecology of the invasion process by non-native species is often difficult to predict and dependent on a variety of ecological factors, especially disturbance. This makes preventing invasion difficult for land managers. McGlone (2013) found no connection between fire and the spread or control of invasive Lehman lovegrass but those results may not be broadly applicable given differences in ecological factors across different habitats within the Chihuahuan Desert. There is some evidence that invasive grasses increase in relative abundance, compared to native grasses, when exposed to grazing pressure from domestic cattle as the cattle tend to selectively browse the native vegetation at higher levels (McClaran and Anable, 1992).

Restoration management techniques, including mowing, herbicide application, mechanical shrub removal, and plantings, are a means of restoring grassland habitat that has been degraded by the expansion of invasive grasses and shrubs. Any restoration management strategy should aim for the optimum habitat parameters for grass cover/height, shrub cover/height, and bare ground as described in the 'Habitat' section. Restoration activities that reduce the cover of invasive grass or shrubs but still leave a particular site far outside the parameter optimums may show little or no improvement in their carrying capacity for grassland obligate birds, due to the non-linear responses by many species to these habitat elements. On the other hand, seemingly small changes that push near-optimum habitats into the optimum parameters may see substantial increases in their carrying capacities.

Figure 16-4. Grassland bird assemblage stressor model: Potential impacts of invasive species and landscape restoration.



It is important to note that restoration activities that reduce the amount of local habitat dominated by shrubs also reduces the carrying capacity for shrub-dwelling bird species. This trade-off is warranted in many situations, given the steep population declines of and conservation focus on grassland obligate birds.

16.3.4 Development

Chapters 2 and 3 discuss the history and consequences of land development across the ecoregion in general, and Chapters 5-7 discuss these consequences specifically across the terrestrial ecological systems of the region, in which the species members of the grassland bird assemblage live. Figure 16-5 presents the stressor model for the grassland bird assemblage in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by land development. See Appendix 1 for a full presentation of the stressor model.

The impacts of land use change and development on the Chihuahuan Desert vary across the ecoregion, with the most substantial difference occurring across the United States-Mexico border. In the United States portion of the ecoregion, a substantial portion of these arid lands are owned by federal and state governments and thus have lower direct threats from development, although there may be increasing risks from oil and gas development. Agricultural, exurban and urban land conversion are likely contributing to habitat loss in some parts of Arizona, New Mexico, and Texas but more work is needed to evaluate the extent of this threat and where it is occurring. The situation is very different in Mexico, where agricultural expansion is converting desert habitats, including grasslands, to cropland at an unsustainable rate. The present REA addresses only the U.S. portion of the ecoregion. However, members of the grassland bird assemblage move throughout the ecoregion. As a result, changes in the Mexico portion of the ecoregion affect conditions in the U.S. portion, and vice versa.

Pool et al. (2014) documented over 64,000 hectares of desert converted to cropland in the Valles Centrales region of Mexico from 2006 to 2007, including over 19,000 hectares of desert grassland and over 49,000 hectares of mixed shrub-grassland habitat. By their estimates, this accounts for an average annual decrease of 2.8% in grassland habitat. Based on previous work that demonstrated that cropland does not support as many grassland birds (Macías-Duarte and Panjabi 2013, Pool et al. 2012), Pool et al. (2014) calculated the loss in grassland bird carrying capacity for Valles Centrales between 2006 and 2011 at 600,000 individual birds, including an estimated 130,000 chestnut-collared longspurs. Our understanding of desert grassland loss in Mexico is likely incomplete, as the country's current system of recording land cover classifications does not discriminate between high quality grasslands (those with <5% shrub cover) and lower quality grasslands that can have shrub cover as high as 25% (Pool et al. 2014). Anecdotal observations from the same study suggested that areas of high quality grassland may be undergoing agricultural conversion at a higher rate than lower quality grasslands or mixed habitat. Much of this rapid conversion to cropland in Mexico has been blamed on lax enforcement of conservation laws and regulations, as Pool et al. (2014) found that only 2.8% of new agricultural development was legally permitted during the timeframe of their study.

Land conversion may also affect predator density and community composition, especially mammalian predators. Human development has the potential to introduce or increase the population of

mesomammal predators, including feral cats, armadillo, and raccoons, which prey on grassland birds or their nests. However, the human-facilitated increase of larger predators (especially coyotes) may also help to control the population of these mesomammal predators (Rollins and Carroll 2001). Scaled quail is the only focal species that has been directly studied for the impacts of predation in the Chihuahuan Desert. While predation is the largest source of mortality for both juvenile and adult birds, there is little evidence that predator removal will actually result in substantial increases in scaled quail abundance (Rollins and Carroll 2001). Further study is needed to understand the interactions between predator populations and landscape level changes that impact predation on grassland birds.

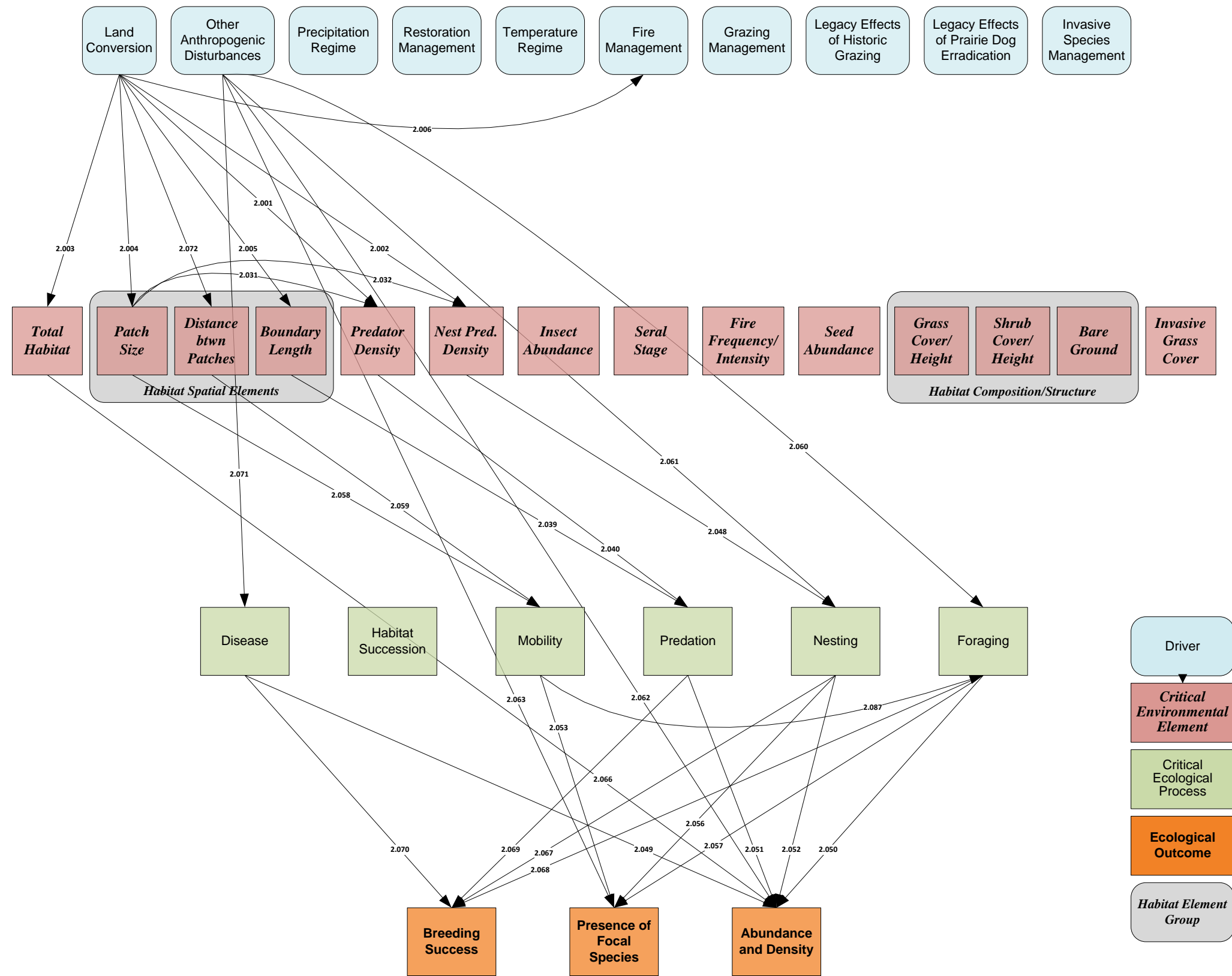
Human development may have other anthropogenic disturbances beyond habitat loss from land conversion. Some of these other disturbances may stem from the proximity of development to intact habitats and could potentially include anthropogenic noise that may disturb birds and discourage nesting, pollution from agricultural and urban areas that may contribute to disease and mortality, and disturbances from direct recreational use of grassland bird habitats. To date, the potential impacts of these disturbances on grassland obligate birds have not been evaluated in the Chihuahuan Desert, possibly because other change agents are believed to be higher priority.

Changes to habitat and other human activities are likely affecting the composition and density of grassland bird predator communities in the Chihuahuan Desert. The overall impact of these shifts on grassland bird abundance is likely mixed. Increasing density of tall shrubs may provide hunting perches and roosts for raptors that prey on small predators (rodents, lizards, snakes), thus lowering the predation pressure from these small predators on grassland birds and their nests (Macías-Duarte et al. 2009). However, these same large shrubs may also provide hunting perches for avian predators (falcons and shrikes) and avian nest predators (corvids) that increase the predation pressure on grassland birds. This potential correlation between shrub cover and predation rates on the focal species of grassland birds has not been directly evaluated. Shrub encroachment and increased grassland fragmentation may also facilitate the access of predators that otherwise would be excluded by extensive, continuous grasslands but this assertion needs to be tested further (Mason et al. 2005).

The effects of predation and brood parasitism on grassland bird abundance in the Chihuahuan Desert are also poorly understood and in need of more study. Basic information on predator density, predator community composition, and estimates of predation rates would help improve the understanding of survivorship in both juvenile and adult grassland birds. In the future, more specific information on the composition of the predator community and the respective predation rates of individual predator species may also advance understanding of these interactions. For the focal species that nest in the Chihuahuan Desert, the lack of knowledge on brood parasite and nest predator density is another key data gap.

As discussed in previous chapters (e.g., Chapters 12-15), development increases the ease with which diseases may be introduced to or spread within the ecoregion. However, disease and its impacts on the fitness, survival, and abundance of grassland obligate birds have been understudied in the Chihuahuan Desert. Understanding how drivers, including climate change and anthropogenic disturbances, affect the prevalence and type of disease found in these bird populations may be an important part to understanding their abundance, distribution, and breeding success.

Figure 16-5. Grassland bird assemblage stressor model: Potential impacts of development.



Overall, the cascading effects urban, industrial, and agricultural development ultimately will alter the quality and distribution of habitat for grassland obligate birds; the diversity of avifauna present, including the presence and abundance of the species in the grassland bird assemblage; and the health and breeding success of avifauna in this habitat.

16.3.5 Excessive Domestic Grazing

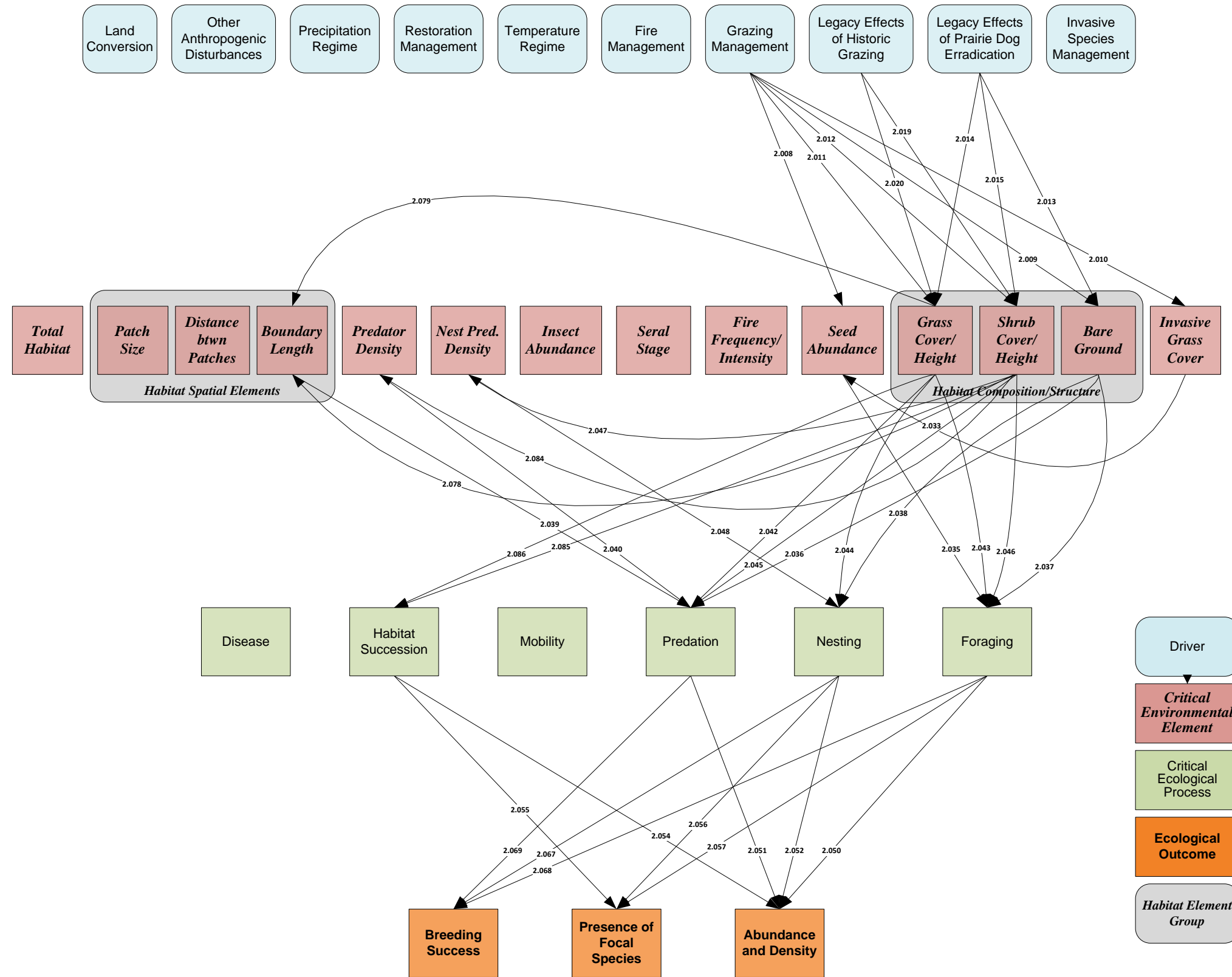
Chapters 2-3 discuss the ecological consequences of excessive domestic grazing across the U.S. portion of the ecoregion in general and Chapters 5-7 discuss the history and consequences of livestock grazing specifically across the terrestrial ecological systems of the region in which the species members of the grassland bird assemblage live. Figure 16-6 presents the stressor model for the grassland bird assemblage in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by excessive domestic grazing and its management. Appendix 1 presents the rationale and citations for each causal link shown in Figure 16-6.

The relationship between domestic livestock grazing and habitat in the Chihuahuan Desert is complex, varies with the intensity of grazing, and is impacted by interactions with local climate and other change agents. There is evidence that grazing by native mammals was historically an important part of maintaining grassland habitat within the region. The primary grazers prior to the introduction of domestic livestock in the late 17th century were prairie dogs (Desmond 2004, Askins et al. 2007) as discussed in detail in Chapter 15. Pronghorn and other large grazers were likely present only at low densities. Prairie dogs selectively browse shrubs to maintain a very low height and allow for visual detection of predators around their towns. This browsing may help limit both shrub height and percent cover within towns to a level that can be near optimum for some grassland obligate birds. Prairie dogs also browse on grasses as a food resource, reducing both their average height and percent cover. The effects of prairie dog grazing can create a mosaic environment with grasses of different heights, low shrubs, some bare ground, and taller shrubs around the periphery (Desmond 2004), which may benefit some grassland obligate birds.

Desmond (2004) closely studied the relationship between prairie dog colonies (towns) in Mexico and the abundance of three members of the grassland bird assemblage. Chestnut-collared longspur abundance correlates strongly with prairie dog abundance, and the bird regularly occurs within prairie dog colonies. This association is likely the result of chestnut-collared longspur's preference for shorter grass and some open ground, compared to other obligate grassland birds that prefer more moderate levels of grass cover. In contrast, Desmond (2004) found that Baird's and Cassin's sparrows tended to occur mostly around the periphery of prairie dog colonies where both average grass height and cover percentage were higher. The full impact of these mosaic environments on grassland obligates and the broader ecosystem is not clear and needs further examination. Even if some grassland obligates do not occur in high abundance within prairie dog colonies, the presence of these rodents may help maintain large tracts of grassland that benefit these birds (Desmond 2004, Askins et al. 2007). Prairie dog populations have declined dramatically across most of the Chihuahuan Desert since the late 19th century, with eradication efforts by settlers leading to their extirpation across much of the region, as discussed in detail in Chapter 15. The legacy effects of prairie dog eradication efforts may still be impacting habitat

composition and structure in the ecoregion. Currently, there is only one large complex of prairie dogs remaining in the Chihuahuan Desert, consisting of 58 colonies located in an area of over 30,000 hectares in the Janos-Nuevo Casas Grandes Grasslands of Chihuahua, Mexico (Desmond 2004). There are no complexes of comparable size in the U.S. portion of the ecoregion.

Figure 16-6. Grassland bird assemblage stressor model: Potential impacts of excessive domestic grazing.



Domestic cattle are not a perfect grazing substitute for prairie dogs and their impact on desert grasslands and associated avian communities is mixed. Some studies have shown that grazing management which limits cattle to light to moderate grazing intensities may help maintain grass heights that are closer to optimum for grassland obligate birds, when compared to ungrazed grasslands. At sites in Arizona, Gordon (2000b) found Baird's sparrow in higher abundance when light to moderate grazing intensity was applied. Kelly et al. (2006) found a weak positive relationship between grazing and the abundance of both horned lark and chestnut-collared longspur, and suggest that legacy effects of historic grazing in the 19th and early 20th century are a larger problem than current grazing practices. These authors assert that historic unsustainable overgrazing led to extensive shrub encroachment that is still present in the region, whereas contemporary grazing practices are better managed with smaller effects on habitat succession and there are potential benefits for grassland birds. Another study specifically examined the differences in impact on grassland birds from moderate grazing versus light grazing in a portion of the Chihuahuan Desert in New Mexico (Joseph et al. 2003). While the study found no overall significant difference in bird abundance between the two grazing management strategies, there were significant differences for the only two focal species examined. Cassin's sparrows were six times more numerous on lightly grazed plots compared to moderately grazed plots, while scaled quail relative abundance was double on lightly grazed plots. This suggests that more adaptable habitat generalist are less sensitive to increased grazing intensity and their increases in abundance may be masking the impact of grazing on grassland obligate birds with their more specific habitat requirements. Another study (Saiwana et al. 1998) examined specific impacts of various grazing intensities on scaled quail abundance. They found that no grazing on late seral stage grasslands significantly limited the abundance of scaled quail, likely because it is primarily a ground dweller and very limited bare ground and high grass heights limits mobility and impairs predator detection. Light to moderate grazing maximized abundance in these same late seral stage habitats. However, their results for early to mid-seral grasslands showed that light and moderate grazing negatively affect scaled quail abundance, as these habitats have fewer obstacles to quail mobility and grazing in them may reduce available food resources for the birds.

Grazing in desert grasslands also impacts the spread and abundance of non-native plants. Cattle have been shown to selectively browse native grasses over certain invasive grasses, including Lehmann lovegrass, facilitating increased invasive grass cover (McClaran and Anable 1992). These invasive grasses provide lower quality habitat and reduced food resources for grassland obligates, so their spread can be problematic even in habitats that are considered grasslands (<5% shrub cover) (Medina 1988).

While short-term studies (1-2 years) of grassland obligate bird abundance have shown mixed effects for light to moderate grazing intensity, on a longer time scale domestic cattle may be affecting the degree of grassland loss through shrub encroachment. Unlike prairie dogs, cattle avoid browsing on many shrubs, especially taller, more mature plants. Their focused browsing on grasses and forbs thus exerts a positive selective force for shrub cover/height (Kerley and Whitford 2000). This problem may not occur if grazing occurs at low enough intensity that it does not substantially reduce the amount of grass cover and height (Molinar et al. 2011). However, drought conditions can reduce or reverse any grazing benefits for grassland obligate birds. When drought conditions and grazing are combined they likely

cause accelerated habitat succession (Macías-Duarte and Panjabi 2013; Askins et al. 2007). Given that climate models predict increased aridity and reduced precipitation for the Chihuahuan Desert in the coming decades, grazing activities of any kind may become a negative stressor on grassland obligate birds in the near future.

16.4 Grassland Bird Assemblage Key Ecological Attributes

All ecological outcomes and critical ecological processes in the grassland bird assemblage stressor model constitute the key ecological attributes for the CE. The list below identifies nine key ecological attributes for the grassland bird assemblage based on these criteria. *Fully characterizing the present condition of the CE will require data on indicators for its key ecological attributes.* Indicators are determined during Phase II of the REA process. The definitions for the key ecological attributes are the same as the definitions for these model components presented above.

- **Ecological Outcomes**
 - Abundance and Density
 - Breeding Success
 - Presence of Focal Species
- **Critical Ecological Processes**
 - Disease
 - Foraging
 - Habitat Succession
 - Mobility
 - Nesting
 - Predation

17 Grassland Small Mammal Assemblage Conceptual Model

Seven species of small mammals have been identified as being of particular concern to resource managers in the U.S. portion of the Chihuahuan Desert ecoregion: banner-tailed kangaroo rat (*Dipodomys spectabilis*), black-tailed prairie dog (*Cynomys ludovicianus*), Chihuahua deer mouse (*Peromyscus maniculatus blandus*), hispid cotton rat (*Sigmodon hispidus*), Southern Plains woodrat (*Neotoma micropus*), tawny-bellied cotton rat (*Sigmodon fulviventor*), and the yellow-nosed cotton rat (*Sigmodon ochrognathus*). The banner-tailed kangaroo rat and the black-tailed prairie dog are addressed as separate CEs for the Chihuahuan Desert REA. The remaining five species of rodents are the focus of this report and make up the grassland small mammal assemblage CE for the Chihuahuan Desert REA.

This chapter presents the conceptual ecological model for the grassland small mammal assemblage CE for the Chihuahuan Desert REA. The presentation follows the structure described in Chapter 4, with sections on sources of information, an overview of the assemblage, the stressor model, and key ecological attributes. As noted in Chapter 4, the conceptual models for species and species assemblages do not include a separate control model.

17.1 Sources of Information

The grassland small mammal assemblage overview and stressor model integrate information from numerous sources, including those identified in the Grassland Small Mammal Assemblage Overview, below.

17.2 Grassland Small Mammal Assemblage Overview

The Chihuahuan Desert ecoregion supports a large number of rodent species (Frey and Yates 1996, Lopez-Gonzalez and Garcia-Mendoza 2012, Parmenter and Van Devender 1995). These animals are important components of desert ecosystems for two reasons: (1) they are found near the bottom of the food chain and support the upper levels of the food chain (Chew 1979), and (2) they are important consumers of plants, seeds, and invertebrates and are often considered to be ecological engineers. Maintaining healthy populations of small mammals is essential to maintaining a healthy ecosystem.

17.2.1 Distribution

The five species of concern for this paper live in a variety of habitats and feed on a variety of organisms, but all can be found in the Chihuahuan Desert grasslands and all share grasslands, or a component of grasslands, as critical habitat (see Table 17-1). All five of these species are listed as “Least Concern” by the International Union for Conservation of Nature (IUCN) (Linzey and Timm 2008). However, the yellow-nosed cotton rat is listed as imperiled in New Mexico (Natural Heritage New Mexico 2015). These species can serve as indicators for changes in the grasslands, and management that benefits these species should have positive impacts on other small mammals as well as carnivores found higher in the food chain.

17.2.2 Habitat

Over the past 150 years, up to 50% of the grasslands of the Chihuahuan Desert have been converted to shrublands, as discussed in detail in Chapters 2-3 and 5-7 (Bahre and Shelton 1993, Brown et al. 1997, Buffington and Herbel 1965, Dinerstein et al. 2000, Eve et al. 1999, Roundy and Biedenbender 1995). The term shrubland is used rather generically to include all landscapes dominated by low woody vegetation, especially mesquite (*Prosopis* spp.) and creosote bush (*Larrea tridentata*). Chapters 2-3 and 5-7 also discuss the ongoing debate over the causes of this shift to shrubland from grassland, including the relative roles of climate change (Brown et al. 1997), cattle grazing (Curtin et al. 2002), fire suppression (McPherson and Weltzin 2000), and interactions among these factors. Little is known about the effects of these habitat changes on small mammals. However, shrubland habitats host a different suite of species than do grassland habitats.

Brown and Zeng (1989) compared the population ecologies of 11 rodent species and noted great interspecific variation within the rodent species that coexist in the Chihuahuan Desert. Subsequently, Hope and Parmenter (2007) described seasonal dietary composition for 15 species of rodents collected in all major habitats on the Sevilleta National Wildlife Refuge in central New Mexico. Zwartjes et al. (2005) compiled detailed notes on four of the five small mammal species of present interest. The following paragraphs summarize this information, combined with information from other sources. Table 17-1 summarizes habitat and food preferences among the five species.

17.2.2.1 Chihuahua deer mouse

The deer mouse is common in North America and lives in nearly all habitats, including grasslands, brushy areas, woodlands, and forests (Hoffmeister 1986, Hooper 1968, Whitaker 1996). Deer mice are common to abundant in Arizona and seem capable of living in almost any habitat in New Mexico (Hoffmeister 1986). Preference for disturbed habitats has also been reported for some sagebrush and grassland communities (Oldemeyer and Allen-Johnson 1988). Deer mice are omnivorous; the main dietary items usually include arthropods and seeds. Deer mice also consume nuts, berries and other small fruits, and fungi.

In this paper we are focused on one subspecies of deer mouse, *Peromyscus maniculatus blandus*, the Chihuahua deer mouse. It is found in the U.S.A. from the lower Sonoran Zone of western Texas from Pecos Valley westward; north along the Pecos and Rio Grande Valleys of southern New Mexico (to about lat. 34° N.); westward through southwestern New Mexico to southeastern Arizona; and south in Mexico, east of the Sierra Madre in Chihuahua, southern Coahuila, southwestern Nuevo Leon, western Tamaulipas, northwestern San Luis Potosi, Durango, and Zacatecas (Boudet 2009). This subspecies is generally rare; it is now usually only found in locations that have retained a grassland character and it may be intolerant of shrub encroachment and other aspects of desertification (Brown and Munger 1985).

17.2.2.2 Hispid cotton rat

The hispid cotton rat ranges chiefly from southeastern Arizona eastward through the southern Central Plains and along the coast of the Gulf of Mexico to the east coast, from Virginia to Florida (Hall 1981;

Mohlhenrich 1961). Its distribution also extends south through Mexico into Central America, and isolated populations of the species are known in southeastern California and southwestern Arizona (Cameron 1999). The hispid cotton rat is at the western edge of its distribution in the southwest. This species seems to be limited to those areas with a mean annual temperature of more than 13 °C and a growing season of 180 days or more (Mohlhenrich 1961).

The hispid cotton rat is a fairly common species found in areas of dense grass cover, especially in riparian areas. It is a resident of grasslands, irrigated fields, marshy vegetation, and grassy or weedy riparian areas (Bailey 1932, Findley et al. 1975, Hoffmeister 1986) and is limited to areas providing dense overhead cover for runways (Mohlhenrich 1961). This species tolerates more open, disturbed habitat than does the tawny-bellied cotton rat (Mohlhenrich 1961).

Cotton rats are herbivores and consume chiefly grass and to a lesser extent forbs (Baker 1971) and they also opportunistically consume seeds, insects, and other food items (Nowak 1999). This species has been found near, and in, agricultural fields (Bailey 1932, Hoffmeister 1986) and can be considered an occasional pest of agriculture and upland game birds, especially bobwhite quail (*Colinus virginianus*) (Cameron and Spencer 1981, Findley et al. 1975, Fitzgerald et al. 1994, Schmidly 1977).

17.2.2.3 Southern Plains woodrat

The southern plains woodrat lives in short-grass and desert grasslands (Findley et al. 1975; Fitzgerald et al. 1994) and ranges from southeastern Colorado and southwestern Kansas south through grassland portions of New Mexico, western Oklahoma, and southern and western Texas into Mexico along the Atlantic slope (Fitzgerald et al. 1994, Hall 1981, Whitaker 1996). In New Mexico, it occurs in low and mid-elevation grasslands throughout most of the state (Findley 1987, Findley et al. 1975). It is not found in Arizona (Hoffmeister 1986).

The southern plains wood rat is abundant in open arid valleys and prefers grasslands associated with cactus or thorny shrubs such as creosote and mesquite (Best et al. 1993, Hallett 1982, Peterson and Boyd 1998, Schmidly 1977). These plants provide food as well as structure for construction of the nest (Findley et al. 1975, Fitzgerald et al. 1994, Whitaker 1996).

The southern plains woodrat can efficiently digest plant fiber (Braun and Mares 1989) and in Colorado and New Mexico, it preferentially consumes joints and fruits of tree cholla (*Cylindropuntia imbricata*), Opuntia cactus, *Yucca* spp., and blue grama grass (*Bouteloua gracilis*) (Braun and Mares 1989, Finley 1958). In other areas, such as the shinnery oak-mesquite grassland habitat of southeastern New Mexico, they may specialize on shinnery oak leaves and acorns; mesquite leaves, beans and pods; and other shrubs (Best et al. 1993, Braun and Mares 1989). Woodrats derive the water they need from their diet (Fitzgerald et al. 1994).

17.2.2.4 Tawny-bellied cotton rat

The tawny-bellied cotton rat ranges from Arizona, New Mexico, and western Texas south to Michoacan, Mexico (Hoffmeister 1986, Shump 1999). In New Mexico, it formerly inhabited the southern Rio Grande Valley (Findley et al. 1975). In Arizona, the tawny-bellied cotton rat is at the northern edge of its

distribution and is not common (Hoffmeister 1986).

The tawny-bellied cotton rat is dependent on a high percentage of tall, dense, undisturbed grass cover (Baker and Shump 1978, Cook 1986, Heske et al. 1994) and is considered an indicator of grasslands (Baker 1971, Hafner and Yensen 1998). Cotton rats are herbivores and consume chiefly grass and to a lesser extent forbs (Baker 1971, Cahalane 1954). They also opportunistically consume seeds, insects, and other food items (Nowak 1999). It appears to compete with banner-tailed kangaroo rats for forage as the tawny-bellied cotton rat was one of the rodents that increased in density after the experimental removal of kangaroo rats (Heske et al. 1994).

17.2.2.5 Yellow-nosed cotton rat

The yellow-nosed cotton rat’s distribution extends from southeastern Arizona, southwestern New Mexico, and southwestern Texas south into northern Mexico. It is found in grasslands and grassland-woodland associations (Findley et al. 1975) and requires large plants sufficiently close together so the animal will not be exposed for too great a distance when moving through the grass (Hoffmeister 1986). Cotton rats are herbivorous and consume chiefly grass and to a lesser extent forbs (Baker 1971). They also opportunistically consume seeds, insects, and other food items and, if available, prickly pear fruits (Davis and Sidner 1999, Nowak 1999).

Table 17-1. Grassland small mammal assemblage species, habitat preferences, and food preferences.

Species	Habitat preference	Food preference
Chihuahuan deer mouse	Grasslands	Seeds; opportunistic omnivore
Hispid cotton rat	Dense, tall grass cover; live in mesquite clumps	Grass, forbs, some seeds
Southern Plains woodrat	Grasslands with shrubs and cactus areas	Cholla, <i>Opuntia</i> , yucca
Tawny-bellied cotton rat	Tall dense undisturbed grasslands	Grasses
Yellow-nosed cotton rat	Grassy, rocky slopes with cover plants but often found in grasslands	Grasses; some seeds and insects

All five species can be found in grassland habitat that provides sufficient cover to limit exposure to predators. The hispid cotton rat and the southern plains woodrat may be more tolerant of woody vegetation than the other three species, but it is unknown if there are any thresholds in woody vegetation cover. As the small mammal assemblage changes, we may see alterations in the plant communities as well. Brown and Munger (1985) showed that removal of all *Dipodomys* species from a plot resulted in large increases in four out of five seed-eating rodents, but had no effect on insectivorous rodents. However, that research group concluded that responses to their manipulations were complex, included long time lags, and were often asymmetrical. In addition, Curtin et al. (2000) found that different rodents exert qualitatively different impacts on the environment and structural changes in plant communities. These impacts include changes in tall herbaceous growth as well as mean patch size. Brown and Heske (1990) and Whitford and Steinberger (2010) have suggested that *Neotoma* spp. are keystone ecological engineers because of their effects on nitrogen mineralization in soils surrounding their middens, and the moderation of humidity and temperature within their middens providing a

microhabitat for many invertebrate and vertebrate species. Brown and Heske (1990) even suggest that the small mammal assemblage is a “keystone granivore guild” because they act together in complex ways affecting vegetation cover and structure and that understanding the activities of small mammals in these habitats is likely to be important to understanding vegetative responses to long-term changes in climate and other stresses on the Chihuahuan Desert.

This small mammal assemblage is an important part of the food chain because of large numbers and wide distribution of small mammals in the ecoregion. Predators include raptors, ringtails (*Bassariscus astutus*), foxes (*Vulpes* spp.), snakes, coyotes (*Canis latrans*), bobcats (*Lynx rufus*), weasels (*Mustela* spp.), skunks (*Mephitidae* spp.), and many more opportunistic predators (Brown and Ernest 2002; Fitzgerald et al. 1994, Friggens et al. 2013). However, the suite of predators may be different for each species of small mammal, because of variation in the activities levels of the five species over the daily cycle: The deer mouse, hispid cotton rat, and southern plains woodrat are primarily nocturnal while the tawny-bellied and the yellow-nosed cotton rats are primarily diurnal. Since most predators are generalists, it is not well understood how changes in the suite of small mammals will affect predator abundance.

Management for this small mammal assemblage will be challenging because as conditions change, some species will decrease in abundance while others may increase in abundance. In fact, we can anticipate conflicting abundance trajectories for each species of the small mammal assemblage as the habitat changes. As a result, monitoring will require careful attention to the abundance of each species of the entire suite of small mammals because species richness or total small mammal abundance data may be misleading.

17.3 Grassland Small Mammal Assemblage Stressor Model

Table 17-2 presents and defines the drivers, critical environmental elements, critical ecological processes, and ecological outcomes represented in the grassland small mammal assemblage stressor model. The stressor model follows the methodology for species described in Chapter 4.

Table 17-2. Grassland small mammal assemblage definitions of stressor model components.

Model Component	Definition
Drivers	
Fire Management	Human activities to control the size, timing, intensity, or frequency of fires. Fire management applies to the control and prevention of wildfire as well as the use of prescribed fires to achieve management goals and objectives.
Grazing Management	Human activities to manage and control where and how domestic cattle are grazed within the Chihuahuan Desert. This can include the number of cattle grazed per hectare or other measures of grazing intensity as well as any grazing rotation strategies.
Human Uses and Urban & Industrial Growth	Human activity that converts native habitats (shrubland or grassland) into developed areas and includes barriers to movement, collisions with motor vehicles, attraction of domesticated predators, etc. Developed areas include urban, suburban, industrial, and agricultural development. Also refers to areas of intense recreational, military, borderland security, and other anthropogenic activities.

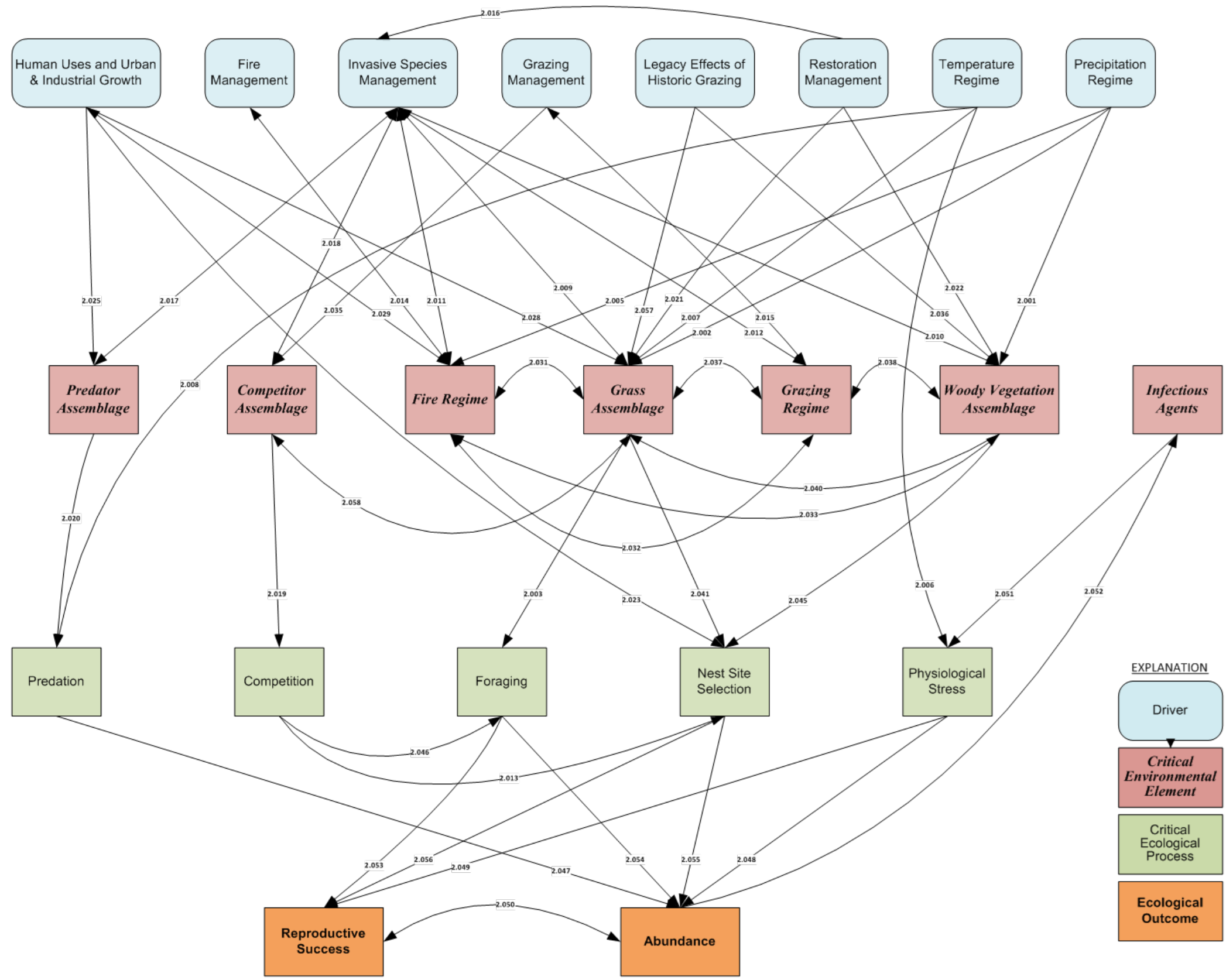
Model Component	Definition
Invasive Species Management	Human activities to reduce the abundance of invasive species in the Chihuahuan Desert. This includes two primary foci: exotic grasses, and native and exotic shrubs. This management may include mechanical removal, herbicide application, and native grass plantings.
Legacy Effects of Historic Grazing	Ongoing impacts to vegetation community composition and structure, especially increasing conversion of grasslands to shrublands, resulting from the legacy effects of unsustainable domestic cattle grazing in the 19th and early 20th centuries.
Precipitation Regime	The pattern, timing, intensity, and average annual rate of precipitation in the Chihuahuan Desert.
Restoration Management	Human activities including mechanical shrub removal, herbicide application and native grass plantings, which maintain or increase the percent cover of native grasses and/or reduce the percent of shrub cover in a specific region. This driver does not include invasive species management, fire management, or grazing management, which are treated separately in this framework.
Temperature Regime	The patterns of air temperature in the Chihuahuan Desert within and across all seasons.
Critical Environmental Elements	
Competitor Assemblage	The composition, abundance, and activity level of species that compete with small mammals for territory, food, nests, or other resources.
Fire Regime	The frequency, intensity, severity, and seasonality of both wildfire and prescribed fire.
Grass Assemblage	The composition and abundance of grass species.
Grazing Regime	The number of cattle grazed per acre, utilization, season of use, duration, and frequency of grazing. In the literature, grazing intensity, indicated by greater numbers of cattle and higher frequencies of grazing, is generally characterized as light, moderate, or heavy.
Infectious Agents	The presence and abundance of pathogens and parasites that can weaken or kill small mammals.
Predator Assemblage	The composition, abundance, and activity level of species that prey upon small mammals.
Woody Vegetation Assemblage	The composition and abundance of woody species, including shrubs as well as trees. Increased abundance can be associated with desertification.
Critical Ecological Processes	
Competition	The intensity with which small mammals contend with other species for territory, food, nests, and other resources.
Foraging	The rate of success of small mammals to actively find and consume food resources of suitable quality within their habitats.
Nest Site Selection	The frequency and extent to which the landscape provides a suitable amount and spatial arrangement of grassland, shrubland, and grass-shrub composite present in the Chihuahuan Desert for small mammals to successfully secure nests that provide adequate shelter and opportunity for reproduction.
Physiological Stress	The frequency and intensity of factors that negatively affect the health of small mammals.
Predation	The rate of killing small mammals by species such as snakes, raptors, bobcats, foxes, coyotes, domestic animals, etc.
Ecological Outcomes	
Abundance	The population size and spatial distribution of small mammals.
Reproductive Success	The rate with which small mammals successfully give birth and raise young to self-sufficiency.

Figure 17-1 shows the full grassland small mammal assemblage stressor model, displaying the model

components listed in Table 17-2 along with their causal relationships. Appendix 1 describes and presents the rationale for including every causal relationship in the stressor model, with citations for each causal relationship, providing a comprehensive presentation of the stressor model.

Figure 17-1 indicates the presence or absence of causal relationships between the system model components but does not indicate the potential magnitude or other characteristics of these relationships, as explained in Chapter 4. Figure 17-1 identifies the causal relationships that have affected how the system likely has changed in the past, in response to changes in its drivers. The diagram provides a means for articulating how the condition of the system will likely change in response to changes in its drivers. As discussed and illustrated below, the stressor model makes it clear: (a) which critical environmental elements would likely be affected by a change in one or more particular drivers, including change agents; (b) which critical ecological processes would likely be affected by the cascading effects of these changes in critical environmental elements; and (c) which system characteristics (ecological outcomes) would likely be affected by the cascading effects of these changes in critical environmental elements and ecological processes. By doing so, in turn, the stressor model also highlights those components of the model—drivers, critical environmental elements, critical ecological processes, and ecological outcomes—that demand indicator data.

Figure 17-1. Grassland small mammal assemblage full stressor model.



EXPLANATION

- Driver
- Critical Environmental Element
- Critical Ecological Process
- Ecological Outcome

A causal relationship exists when a change in one component of the system results in a change in some other component. Change in the first component is said to “cause a change in the second component. Each chain of causation, from driver to outcome, describes how the condition of the system likely has changed in the past, or likely would change in the future, in response to changes in its drivers.

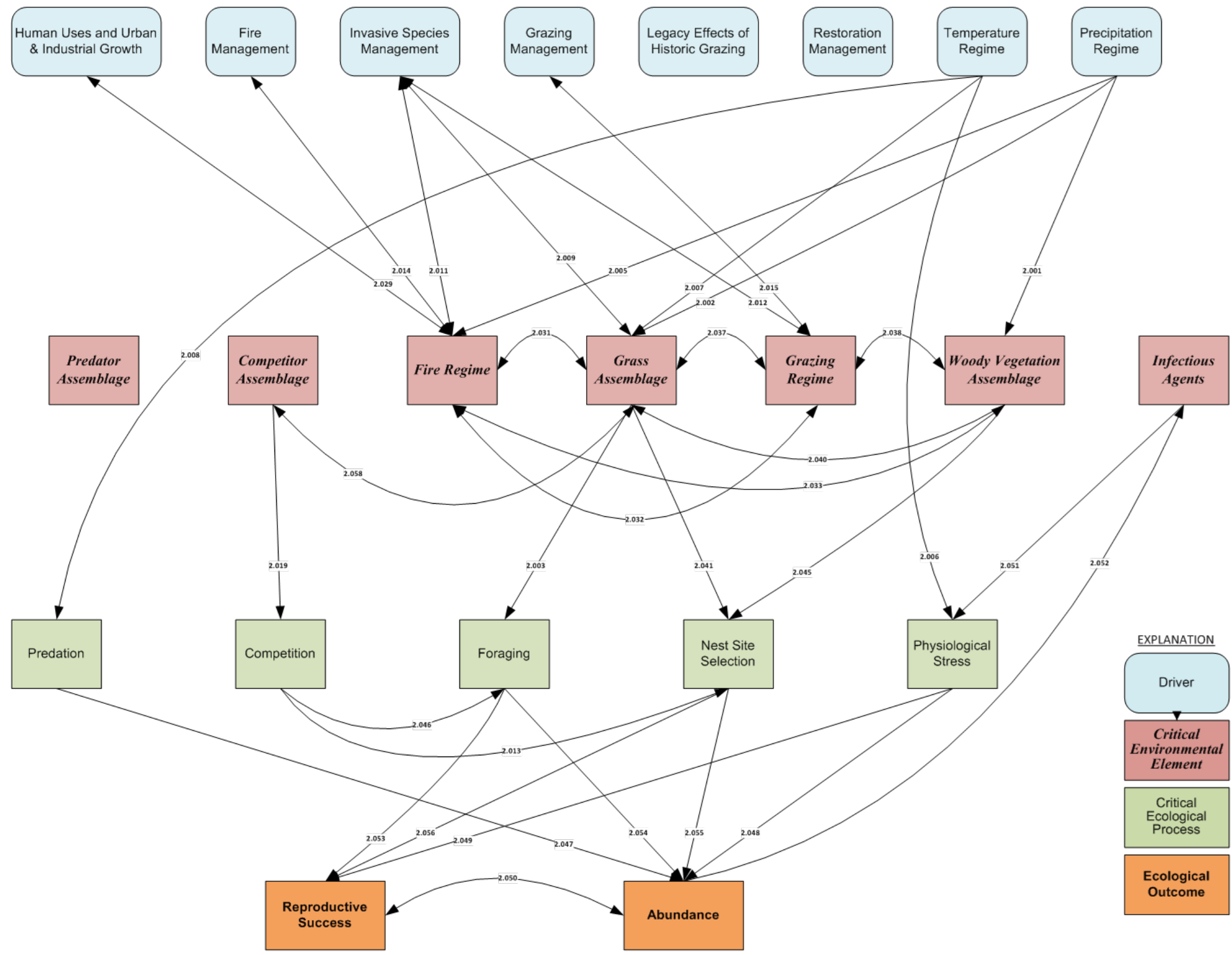
17.3.1 Climate Change

The final report for the Chihuahuan Desert REA will include a discussion of current forecasts of the ways in which climate change will affect the ecoregion and its ecological resources. The present chapter presents only the conceptual model of the causal relationships and outcomes that potentially will be affected for the grassland small mammal assemblage. Figure 17-2 presents the grassland small mammal assemblage stressor model for the U.S. portion of the ecoregion, simplified to show only those causal relationships that potentially will be affected by changes in the air temperature and precipitation regimes. Appendix 1 presents the rationale and citations for every causal link shown in the diagram.

Most climate change models predict that over the next 100 years the Chihuahuan Desert will become hotter and drier and experience more extreme weather events but also indicate that the response of arid lands to climate change will be strongly influenced by interactions with non-climatic factors (Archer and Predick 2008). The effects of a drier environment will most likely favor woody vegetation because of their deep roots and ability to reach moisture deeper in the soil (Backlund et al. 2008, White et al. 2011). This increase in woody vegetation will be at the expense of grassland. At some point, as conditions become drier, primary productivity will be reduced which will lower food resource availability, especially for granivores that feed primarily upon grass seeds (Kerley and Whitford 2000). However, those species that prefer shrub-dominated habitats (e.g., southern plains woodrat) may increase in number, but research is needed to determine this.

Brown and Ernest (2002) and Friggens et al. (2013) developed a vulnerability scoring system to assess the vulnerability of 117 vertebrate species that occur in the Middle Rio Grande Bosque to expected climate change. The purpose of their project was to guide wildlife managers on options and considerations for climate change adaptation. Included in their study were two of the species of interest for the Chihuahuan Desert REA. They conclude that the hispid and tawny-bellied cotton rats are expected to be resilient to phenology changes related to weather due to future climate change. The hispid cotton rat in fact lives well in agricultural, riparian and even in mesquite dominated areas. On the other hand, the tawny-bellied cotton rat may be more vulnerable because of its close association with tall, dense, undisturbed grasslands. Friggens et al. (2013) did not assess the other three species of interest. However, because the southern plains woodrat thrives in grasslands with shrubs and cactus we may expect this species to benefit from decreased precipitation and increased woody vegetation cover. The effects of climate change on the Chihuahua deer mouse and the yellow-nosed cotton rat may be negative with reduced grass cover but more research is needed. The yellow-nosed cotton rat may be vulnerable to changes in climate simply because it is already imperiled (in New Mexico), and changes to the grass assemblage may affect its nest site selection. The deer mouse overall should do well but it is unknown how the Chihuahua deer mouse subspecies will do since so little is known about it.

Figure 17-2. Grassland small mammal assemblage stressor model: Potential impacts of climate change.



Brown and Ernest (2002) show that both species richness and total rodent populations are correlated with mean annual precipitation. However they go on to show that fluctuations in rodent populations cannot be explained simply in terms of precipitation events (Brown and Ernest 2002). The ecological impacts of climate change on precipitation in the Chihuahuan Desert depend not only on how much precipitation falls, but also on the timing of the precipitation. Generally, most rain falls in the late summer and is associated with the summer monsoon, with another peak of precipitation in the winter. The timing of these rains is important. Winter rains tend to favor the woody vegetation assemblage (Neilson 1986) while summer rains tend to favor grasses (Brown et al. 1997). If the phenology of the precipitation changes, even if the amount of precipitation does not change, there can be profound effects on the plant community. Winter rains also favor the growth of woody invasive shrubs, as well as annual plants that uptake soil nutrients and reduce nutrient availability for warm season grasses that start growing later in the year (Burgess 1995, Neilson 1986). For example, creosote bush and mesquite are C₃ species best adapted to wet winters, while black grama grass is a C₄ species adapted to wet summers.

The phenology of precipitation also has profound effects on grazing management and fire management. Prescribed burns followed by a drought can lead to continued degradation of the grasslands (Ladwig 2014). Because precipitation is so difficult to predict, Lagwig (2014) suggests that fire managers should schedule prescribed burns only after precipitation events. In conditions where precipitation is higher, fire managers may use prescribed burns as a beneficial tool in maintaining the health of the grasslands. However, some people question the benefits of prescribed burns in these grasslands because fire-induced mortality of mesquite is low and fire has an adverse effect on black grama grasslands (Kilgore et al. 2009). This will be discussed in more detail under the fire management and fire regime section. Nevertheless, fire regime must be considered in context with precipitation patterns and managed based upon the amount and intensity of precipitation events.

Moisture and temperature conditions are also important drivers for disease (Kolivras and Comrie 2004), and it is well documented that a number of different infectious agents in the southwestern United States are linked to climatic factors including mosquito-borne viruses, sylvatic plague, hantavirus, and coccidioidomycosis (Glass et al. 2002, Kolivras and Comrie 2004). Savage et al. (2011) examined how environmental conditions, including temperature regimes and precipitation regimes, affect the persistence of both plague bacteria and their primary vector, fleas. Incidence of plague outbreaks is higher in years with warmer winters and cooler, wetter summers, likely because these conditions are more conducive to large flea populations. Flea populations are substantially reduced in colonies when air temperatures dip below 10 degrees Celsius or exceed 35 degrees Celsius. In most cases, moisture does not directly cause disease but increases in precipitation increase the food sources of the small mammals, and this in turn increases small mammal abundance (Parmenter et al. 1999). These increases in small mammal numbers provide a greater chance that disease will spread throughout populations, increasing the number of infected individuals. The model in Figure 17-2 shows precipitation affecting the grass assemblage and the effects of precipitation continue on to impact the model components of nest site selection, small mammal abundance, and infectious agents. It is not well understood how decreased

precipitation or changes in the phenology of precipitation will affect infectious diseases (Balbus and Wilson 2000)

Precipitation amounts and the phenology of precipitation also have profound effects on grazing management. Navarro et al. (2002) suggest a moderate, controlled amount of livestock grazing is sustainable on the Chihuahuan Desert rangelands receiving 26-35 cm of precipitation annually. However, during a drought, the amount of forage available to cattle is reduced and may lead to increased intensity of grazing to support existing herds (Kerley and Whitford 2000). When overgrazing is coupled with drought in the Chihuahuan Desert, woody vegetation increases (Brown et al. 1997). Grazing practices need to be monitored and adjusted as needed to help prevent the invasion of woody vegetation (Beck et al. 2007). The challenge with the small mammal assemblage is that monitoring indicators need to be carefully selected. Changes in precipitation phenology and amounts may differentially affect our species of interest. Habitat changes may benefit some species, be neutral to some, and yet be detrimental to others.

The potential effects of increased temperature on the small mammal assemblage will likely be complex and mainly be determined by how these temperature changes affect the plant community and overall primary productivity (Friggens et al. 2013, Throop et al. 2012). Most small mammals are well adapted to hot, dry environments but many arid-adapted endotherms already experience conditions at their physiological limits. It is conceivable that they could be negatively affected by warming temperatures (Moses et al. 2012), although Friggens et al. (2013) and Moses et al. (2012) did not find evidence that higher temperatures would negatively affect many nocturnal fossorial species. In the present case, the deer mouse, hispid cotton rat, and southern plains woodrat are mostly nocturnal. On the other hand, Moses et al. (2012) showed that even small increases in body temperature for the banner-tailed kangaroo rat (see Chapter 14) can be fatal and Macedo and Mares (1988) as well as Smith et al. (1998) feel that *Neotoma* spp. may already exist close to their upper threshold (i.e., lethal limit) of temperature in desert areas.

Smith and Betancourt (2006) used paleomidden data to conclude that woodrats respond to Bergman's rule: colder climatic conditions select for larger body size and warmer conditions select for smaller body size. Smith and Betancourt (2006) and Smith et al. (1998) suggest body size evolution is a likely outcome of climate change. They also suggest that such shifts are part of a normal spectrum of adaptation but may also have dramatic effects on woodrat life history and indirectly affect the dynamics and structure of the entire community. It is hard to predict how the entire small mammal assemblage will adapt to increased temperatures but it can be assumed that it will certainly add additional physiological stress to each of the species.

Elevated land surface temperature coupled with high aridity may cause plant material including seeds to desiccate, reducing available water in forage, and lead to negative water balance and reduced fitness (Alpert 2000, Schmidt-Nielsen 1964). This combination of increased temperature and reduced precipitation will also cause higher grass mortality from water stress and increase the percentage of bare ground within habitats, which will increase wind and water driven erosion (Backlund et al. 2008). This may cause increased fragmentation of habitat.

Increased temperature also may have several indirect negative effects on small mammals. Increased temperature may change the phenology of small mammal predators such as coyotes (Friggens et al. 2013, Moses et al. 2012) and snakes, increasing the time they are active and therefore, potentially the time they are able to prey on the small mammals (Moses et al. 2012). In addition, Brown and Ernest (2002) suggest that rodent populations are regulated not only from the bottom up by resource availability, but also from the top down by predation.

17.3.2 Uncharacteristic Wildfire

The fire regime of the U.S. portion of the Chihuahuan desert has changed as a result of the interaction of several drivers, as discussed in several previous chapters. Chapters 2 and 3 discuss the history, causes, and consequences of uncharacteristic wildfire across the ecoregion in general, and Chapters 5-7 discuss the causes and consequences of altered fire regimes specifically across the terrestrial systems of the region, in which the species members of the grassland small mammal assemblage live. Figure 17-3 presents the stressor model for the grassland small mammal assemblage in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by uncharacteristic wildfire.

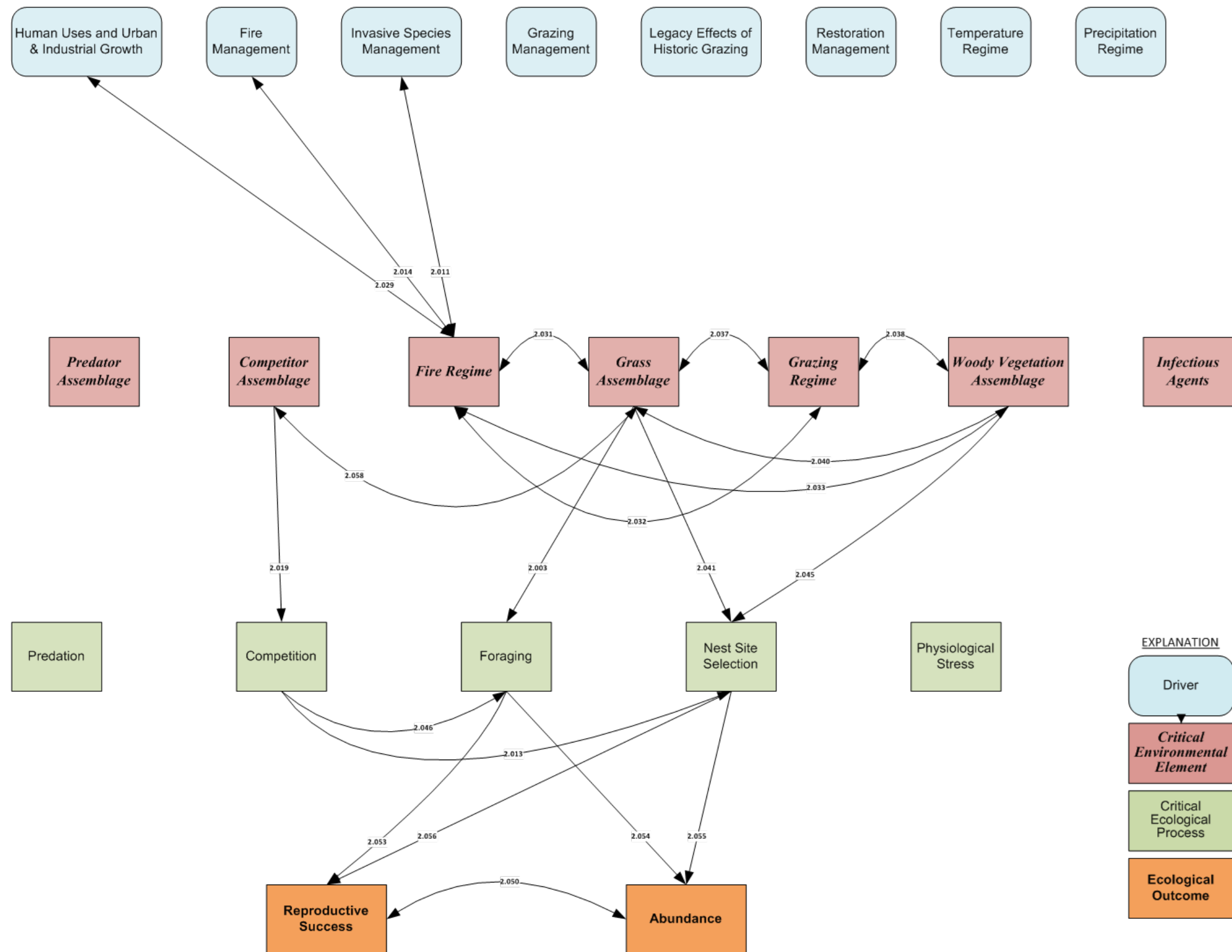
The historical importance of fire in the Chihuahuan Desert grasslands is not well understood. However, most investigators believe that fire suppression has significantly altered the vegetation of the ecoregion (Dick-Peddie and Alberico 1977, McPherson and Weltzin 2000, Webster and Bahre 2001, Kilgore et al. 2009). As the plant community changes, so does the fire regime. Starting in the late 1800s, overgrazing eliminated many of the grasses and other fine fuels required to support grassland fires, and as a result there have been fewer fires (Bahre 1991). Later, fire suppression and the establishment and growth of mesquite and other woody plants resulted in a shrub-dominated community over time that further reduced fires. Ultimately, fires became less frequent regardless of grazing pressure (Curtin et al. 2002, Drewa and Peters 2001) (see Chapters 2-3 and 5-7).

Grazing and invasive species both affect the fire regime in different ways. For example, areas where Lehmann lovegrass has become established are at greater risk of fire because lovegrass generates greater quantities of fine fuels than do native grasses (Anable et al. 1992, Cox et al. 1990). Grazing can reduce these fuel loads and decrease the risks of fires (Holechek et al. 1994). Prescribed burning has also been suggested as a method to prevent invasive plant encroachment on desert grasslands (McGlone and Huenneke 2004). However, this may not be an effective tool. Fire-induced mortality of mesquite is low, and Kilgore et al. (2009) found that fire has an adverse effect on black grama grasslands. In addition, suburban development has increased the liability of such efforts, where even a small number of homes make fire management effectively impossible at the landscape level (Curtin et al. 2002). Lastly, the timing and intensity of precipitation makes predicting the post-fire response of desert plant communities difficult. Grasses show a strong positive response to fire when accompanied by relatively high soil moisture and neutral response, or short-term decline, during drought (Curtin et al. 2002). Kilgore et al. (2009) sum it up when they conclude that the absence of significant changes in the native plant community gives little credence to the utility of prescribed fire as a positive management tool in the northern Chihuahuan Desert.

The direct effects of fire on small mammals vary between species and it is unclear if there is any long-

lasting effect on small mammal abundance. Bock and Bock (1984) showed a small decrease in overall small mammal abundance, but these reductions were short lived. The effects of fires seem rather dynamic because Bock et al. (1976) observed that most small mammal species recovered to pre-fire or control levels the second year after spring wildfires and the authors speculate that burning at approximately 5-year intervals and maintaining a mosaic of habitat may minimize the negative effects of fire on native flora and fauna (Bock and Bock 1984).

Figure 17-3. Grassland small mammal assemblage stressor model: Potential impacts of uncharacteristic wildfire.



Others suggest that small mammals such as *Neotoma* spp. that do not burrow may suffer much greater mortality from fire (McPherson 1995). In most cases, fires create a burned-unburned mosaic. This mosaic may partially be caused by the activities of small mammals themselves, especially *Dipodomys* spp., which produce relatively bare areas near mounds (Kilgore et al. 2009). In addition, many small fossorial mammals can hide in their underground nests during fires and may suffer low mortality rates (McPherson 1995). These factors may also contribute to the lack of a long-lasting fire effect on the rodent community, as Fitzgerald et al. (2001) and Valone et al. (2002) showed no significant differences in abundance or species richness on fire-treated plots and unburned plots, although there was some evidence that the species composition seemed to differ between treatments. These changes in small mammal community composition may be attributed to the food habits of these animals. For example, cotton rats feed largely on green vegetation, while most of the other species are chiefly seed predators (Baker 1971).

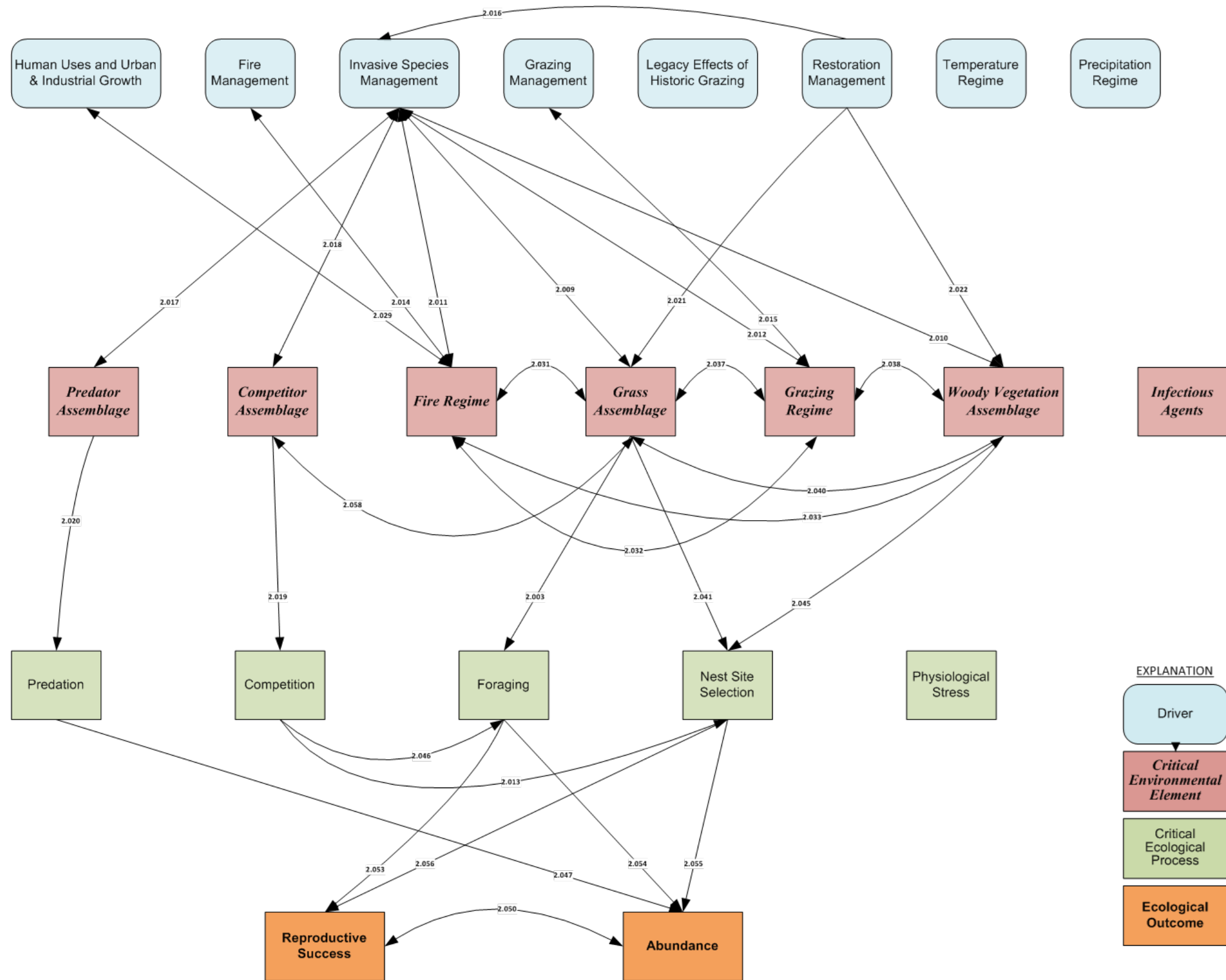
As mentioned above, abundance and even species richness may be maintained after fires. However, species composition may change. Very little research has been conducted on the effects of fires on small mammal communities. The most applicable research concerning the effect of fires on animal communities in the Chihuahuan desert was by Bock and Bock (1977) who explained the complexity of fires on bird habitat. While the topic of this document is not about birds, they show that fire can have two categorically different effects on desert grassland vegetation and these in turn can have very different effects on the bird community. In the short term, fire reduces grass cover for one to three post-fire growing seasons, while stimulating the abundance and variety of forbs, and generally increasing seed production (Bock and Bock 1977; 1992, Bock et al. 1976, McPherson 1995). Results of several studies in Arizona grasslands indicate that these short-term effects can improve habitat for seedeaters and open-ground species. At the same time, fire-caused reductions of grass cover temporarily reduce habitat quality for species dependent upon heavy ground cover. They explain that it is important to recognize that certain kinds of birds require or prefer unburned areas, even in ecosystems that have a long evolutionary association with fire. For every sparrow that depends upon the seeds produced by recently burned desert grassland, there is another that requires heavy grass cover that a fire temporarily destroys. The complex relationship between fire and the grassland small mammal assemblage may be presumed to be as complex. Some species will benefit by fire, while others do not fare as well. The development of effective measures of abundance will be critical in evaluating the effects of fire on small mammals.

17.3.3 Invasive Species and Landscape Restoration

Chapters 2-3 discuss the ecological consequences of non-native species across the U.S. portion of the ecoregion in general and Chapters 5-7 discuss the consequences of invasive species introductions specifically across the terrestrial ecological systems of the region in which the species members of the grassland small mammal assemblage live. Figure 17-4 presents the stressor model for grassland small mammal assemblage in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by non-native species and their management. Figure 17-4 also includes those causal relationships affected by landscape restoration that in turn affect the grassland small mammal assemblage in the U.S. portion of the ecoregion. As discussed in Chapters 2-3 and 5-7, much if not most

landscape restoration in the U.S. portion of the ecoregion is carried out to remove or control invasive plant species (see also Chapters 8-15). Appendix 1 presents the rationale and citations for each causal link shown in Figure 17-4.

Figure 17-4. Grassland small mammal assemblage stressor model: Potential impacts of invasive species, their management, and landscape restoration.



There are numerous introduced species in the Chihuahuan Desert as well as a number of native species that have increased dramatically in abundance. These species can be defined in two broad categories: grass assemblage and woody vegetation assemblage. As mentioned above, the relative proportion of grass to woody vegetation is often the most important habitat criterion affecting the composition and abundance of the grassland small mammal assemblage. Shrub habitat is usually associated with degraded grasslands and is used synonymously with desertification in many publications. Most models show increased shrub density as a negative influence on organism abundance. However, the results are somewhat mixed with small mammals. The management of invasive species is also part of restoration management discussed later in this section; in some cases, invasive species management can be synonymous with restoration management or landscape restoration.

More data are needed on how invasive plants affect the small mammal assemblage. Various studies indicate that species richness of rodents and other vertebrates is higher in shrub- and tree-invaded areas than in pure grasslands (Kerley and Whitford 2000, Lloyd et al. 2013). In studies of small mammals in the Jornada Basin, the black grama grasslands had the lowest mean rodent densities of any area and contained only six of the 12 species that were present in the other habitat types (Jones et al. 2003, McCulloch 1969). The highest species richness was in an area dominated by annual weeds. Jones et al. (2003) suggest that a landscape mosaic of grass and shrublands with varying amounts of ground cover, including some relatively dense grassland, likely will maintain the highest regional diversity of rodents. However, Whitford (1997) cautions that other measures of faunal biodiversity, such as keystone species may be better measures of the impact of desertification on animal biodiversity. For example, one species of small mammal that has been adversely affected by desertification is the banner-tailed kangaroo rat, discussed in Chapter 14. That species may be a keystone species in desert grassland ecosystems (Moorhead et al. 1988, Moroka et al. 1982, Schlesinger et al. 1990) (see Chapter 14).

17.3.3.1 Grass Assemblage

The five species of small mammals comprising the grassland small mammal assemblage can be found in desert grasslands historically dominated by black grama grass (*Bouteloua eriopoda*). In these grasslands, black grama cover can range from 44% in dry years to 75% in wet years (Paulsen and Ares 1962; Peters and Gibbens 2006). In 1858, good grass cover was present on more than 90% of the 58,492 ha studied. By 1963, less than 25% of the area had good grass cover (Havstad and Beck 1996). One of the many factors leading to the decline of these grasslands was changes in the grass assemblage caused by the numerous exotic and invasive species that have adversely affected these black grama grasslands. Some of the more notable non-woody species include Lehmann lovegrass (*Eragrostis lehmanniana*), African buffelgrass (*Pennisetum ciliare*), Red brome (*Bromus rubens*), *Eragrostis cilianensis*, *Erodium cicutarium*, *Fumaria parviflora*, *Salsola tragus*, and *Sisymbrium irio* (Van Devender et al. 2013).

Lehmann lovegrass seems to be the most prevalent and most studied of these exotic species (Crawley and Gray 1987, McClaran and Anable 1992, Paulsen and Ares 1962). As with many of the exotic grasses, Lehmann lovegrass was introduced and seeded from 1950-1980 to improve rangeland production and reduce erosion. While these exotic grasses may or may not have improved cattle forage, the lovegrass has displaced native grasses on large areas of the Chihuahuan desert, especially where soils have been

disturbed (Crawley and Gray 1987, Paulsen and Ares 1962). Bock and Bock (1977) showed a serious decrease in most native plant species and native animals in the presence of Lehmann lovegrass.

The most important effects of Lehmann lovegrass and other invasive species on small mammals may be in the interactions it has with other forces. Many of these nonnative plants change the fire regime because they are highly flammable and create more fine fuels than the grasses native to the Chihuahuan Desert (Brooks et al. 2004, Brooks and Pyke 2001, McGlone and Huenneke 2004). Lovegrass is highly tolerant of fire, as this species creates deep roots and fire stimulates seed production. Because they increase fine fuel loads, they can significantly alter the fire regime by increasing the potential for wildfire. They also recover relatively quickly after fire, thereby altering fire frequency (Brooks et al. 2004, McPherson 2006). This may, in turn, pose a risk of generating a positive-feedback cycle enhancing both the frequency of fires and the expansion of Lehmann lovegrass (Anable et al. 1992), although McGlone (2013) did not find evidence for this in his study. By contrast, members of the grass and forb assemblages may take many years post-fire to recover, and that recovery is slowed if precipitation is below normal.

Exotic plant species often flourish in disturbed areas. Human activities and grazing are often the primary sources for disturbance. Mack (1985) suggests that disturbance may be necessary only for initial establishment, but not for subsequent spread, of lovegrass. In established Lovegrass populations, density tends to increase over time but does not seem to be affected by different grazing regimes. Grazing seems to disproportionately affect native grasses because native grass density decreases while lovegrass relative abundance increases with time and increased grazing intensity. This is most likely a result of preferential grazing by the cattle. Interestingly, lovegrass density and relative abundance do not differ between adjacent ungrazed and grazed areas, and livestock grazing was not necessary for Lehmann lovegrass to spread (McClaran and Anable 1992).

The effects of additional exotic grass species beyond Lehmann lovegrass to the grass assemblage are not well understood. One area of particular concern is foraging. Small mammals may forage stems and use seeds from these exotic species as food. However, little is known about this. In addition, the effects on the small mammal assemblage due to changes in the grass assemblage from perennial grasses to annual grasses are not well understood although it may be presumed that small mammals favor native perennial plants grasses (Monson 1943). Most of these small mammals depend on tall dense grass for cover and food. Any change to this would be detrimental to small mammals (Hoffmeister 1986).

One area of research that may be considered is the effect the small mammal assemblage on the grass assemblage and plant community as a whole. Indirect evidence of changes in plant community dynamics have been shown in experiments removing banner-tailed kangaroo rats. In one study, tall stature perennial and annual grasses colonized open spaces between shrubs and increased approximately threefold in the absence of kangaroo rats. Much of this response can be attributed to two species: Lehmann lovegrass, which increased more than 20-fold, and the annual, *Aristida adscensionis*, which increased approximately threefold (Brown and Heske 1990). It can be inferred that banner-tailed kangaroo rats maintain different plant species at different population densities through preferential foraging. Similarly, Kerley and Whitford (2000) found that elevated levels of rodent granivory influences

the lack of recovery of degraded desert grasslands and plant regeneration. It has been observed that small mammal presence in severely overgrazed areas can almost eliminate herbaceous vegetation, especially during drought and grass recovery can occur rapidly with the removal of small mammals (Norris 1950, Vorhies and Taylor 1993).

17.3.3.2 Shrub Density

The most significant factor affecting the Chihuahuan grasslands and small mammals is the invasion of woody species into these grasslands. During the past 150 years, low woody vegetation has substantially increased (Van Auken 2000). Within the woody vegetation assemblage, the desert shrubs that seem to have the most impact on the small mammal assemblage include, but are not limited to, creosote bush (*Larrea tridentata*), mesquite (*Prosopis* spp.), broom snakeweed (*Gutierrezia sarothrae*), tarbush (*Flourensia cernua*), and whitethorn (*Acacia neovernicosa*) (Anderson and Kay 1999, Krogh et al. 2002). Many of these woody plants are native species that have increased significantly in abundance and geographic distribution. Grasses historically dominated the areas where small mammals are found, even though shrubs and other woody plants may have been present—but in low densities (Johnston 1977, Van Auken 2000). Grasslands that have suffered drought, livestock overgrazing and altered fire regimes have been invaded by woody vegetation (Gao and Reynolds 2003, Yanoff and Muldavin 2008). The once primarily dominant black grama grass is now on 1% or less of the area, as found in one study at the Jornada Basin, and has been completely extirpated from many areas (Gibbens et al. 2005). One reason for the success of woody vegetation is its ability to outcompete grasses during periods of drought or extreme rain events (Gao and Reynolds 2003, Gibbens et al. 2005, Schlesinger et al. 1990). With increasing numbers of low-value shrubby plants on semi-desert grass-shrub ranges, the grasses inevitably decline and are unable to regain dominance of the site even when rainfall is not limiting (Paulsen and Ares 1962). Shrub encroachment into the desert grasslands of the Chihuahuan Desert is still occurring (Grover and Musick 1990) and shrub encroachment is recognized as the main feature of desertification in this region (Krogh et al. 2002).

Furthermore, the cumulative effects of severe encroachment by woody species on an ecosystem can drive the system into an alternative steady state that may be difficult or impossible to resolve to its historical community type (Schlesinger et al. 1990). These changes to the grassland communities have encouraged much effort devoted to restoring these degraded grasslands. However, as with all invasive species, it is very hard to undo an introduction and subsequent invasion. The best approach going forward may be to contain the spread of the plants. Efforts to minimize soil disturbance from OHVs, cattle, and even cross-border traffic may mitigate the spread of some species. Research on the effects of various grazing regimes as well as fire regimes may be warranted, as they most likely affect different species in different ways.

New animal introductions also can affect grassland small mammals, changing the competitor assemblage, increasing competition, changing the predator assemblage, and increasing predation. Examples of these new species in the ecoregion include the house mouse (*Mus musculus*), the Norway rat (*Rattus norvegicus*), feral cats, and feral dogs. These four species are closely associated with humans and are discussed in more detail in the section on development, below, this chapter. However, in all

cases, these new species have a negative influence on small mammal reproductive success and small mammal abundance.

17.3.3.3 Restoration Management

Landscape restoration treatments have been applied to thousands of hectares in the Chihuahuan Desert. These revegetation projects typically focus on reestablishing grass for soil conservation and livestock forage rather than restoring small mammal habitat. Unfortunately, efforts to improve forage for cattle and reduce woody vegetation may not be beneficial to many small mammals. As mentioned before, shrub-invaded grasslands contain higher small mammal richness, biomass, and abundance compared to native grasslands (Kerley and Whitford 2000, Lloyd et al. 2013, Whitford 1997)

Restoration efforts that focus on returning shrub-invaded habitat back to grassland habitat may use fire, chaining, and herbicide to push the community succession. Treatments applied at intense levels severe enough to kill shrub roots may transition a shrub-dominated state back to a grassland state, although the grasses may not all be native grasses (Hemstrom 2014). Less severe treatments do not kill shrub roots and only set the plant community back to a grassland state with live shrub roots remaining that can resprout (Hemstrom 2014). This leads to a faster return of shrub cover and a need for more frequent treatments. Many of these restoration projects relied on non-native grasses, even though many native grasses produce highly germinable seed. In many cases, the native grasses have not established as well as non-native grasses, and the application of treatments sometimes disturbs the soil and inadvertently increases non-native or woody species (Roundy and Biedenbender 1995). It is not well understood how the species in the small mammal assemblage are able to use many of the exotic grasses as forage.

Attempts to improve plant community composition have not always been successful. External factors such as precipitation amounts and precipitation phenology may have more to do with the increase or decrease in densities of particular species of invasive shrubs than do any restoration activities. In addition, experience has shown that shrub-dominated ecosystems are very resistant to attempts to restore them back to grasslands (Kerley and Whitford 2000). In many cases, revegetation success is largely determined by the pattern of summer precipitation in a given year (Roundy and Biedenbender 1995). Ultimately, because of the complexity and cost of landscape level restoration, true grassland restoration will probably be confined to small areas and will mainly serve educational and research purposes (Roundy and Biedenbender 1995). These small-scale efforts may not be substantial enough to change small mammal abundance in any significant way. Eve et al. (1999) present an even more pessimistic view, stating that the probability that monoculture creosote bush ecosystems or mesquite coppice dune ecosystems will recover or be restored is virtually zero.

More recently, Cosentino et al. (2014) showed that large-scale herbicide treatments started more than 30 years ago indicate grassland restoration has been generally effective for increasing banner-tailed kangaroo rat abundance (see Chapter 14). Cosentino et al. (2014) discuss the Restore New Mexico program ("Restore New Mexico," 2015), which has an explicit goal to restore grasslands and grassland-dependent wildlife. They note that improving local habitat quality by reducing shrub cover is a critical first step to recovery of banner-tailed kangaroo rats. However, their data showed that recovery of

banner-tailed kangaroo rat populations can take decades even when shrub cover is reduced, and they attribute this time lag primarily to limited dispersal and inadequate connectivity to source populations. Similar research is needed on how these restoration efforts affect other small mammals, and if they have similar long lag times before measurable results are obtained.

17.3.4 Development

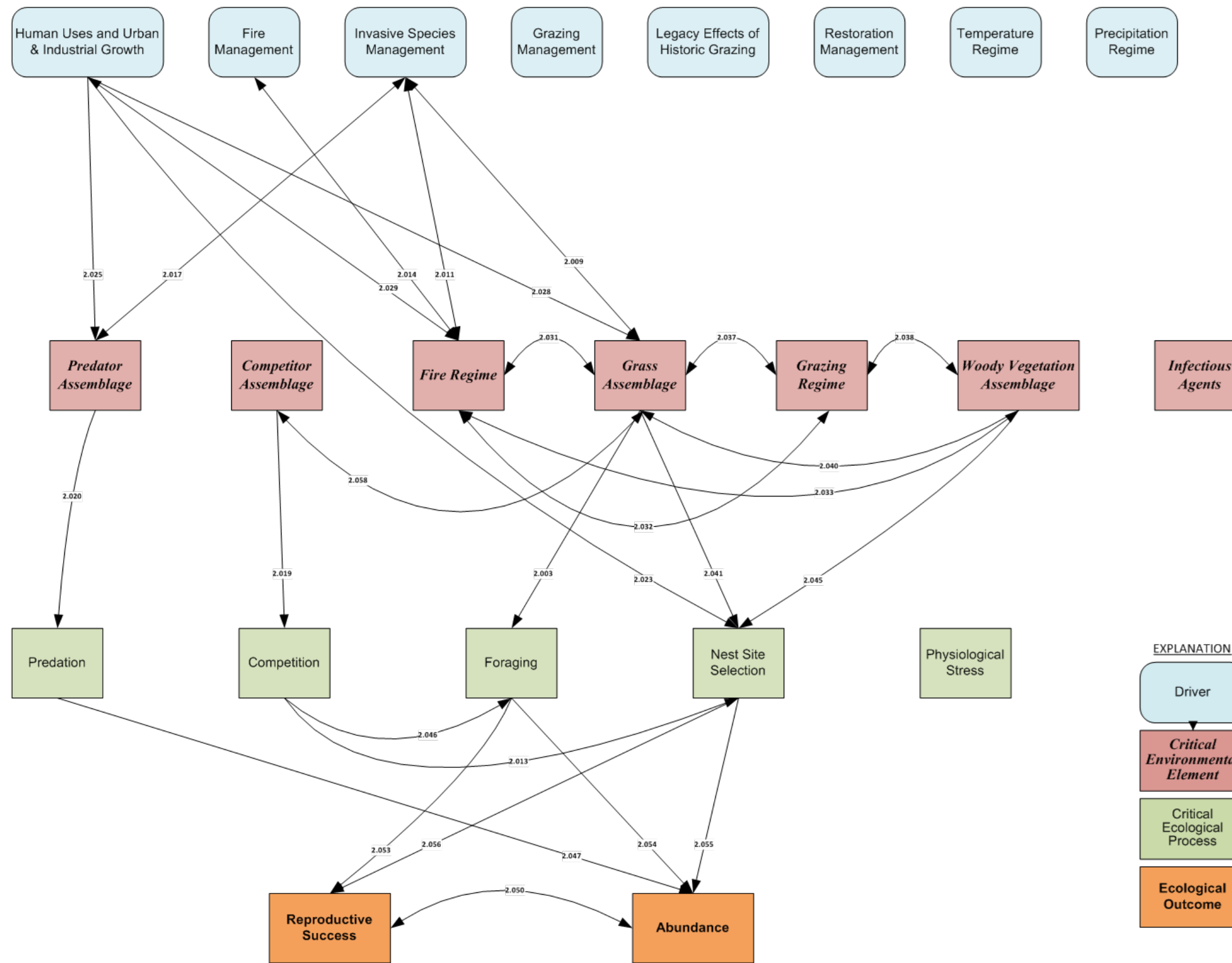
Chapters 2 and 3 discuss the history and consequences of land development across the ecoregion in general, and Chapters 5-7 discuss these consequences specifically across the terrestrial ecological systems of the region, in which the species members of the grassland small mammal assemblage live. Figure 17-5 presents the stressor model for the grassland small mammal assemblage in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by land development. See Appendix 1 for a full presentation of the stressor model.

The impacts of land conversion (including expanding urban, suburban and industrial conversion), energy extraction, and agricultural development on small mammal habitat can be severe and less repairable than the impacts of other change agents (Archer and Predick 2008, Curtin et al. 2002, Havstad and Coffin Peters 1999, Stacey and Post 2009). Although more limited spatially, and more recent than ranching, subdivision and suburbanization is rapidly expanding and altering large areas in the Chihuahuan Desert (Archer and Predick 2008, Curtin et al. 2002). It can be assumed that this suburbanization will continue to occur as land value for real estate development can be 4-100 times the value of ranching (Curtin et al. 2002), and as these grasslands are developed, large areas of habitat are lost. Fortunately, a substantial portion of these lands are owned by federal and state governments and have lower direct threats from development.

Land conversion inevitably leads to increased human presence and human activities. Maintenance of roads and rights-of-way as well as recreational OHV use will have the potential to seriously harm small mammals. Joule and Cameron (1975) reported that a 15m paved highway restricted crossing of cotton rats and Swihart and Slade (1984) showed that even small unimproved rural roads inhibit hispid cotton rat movement. Mowed areas also caused hispid cotton rat population declines (Goertz 1964). However, Stacey and Post (2009) detected a richer than expected community of small mammals at roadside sites. Increased traffic can also have negative indirect effects such as impacting the soil and can lead to an increase in woody vegetation and other invasive plants (Whitford and Bixby 2006).

More people living in close proximity to small mammals also means the addition of exotic species associated with humans. Examples of these new species include the house mouse, the Norway rat, feral cats and feral dogs. The addition of domesticated dogs and cats into the predator assemblage leads to increased predation and subsequent drop in small mammal abundance (Archer and Predick 2008). Addition of the house mouse and Norway rat can increase competition for food and increase the chance for diseases ("Diseases directly transmitted by rodents," 2011). However, some native species also seem to do well with human-altered landscapes. Deer mice have been shown to live in close proximity of humans and live in both used and unused human structures (Hoffmeister 1986), and tawny-bellied cotton rats are often found associated with agricultural areas (Cahalane 1954, Savage et al. 2011).

Figure 17-5. Grassland small mammal assemblage stressor model: Potential impacts of development.



Increased human presence via land conversion, as well as associated human activities, typically result in changes to the fire regime. Fragmented landscapes may not burn as often or as hot as native grasslands. Furthermore, more effort will need to be made to protect people and structures, often at the expense of protecting good habitat. Under existing institutional arrangements, it takes only a small number of homes to render fire management effectively impossible at the landscape level (Curtin et al. 2002). It is very difficult to undo suburbanization, agricultural development, or other human-induced land conversion. In most cases, prevention or management of this development is the best strategy.

17.3.5 Excessive Domestic Grazing

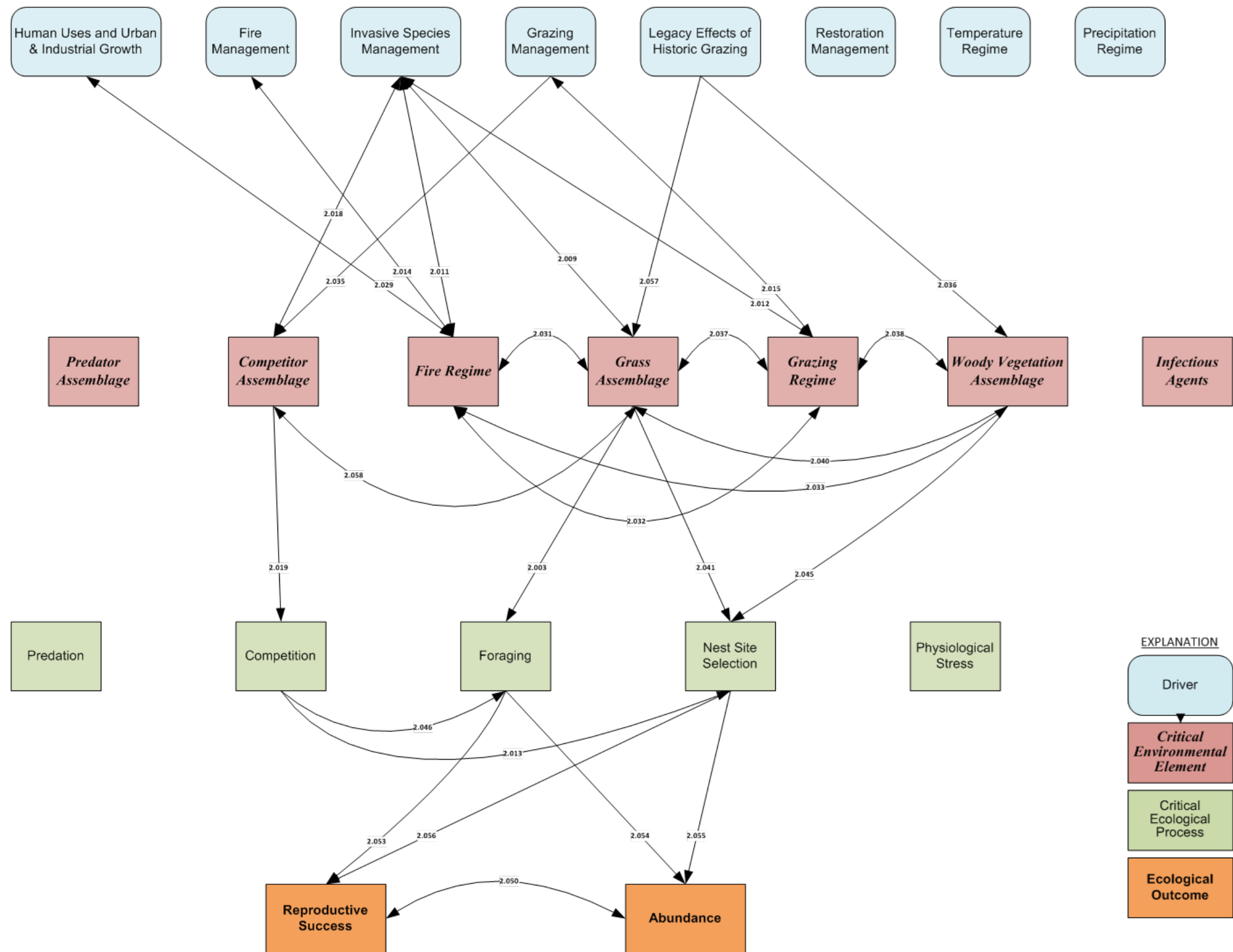
Chapters 2-3 discuss the ecological consequences of excessive domestic grazing across the U.S. portion of the ecoregion in general and Chapters 5-7 discuss the history and consequences of livestock grazing specifically across the terrestrial ecological systems of the region in which the species members of the grassland small mammal assemblage live. Figure 17-6 presents the stressor model for the grassland small mammal assemblage in the U.S. portion of the ecoregion, simplified to show only those causal relationships affected by excessive domestic grazing and its management. Appendix 1 presents the rationale and citations for each causal link shown in Figure 17-6.

Livestock grazing has been, and continues to be, an important land use in the Chihuahuan Desert, and the legacy effects of this grazing have been catastrophic for the Chihuahuan Desert. Much of the environmental degradation caused by livestock occurred right after Anglo settlement (Curtin et al. 2002, McPherson and Weltzin 2000). Grazing in this area has been boom-and-bust for the past 150 years, alternating between extremely large cattle and sheep herds while the market was good and subsequent busts when environmental and market conditions caused widespread failure in ranching. Unfortunately, these boom and bust cycles were also associated with extreme environmental damage from overgrazing associated with drought (Curtin et al. 2002).

The manifestation of the damage caused by grazing is in the desertification of the grasslands and associated invasion of woody plants (Bahre and Shelton 1993). This desertification causes a cascade of interactive effects changing the fire regime, grass assemblage, woody vegetation assemblage, soil characteristics, and soil moisture-holding abilities (Bahre and Shelton 1993). The legacy effects of grazing on small mammals is not well understood but rodents that prefer relatively tall and dense grass cover have been the most negatively affected by livestock grazing, especially pygmy mice, harvest mice, and cotton rats (Jones et al. 2003). Grazing also has caused large scale declines in other species such as the black-tailed prairie dog (Miller et al. 1994) and the banner-tailed kangaroo rat (Krogh et al. 2002, Waser and Ayers 2003). The increase in woody vegetation associated with grazing may have benefited some small mammals such as the southern plains woodrat (Birney 1973; Curtin et al. 2002, Finley 1958); however, cattle eating *Opuntia* actually reduced southern plains woodrat populations in southern Texas (Raun 1966).

While many advocate for removal of cattle from the grasslands, some think that reducing or eliminating livestock now will not cause the land to heal (Curtin et al. 2002). Excluding livestock grazing from most sites now will have little to no impact on the abundance of woody plants or non-native herbs during the next several decades (McPherson and Weltzin 2000). Curtin et al. (2002) feel that climate, substrate, evolutionary history, and other disturbance factors are often more important in determining vegetation response than the number, or presence, of livestock.

Figure 17-6. Grassland small mammal assemblage stressor model: Potential impacts of excessive domestic grazing.



Overall, Heske and Campbell (1991) determined that most of the long-term trends in rodent abundance, observed during the 12 years of rodent censuses, appear to be independent of cattle exclusion. On the other hand, Jones et al. (2003) indicate that multiple factors such as precipitation, grazing, and fire are important factors in grasslands and these three forces affect rodent communities through their impacts on ground cover of grasses, forbs and low shrubs. This is not to suggest that grazing is benign with respect to these grasslands. In another study, Jones (2000) revealed that grazing is generally unfavorable for rodent communities in arid grasslands. The only exception to this may be the deer mouse which is generally more abundant on grazed sites (Hanley and Page 1982, Heske and Campbell 1991, Larrison and Johnson 1973). However, the deer mice in this study were not likely the Chihuahua deer mouse that is a focus of this paper. Heske and Campbell (1991), in studies excluding cattle from certain areas, found a modest negative effect of cattle grazing on banner-tailed kangaroo rats, and they found that livestock can negatively impact rodent populations directly by trampling burrows and compacting soil or harvesting seed heads and other plant parts while grazing, thus removing resources that otherwise would be available to rodents. Jones et al. (2003) found that four species of *Muridae* (*Sigmodon fulviventer*, *Baiomys taylori*, *Reithrodontomys megalotis*, and *R. fulvescens*) were significantly more common on ungrazed plots, while no species was more abundant on grazed plots.

Other earlier researchers have reported both positive and negative responses of rodents to livestock grazing, depending on the particular species of rodent (Linsdale 1946, Reynolds 1950, Whitaker 1996). Rodent species richness, total abundance, and mean rank abundance have been shown to be higher in ungrazed than in grazed areas, independent of exurbanization or habitat, although there was a marginally significant three-way interaction among these variables for species richness (Brooks et al. 2004, Jones et al. 2003, McClaran and Anable 1992).

These mixed results are understandable and common in studies of small mammals. Mathis et al. (2006) showed a similar variable response. Diversity, richness, biomass, and capture rates of small mammals showed no consistent response to shrub removal and grazing treatments. He cited other studies that have experienced similar variable responses (Heske and Campbell 1991, Valone et al. 2002). Mathis et al. (2006) suggest that this environmental stochasticity suggests the need for long-term, replicated experimental studies. They suggest that even though no pattern may be evident, this does not imply that no pattern exists.

Grazing management that will mitigate the harm done to small mammals by cattle and sheep will be complex. Natural variation in year-to-year rodent abundance, diversity and biomass may confound the effects of grazing on small mammal populations. In addition, it appears that different small mammal species react differently to grazing. Maintaining the grasslands in such a way as to favor small mammals will require careful, long term studies and monitoring in order to illuminate the effects of management actions regarding grazing.

17.4 Grassland Small Mammal Assemblage Key Ecological Attributes

All ecological outcomes and critical ecological processes in the grassland small mammal assemblage stressor model constitute the key ecological attributes for the CE. The list below identifies seven key ecological attributes for the grassland small mammal assemblage based on these criteria. *Fully characterizing the present condition of the CE will require data on indicators for its key ecological attributes.* Indicators are

determined during Phase II of the REA process. The definitions for the key ecological attributes are the same as the definitions for these model components presented above.

- **Ecological Outcomes**
 - Abundance
 - Reproductive Success
- **Critical Ecological Processes**
 - Competition
 - Foraging
 - Nest Site Selection
 - Physiological Stress
 - Predation

18 Synthesis of Assessment Priorities

Bureau of Land Management (BLM) Rapid Ecoregional Assessments (REAs) seek to provide information to natural resource managers concerning (a) ecoregional-scale ecological conditions and trends, (b) the major factors that shape these conditions and trends, and (c) opportunities to conserve ecological resources across administrative boundaries. The REA approach integrates diverse sources of information needed to support conservation, restoration, and the development of ecological management programs in a cohesive manner.

The present Pre-Assessment report for the Chihuahuan Desert REA identifies the Conservation Elements, Change Agents, and Management Questions on which to focus the REA. Additionally, this Pre-Assessment report presents conceptual ecological models for all Conservation Elements. These conceptual models identify potentially measurable key ecological attributes for each Conservation Element, document present understanding of how each Change Agent may affect each Conservation Element, and provide a means for translating the Management Questions into terms specific to each individual Conservation Element and/or Change Agent.

The second or Assessment phase of the Chihuahuan Desert REA, reported separately, builds directly on the information and priorities established during the Pre-Assessment phase of work. The Assessment uses existing geospatial data and publications to map the distribution of the Conservation Elements and, where feasible, assess the condition of these Conservation Elements, assess the impacts of the Change Agents, assess possible future impacts of Change Agents where appropriate, and address the key Management Questions raised during the Pre-Assessment phase of work.

18.1 Chihuahuan Desert REA Conservation Elements

The Chihuahuan Desert REA identified fourteen Conservation Elements for assessment. These consist of three dry (terrestrial) ecological system types, five wet (aquatic-wetland) ecological system types, and four individual species and two assemblages of species of management concern associated with terrestrial ecological systems. One of the aquatic-wetland CEs, “Playas and Playa Lakes,” has both wet (inundated) and dry phases, and thus shares features with both wet and dry system types.

The Chihuahuan Desert REA Conservation Elements are as follows:

Dry-System Conservation Elements

- Chihuahuan Desert Grasslands
- Chihuahuan Desert Scrub
- Pinyon-Juniper Woodlands

Wet-System Conservation Elements

- Montane-Headwater Perennial Streams
- Lowland-Headwater Perennial Streams
- Large River-Floodplain Systems
- Springs-Emergent Wetlands
- Playas and Playa Lakes

Species and Species Assemblage Conservation Elements

- Pronghorn
- Mule Deer
- Banner-tailed Kangaroo Rat
- Black-tailed Prairie Dog
- Grassland Bird Assemblage
- Grassland Small Mammal Assemblage

18.2 Chihuahuan Desert REA Change Agents

The Chihuahuan Desert REA elected to assess the current distributions and status of six Change Agents, and to forecast the future distributions and status of two of these six, as follows:

Change Agents to be Assessed for Current and Forecasted Conditions

- Climate Change
- Development

Change Agents to be Assessed for Current Conditions Only

- Excessive Grazing
- Uncharacteristic Wildfire
- Invasive Species
- Landscape Restoration

18.3 Chihuahuan Desert REA Management Questions

All REAs, including the Chihuahuan Desert REA, address four basic Management Questions concerning: the geographic distribution of each Conservation Element; how the condition of each Conservation Element varies across its geographic distribution; the geographic distribution of each Change Agent and the relationship of that distribution to the distributions and conditions of the Conservation Elements; and the forecasted future geographic distributions of impacts of those Change Agents for which forecasts are available. Table 1-1, in Chapter 1, above, and Table 3-3, in Chapter 3, above, list these four basic MQs, designated MQ A – MQ D, and indicates the Conservation Element(s) and Change Agent(s) to which each question applies.

The Chihuahuan Desert REA also developed thirteen additional Management Questions focused on management concerns of regional importance. Table 1-1, in Chapter 1, above, and Table 3-3, in Chapter 3, above, also list these thirteen additional Management Questions and indicate the Conservation Elements and Change Agents to which they apply. The thirteen additional Management Questions are as follows:

- 1 Where have restoration treatments been applied to dry-system Conservation Elements, and what is the status (e.g., success rate) of those treatments?
- 2 What is the geographic distribution of the Chihuahuan desert amphibian assemblage?
- 3 Where would uncharacteristic wildfire likely increase sedimentation and loss of habitat among the wet systems?
- 4 What areas of potential black-tailed prairie dog habitat would support restoration?

- 5 Where are the areas of greatest faunal species biodiversity among the species and species-assemblage Conservation Elements taken together?
- 6 Where will urban and industrial growth impact intact grasslands or impede their recovery?
- 7 How do the current and historic geographic distributions of the dry-system Conservation Elements differ?
- 8 How will urban and industrial growth alter the geographic distribution of the grassland bird assemblage?
- 9 What and where are the aquifers and their recharge zones that support the wet systems?
- 10 How do the current and historic geographic distributions of the Pecos River and Gila River fish assemblages differ?
- 11 Where are the breeding, winter, and year-around habitats for pronghorn and mule deer?
- 12 Are there areas where invasive plants are being killed on a broad scale (e.g., by the tamarisk leaf-eating beetle) where managers need to focus on restoration or controlling succession?
- 13 What is the current geographic distribution of the impacts of gypsum in the soil and water, in general and in relation to each Conservation Element and Change Agent?

18.4 Chihuahuan Desert REA Conceptual Ecological Models

The Chihuahuan Desert REA also developed conceptual ecological models for all fourteen Conservation Elements. These models show how the Change Agents may affect each Conservation Element and provide a means for translating Management Questions into terms specific to each individual Conservation Element and/or Change Agent. Overarching “dry system” and “wet system” conceptual models provide a hierarchical framework for organizing and integrating the conceptual models for the individual Conservation Elements, following the recommendations of Miller et al. (2010). The conceptual models for the individual Conservation Elements differ in their methodologies.

The conceptual models developed for the Chihuahuan Desert REA identify potentially measurable “key ecological attributes” for each resource that managers can use to monitor or assess resource condition when appropriate geospatial data are available. These key ecological attributes include defining physical, biological, and ecological characteristics of a Conservation Element, along with its abundance and/or spatial distribution. The lists of key ecological attributes developed for the Chihuahuan Desert REA Conservation Elements guided the search for geospatial datasets with which to assess the distribution and condition of the Conservation Elements during the Assessment phase of the REA.

19 Literature Cited

- Abelho, M., and M. C. Molles, Jr. 2009. Effect of introduced exotic tree litter on consumption patterns of the introduced exotic isopod *Armadillidium vulgare*. *European Journal of Soil Biology* 45:306–311.
- Abella, S.R. 2009. Post-fire plant recovery in the Mojave and Sonoran Deserts of western North America. *Journal of Arid Environments* 73:699–707.
- Adkins, R.N., and L.A. Harveston. 2007. Demographic and spatial characteristics of feral hogs in the Chihuahuan Desert, Texas. *Human-Wildlife Interactions* 1:152–160.
- AECOM. 2011. Hydrological Assessment and Groundwater Modeling Report for the HB In-Situ Solution Mine Project EIS. Ft. Collins, CO.
- Agency Technical Workgroup 2005. Potential effects of climate change on New Mexico. State of New Mexico.
- Alcala-Galvan, C.H., and P.R. Krausman. 2013. Home range and habitat use by desert mule deer in altered habitats. *California Fish and Game* 99:65–79.
- Alcala-Galvan, C., and P.R. Krausman. 2012. Diets of desert mule deer in altered climates in the lower Sonoran Desert. *California Fish and Game* 98, 81–103.
- Alderete, J.A. 1996. Vegetation changes following a prescribed burn on piñon-juniper stands in south-central New Mexico (Thesis). New Mexico State University, Las Cruces, NM, USA.
- Allan, J. D. 1995. *Stream Ecology: Structure and Function of Running Waters*. First. Chapman & Hall, New York, NY.
- Allan, J. D. 2004. Influence of land use and landscape setting on the ecological status of rivers. *Limnetica* 23:187–198.
- Allen, B.D. 2005. Ice Age lakes in New Mexico. In: S. G. Lucas, G. S. Morgan, and K. E. Zeigler (Eds.), *New Mexico's Ice Ages*, Bulletin 28. Albuquerque, NM: New Mexico Museum of Natural History and Science, pp.107–114.
- Allen, C.D. 1989. Changes in the landscape of the Jemez Mountains, New Mexico (Dissertation). University of California, Berkeley.
- Allen, C.D. 2001. Runoff, erosion, and restoration studies in pinyon-juniper woodlands of the Pajarito Plateau. Pages 24-26. In: Johnson, P.S. (Ed.), *Water, Watersheds, and Land Use in New Mexico: Impacts of Population Growth on Natural resources—Santa Fe Region Decision-Makers Field Guide*. New Mexico Bureau of Mines and Mineral Resources, Socorro, NM, USA.
- Allen, C.D., and D.D. Breshears. 1998. Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. *Proceedings of the National Academy of Sciences, U.S.A* 95, pp. 14839-14842.
- Alley, W.M. (Ed.) 1993. *Regional Ground-Water Quality*. Van Nostrand Reinhold, New York, NY.
- Allred, K.W. 1993. *A field guide to the grasses of New Mexico*. New Mexico Agricultural Experiment Station, New Mexico State University, Las Cruces, NM, USA.
- Allred, K.W. 1996. Vegetative changes in New Mexico rangelands. *New Mexico Journal of Science* 36:168–231.
- Alò, D., and T. F. Turner. 2005. Effects of habitat fragmentation on effective population size in the endangered Rio Grande silvery minnow. *Conservation Biology* 19:1138–1148.

- Alpert, P. 2000. The discovery, scope, and puzzle of desiccation tolerance in plants. *Plant Ecology* 151: 5–17.
- Anable, M.E., M.P. McClaran, and G.B. Ruyle. 1992. Spread of introduced Lehmann lovegrass *Eragrostis lehmanniana* Nees. in Southern Arizona, USA. *Biological Conservation* 61:181–188. doi:10.1016/0006-3207(92)91114-8
- Andersen, D.C., and P.B. Shafroth. 2010. Beaver dams, hydrological thresholds, and controlled floods as a management tool in a desert riverine ecosystem, Bill Williams River, Arizona. *Ecohydrology* 3:325–338.
- Anderson, J. B., and J. Gerber. 2007. *Fifty Years of Change on the U.S.-Mexico Border: Growth, Development, and Quality of Life*. University of Texas Press, Austin, TX.
- Anderson, J.B., and J. Gerber. 2008. *Fifty years of change on the U.S.-Mexico border*. University of Texas Press, Austin, TX.
- Anderson, M.C., and F.R. Kay. 1999. Banner-tailed kangaroo rat burrow mounds and desert grassland habitats. *Journal of Arid Environments* 41:147–160. doi:10.1006/jare.1998.0478
- Andrade, E.R., Jr., and W.D. Sellers. 1988. El Niño and its effect on precipitation in Arizona and western New Mexico. *Journal of Climatology* 8:403–410.
- Angle, E.S. 2001. Hydrogeology of the Salt Basin. Pages 232-247. In: R.E. Mace, W.F. Mullican III, and E.S. Angle, (Eds.), *Aquifers of West Texas*. Austin, TX: Texas Water Development Board Report 356. Available at: https://www.twdb.texas.gov/publications/reports/numbered_reports/doc/R356/356_AquifersofWestTexas.pdf.
- Archer, S. 1994. Woody plant encroachment into southwestern grasslands and savannas: rates, patterns and proximate causes. Pages 13-68. In: Vavra, M., W.A. Laycock, and R.D. Pieper (Eds.), *Ecological Implications of Livestock Herbivory in the West*. Society for Range Management, Denver, CO.
- Archer, S.R., and K.I. Predick. 2008. Climate change and ecosystems of the southwestern United States. *Rangelands* 30:23–28.
- Archer, S., and F.E. Smeins. 1991. Ecosystem-level processes. Pages 109-139. In: R.K. Heitschmidt and J.W. Stuth (Eds.), *Grazing Management: An Ecological Perspective*. Timber Press, Portland, OR.
- Arizona Game and Fish Department 2013. *Arizona Statewide Pronghorn Management Plan*. Phoenix, AZ.
- Arnold, J.F., D.A. Jameson, and E.H. Reid. 1964. The pinyon-juniper type of Arizona: effects of grazing, fire, and tree control. Production Research Report 84. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Washington, D.C., USA.
- Arthington, A.H., R.J. Naiman, M.E. McClain, and C. Nilsson. 2010. Preserving the biodiversity and ecological services of rivers: new challenges and research opportunities. *Freshwater Biology* 55:1–16.
- Askins, R.A., F. Chávez-Ramírez, B.C. Dale, C.A. Haas, J.R. Herkert, F.L. Knopf, and P.D. Vickery. 2007. Conservation of Grassland Birds in North America: Understanding Ecological Processes in Different Regions: “Report of the AOU Committee on Conservation.” *Ornithological Monographs* iii–46. doi:10.2307/40166905
- Augustine, D.J., J.F. Cully, and T.L. Johnson. 2007. “Influence of Fire on Black-Tailed Prairie Dog Colony Expansion in Shortgrass Steppe.” *Rangeland Ecology & Management* 60 (5): 538–42.
- Augustine, D.J., and T.L. Springer. 2013. “Competition and Facilitation between a Native and a Domestic Herbivore: Trade-Offs between Forage Quantity and Quality.” *Ecological Applications: A Publication of the Ecological Society of America* 23 (4): 850–63.
- Avery, M.M. 2012. Seasonal effects of forage quantity, quality and dietary composition of pronghorn

(*Antilocarpa americana*) in a semi-arid environment (Master of Science). New Mexico State University, Las Cruces, NM.

- Avey, J.T., W.B. Ballard, M.C. Wallace, M.H. Humphrey, P.R. Krausman, F. Harwell, and E.B. Fish. 2003. Habitat relationships between sympatric mule deer and white-tailed deer in Texas. *The Southwestern Naturalist* 48:644–653.
- Avila-Flores, R., M.S. Boyce, and S. Boutin. 2010. "Habitat Selection by Prairie Dogs in a Disturbed Landscape at the Edge of Their Geographic Range." *Journal of Wildlife Management* 74 (5): 945–53.
- Axtell, R.W. 1977. Ancient playas and their influence on the recent herpetofauna of the northern Chihuahuan Desert. Pages 493–511. In Wauer, R. and D. H. Riskin (Eds.), *Transactions of the Symposium on the Biological Resources of the Chihuahuan Desert Region, United States and Mexico*. U.S. Department of the Interior, National Park Service Transactions and Proceedings Series No. 3, Washington, D.C.
- Ayarbe, J.P., T.L. Kieft. 2000. Mammal mounds stimulate microbial activity in a semiarid shrubland. *Ecology* 81:1150–1154. doi:10.2307/177187
- Babler, A.L., A. Pilati, and M.J. Vanni. 2011. Terrestrial support of detritivorous fish populations decreases with watershed size. *Ecosphere* 2:art76.
- Bachman, G.O. 1980. *Regional Geology and Cenozoic History of Pecos Region, Southeastern New Mexico*, Washington, DC: U.S. Geological Survey, Open-File Report 80-1099.
- Bachman, G.O. 1981. *Geology of Nash Draw, Eddy County, New Mexico*, Washington, DC: U.S. Geological Survey, Open-File Report 81-31.
- Bachman, G.O. 1987. *Karst in Evaporites in Southeastern New Mexico*, Albuquerque, NM: Sandia National Laboratories, Report SAND86-7078.
- Backlund, P., A. Janetos, and D. Schimel. 2008. The effects of climate change on agriculture, land resources, water resources, and biodiversity in the United States (Climate Change Science Program and the Subcommittee on Global Change Research). U.S. Department of Agriculture, Washington, D.C.
- Baddock, M.C., T.M. Zobeck, R.S. Van Pelt, and E.L. Fredrickson. 2011. Dust emissions from undisturbed and disturbed, crusted playa surfaces: Cattle trampling effects. *Aeolian Research* 3:31–41. doi:10.1016/j.aeolia.2011.03.007.
- Bahre, C.J. 1991. *A legacy of change: historic impact on vegetation in the Arizona Borderlands*. University of Arizona Press, Tucson, AZ, USA.
- Bahre, C.J. 1995. Human impacts on grasslands of southwestern Arizona. Pages 265-304. In: McClaran, M.P., Van Devender, T.R. (Eds.), *The Desert Grassland*. University of Arizona Press, Tucson, AZ, USA.
- Bahre, C.J., and M.L. Shelton. 1993. Historic vegetation change, mesquite increases, and climate in southeastern Arizona. *J. of Biogeography*, 20:489-504.
- Bailey, V. 1932. *Mammals of Mexico, North American fauna*. United States Department of Agriculture, Bureau of Biological Survey, Washington, D.C.
- Baillie, M.N., J.F. Hogan, B. Ekwurzel, A.K. Wahi, and C.J. Eastoe. 2007. Quantifying water sources to a semiarid riparian ecosystem, San Pedro River, Arizona. *Journal of Geophysical Research* 112:G03S02.
- Baker, R.H. 1971. Nutritional strategies of myomorph rodents in North American grasslands. *Journal of Mammalogy* 52, 800. doi:10.2307/1378928
- Baker, R.H. and K.A. Shump. 1978. *Sigmodon ochrognathus*. *Mammalian Species* 1. doi:10.2307/3503861
- Balda, R.P. 1987. Avian impacts on pinyon-juniper woodlands. Pages 525-533. In: Everett, R.L. (Ed.),

Proceedings, Pinyon-Juniper Conference. GTR-INT-215. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT, USA.

- Balda, R.P., and G.C. Bateman. 1971. Flocking and annual cycle of the piñon jay *Gymnorhinus cyanocephalus*. *Condor* 73:287–302.
- Balbus, J.M., and M.L. Wilson. 2000. Human health and global climate change: A review of potential impacts in the United States. Pew Center on Global Climate Change, Washington D.C.
- Barnes, F.J., and G.L. Cunningham. 1987. Water relations and productivity in pinyon-juniper habitat types, in: Everett, R.L. (Ed.), Proceedings of the Pinyon-Juniper Conference. GTR-INT-215. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT, USA, pp. 406–417.
- Barney, M.A., and N.C. Frischknecht. 1974. Vegetation changes following fire in the pinyon-juniper type of west-central Utah. *Journal of Range Management* 27:91–96.
- Barud-Zubillaga A. 2000. A Conceptual Model of the Hydrogeology of White Sands National Monument, South-Central New Mexico. M.S. Thesis, University of Texas at El Paso.
- Basnet, N., L.M. Hauck, and D. Pendergrass. 2013. Pecos River Water Quality Data Analysis and Dissolved Oxygen Modeling. Tarleton State University, Texas Institute for Applied Environmental Research, Stephenville, TX.
- Basurto, X., and D. Hadley. 2006. Grasslands Ecosystems, Endangered Species, and Sustainable Ranching in the Mexico-U.S. Borderlands: Conference Proceedings. <http://www.treesearch.fs.fed.us/pubs/22646>.
- Bateman, H.L., A. Chung-MacCoubrey, and H.L. Snell. 2008b. Impact of non-native plant removal on lizards in riparian habitats in the southwestern United States. *Restoration Ecology* 16:180–190.
- Bateman, H.L., A. Chung-MacCoubrey, D.M. Finch, H.L. Snell, and D.L. Hawksworth. 2008c. Impacts of non-native plant removal on vertebrates along the Middle Rio Grande (New Mexico). *Ecological Restoration* 26:193–195.
- Bateman, H.L., A. Chung-MacCoubrey, H.L. Snell, and D.M. Finch. 2009. Abundance and species richness of snakes along the Middle Rio Grande riparian forest in New Mexico. *Herpetological Conservation and Biology* 4:1–8.
- Bateman, H.L., M.J. Harner, and A. Chung-MacCoubrey. 2008a. Abundance and reproduction of toads (*Bufo*) along a regulated river in the southwestern United States: Importance of flooding in riparian ecosystems. *Journal of Arid Environments* 72:1613–1619.
- Bateman, H.L., E.H. Paxton, and W.S. Longland. 2013. Tamarix as Wildlife Habitat. Pages 168–188. In A.A. Sher and M.F. Quigley, (Eds.), *Tamarix: A Case Study of Ecological Change in the American West*. Oxford University Press, New York, NY.
- Baxter, R. M. 1977. Environmental effects of dams and impoundments. *Annual Review of Ecology, Evolution, and Systematics* 8:255–283.
- Bayley, P. B. 1995. Understanding large river-floodplain ecosystems. *BioScience* 45:153–158.
- Bean, M. G., and T.H. Bonner. 2010. Spatial and temporal distribution of the Asian fish tapeworm *Bothriocephalus acheilognathi* (Cestoda: Bothriocephalidea) in the Rio Grande (Río Bravo del Norte). *Journal of Aquatic Animal Health* 22:182–9.
- BEEC. 2014. Environmental Assessment for Restoration of Rio Grande Cutthroat Trout to the Las Animas Creek Watershed. Blue Earth Ecological Consultants, Inc., Report prepared for the New Mexico Department of Game and Fish, U.S. Fish and Wildlife Service, Southwest Region, USDA Forest Service, Gila National Forest, and Turner Ranch Properties, LP, Santa Fe, NM.

- Beck, R.F., and R.P. Gibbens. 1999. The Chihuahuan Desert Ecosystem. *New Mexico Journal of Science* 39.
- Beck, R.F., R. McNeely, M. Thomas, and C. Bailey, C. 2007. Seasonal and yearlong grazing in the northern Chihuahuan Desert: impacts of forage and cow-calf production (Agricultural Experiment Station No. BL-793), Bulletin 793. New Mexico State University, New Mexico.
- Bell, G., J. Baumgartner, J. Humke, A. Laurenzi, P. McCarthy, P. Mehlhop, K. Rich, M. Silbert, E. Smith, B. Spicer, T. Sullivan, and S. Yanoff. 1999. Ecoregional Conservation Analysis of the Arizona-New Mexico Mountains. The Nature Conservancy in New Mexico, Santa Fe, NM.
- Belsky, A.J. 1992. Effects of Grazing, Competition, Disturbance and Fire on Species Composition and Diversity in Grassland Communities. *Journal of Vegetation Science* 3:187–200.
- Belsky, A.J., A. Matzke, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. *Journal of Soil and Water Conservation* 54:419–431.
- Benda, L., K. Andras, D. Miller, and P. Bigelow. 2004a. Confluence effects in rivers: Interactions of basin scale, network geometry, and disturbance regimes. *Water Resources Research* 40:1–15.
- Benda, L., N.L. Poff, D. Miller, T. Dunne, G. Reeves, G. Pess, and M. Pollock. 2004b. The network dynamics hypothesis: How channel networks structure riverine habitats. *BioScience* 54:413–427.
- Bender, L.C. 2012. Guidelines for management of habitat for mule deer: Pinon-juniper, Chihuahuan desert, arid grasslands, and associated arid habitat types. (No. Circular 662). New Mexico State University College of Agriculture, Consumer and Environmental Sciences, Las Cruces, NM.
- Bender, L.C., J.C. Boren, H. Halbritter, S. Cox. 2011. Condition, survival, and productivity of mule deer in semiarid grassland-woodland in east-central New Mexico. *Human-Wildlife Interactions* 5:276–286.
- Bender, L.C., H. Li, B.C. Thompson, P.C. Morrow, R. Valdez. 2003. Infectious disease survey of Gemsbok in New Mexico. *Journal of Wildlife Diseases* 39:772–778. doi:10.7589/0090-3558-39.4.772
- Bender, L.C., L.A. Lomas, J. Browning. 2007. Condition, Survival, and Cause-Specific Mortality of Adult Female Mule Deer in North-Central New Mexico. *Journal of Wildlife Management* 71:1118–1124. doi:10.2193/2006-226
- Bennett, J., and D. Wilder. 2009. Physical Resources Foundation Report-White Sands National Monument. U.S. Department of the Interior, National Park Service, Natural Resource Program Center, Natural Resource Report NPS/NRPC/NRR-2009/166, Fort Collins, CO.
- Benson, L.D., and R.A. Darrow. 1981. Trees and shrubs of the southwestern deserts, 3rd ed. University of Arizona Press, Tucson, AZ, USA.
- Berg, W.A., J.A. Bradford, and P.L. Sims. 1997. Long-Term Soil Nitrogen and Vegetation Change on Sandhill Rangeland. *Journal of Range Management* 50:482–486.
- Bergeron, C.M., W.A. Hopkins, B.D. Todd, M.J. Hepner, and J.M. Unrine. 2011. Interactive effects of maternal and dietary mercury exposure have latent and lethal consequences for amphibian larvae. *Environmental Science & Technology* 45:3781–3787.
- Bergman, E.J., P.F. Doherty, G.C. White, A.A. Holland. 2015. Density dependence in mule deer: a review of evidence. *Wildlife Biology* 21:18–29.
- Best, T. 1999. Banner-tailed kangaroo rat, *Dipodomys spectabilis*, in: *The Smithsonian Book of North American Mammals*. Smithsonian Institution Press, Washington D.C., pp. 542–544.
- Best, T.L. 1988. *Dipodomys spectabilis*. *Mammalian Species* 1. doi:10.2307/3504199
- Best, T.L. 1972. Mound development by a pioneer population of the banner-tailed kangaroo rat, *Dipodomys spectabilis baileyi* Goldman, in eastern New Mexico. *American Midland Naturalist* 87:201.

doi:10.2307/2423893

- Best, T.L., M.P. Skupski, and R.A. Smartt. 1993. Food habits of sympatric rodents in the shinnery oak-mesquite grasslands of southeastern New Mexico. *The Southwestern Naturalist* 38:224–235. doi:10.2307/3671427
- Betancourt, J.L., Pierson, E.A., Rylander, K.A., Fairchild-Parks, J.A., Dean, J.S. 1993. Influence of history and climate on New Mexico pinyon-juniper woodlands, in: Aldon, E.F., Shaw, D.W. (Eds.), *Managing Pinyon Juniper Ecosystems for Sustainability and Social Needs*. GTR-RM-236. US Department of Agriculture, Forest Service, Rocky Mountain Experiment and Range Station, Fort Collins, CO, USA, pp. 42–62.
- Betancourt, J.L., Van Devender, T.R. 1981. Holocene vegetation in Chaco Canyon, New Mexico. *Science* 214: 658–661.
- Bexfield, L.M., and S.K. Anderholm. 1997. *Water-Quality Assessment of the Rio Grande Valley, Colorado, New Mexico, and Texas-Ground-Water Quality in the Rio Grande Flood Plain, Cochiti Lake, New Mexico, to El Paso, Texas, 1995*. U.S. Geological Survey, Water-Resources Investigations Report 96-4249, Albuquerque, NM.
- Bhattacharjee, J., J.P. Taylor, and L.M. Smith. 2006. Controlled flooding and staged drawdown for restoration of native cottonwoods in the Middle Rio Grande Valley, New Mexico, USA. *Wetlands* 26:691–702.
- Birney, E.C. 1973. Systematics of three species of woodrats (genus *Neotoma*) in central North America. *University of Kansas* 58:1–173.
- Blackburn, W.H., and P.T. Tueller. 1970. Pinyon and juniper invasion in black sagebrush communities in east-central Nevada. *Ecology* 51, 841–848.
- Blinn, D.W. 1993. Diatom community structure along physicochemical gradients in saline lakes. *Ecology*, 74(4):1246–1263.
- Block, G., and M.L. Morrison. 2010. Large-Scale Effects on Bird Assemblages in Desert Grasslands. *Western North American Naturalist* 70:19–25. doi:10.3398/064.070.0103
- Blossey, B. 1999. Before, during and after: the need for long-term monitoring in invasive plant species management. *Biological Invasions* 1:301–311.
- Bock, C., and J. Bock. 1977. Effects of fire on wildlife in southwestern lowland habitats (General Technical Report No. RM-191), Effects of fire management of Southwestern natural resources. USDA Forest Service, Fort Collins, CO, USA.
- Bock, C.E., and J.H. Bock. 1984. Response of birds, small mammals, and vegetation to burning sacaton grasslands in southeastern Arizona. *Journal of Range Management* 31:296–300. doi:10.2307/3897606
- Bock, C.E., and J.H. Bock. 1993. Cover of perennial grasses in southeastern Arizona in relation to livestock grazing. *Conservation Biology* 7(2): 371-377. DOI: 10.1046/j.1523-1739.1993.07020371.x.
- Bock, C.E., and J.H. Bock. 1992b. Response of Birds to Wildfire in Native versus Exotic Arizona Grassland. *The Southwestern Naturalist* 37, 73–81. doi:10.2307/3672149
- Bock, J.H. and C.E. Bock. 1992a. Short-term reductions in plant densities following prescribed fire in an ungrazed semidesert shrub-grassland. *Southwestern Naturalist* 37:49–53.
- Bock, J.H., C.E. Bock, J.R. McKnight. 1976. A study of the effects of grassland fires at the research ranch in southeastern Arizona. *Journal of the Arizona Academy of Science* 2:49–57.
- Bock, C.E., and W.M. Block. 2005. Fire and birds in the southwestern United States. Pages 14-32. In: Saab, V., and H. Powell (Eds.), *Fire and Avian Ecology in North America*, Studies in Avian Biology. Cooper

Ornithological Society, Camarillo, CA.

- Boeing, W.J., K.L. Griffis-Kyle, and J.M. Jungels. 2014. Anuran habitat associations in the northern Chihuahuan Desert, USA. *Journal of Herpetology* 48.
- Boersma, K.S., M.T. Bogan, B.A. Henrichs, and D.A. Lytle. 2014. Invertebrate assemblages of pools in arid-land streams have high functional redundancy and are resistant to severe drying. *Freshwater Biology* 59:491–501.
- Bogan, M.T. 2016. Personal communication. Dr. Michael T. Bogan, School of Natural Resources and the Environment, University of Arizona, Tucson, AZ, January 15, 2016.
- Bogan, M.T., and K.S. Boersma. 2012. Aerial dispersal of aquatic invertebrates along and away from arid-land streams. *Freshwater Science* 31:1131–1144.
- Bogan, M.T., K.S. Boersma, and D.A. Lytle. 2013. Flow intermittency alters longitudinal patterns of invertebrate diversity and assemblage composition in an arid-land stream network. *Freshwater Biology* 58:1016–1028.
- Bogan, M.T., K.S. Boersma, and D.A. Lytle. 2014a. Resistance and resilience of invertebrate communities to seasonal and suprasedational drought in arid-land headwater streams. *Freshwater Biology* 60:2547–2558.
- Bogan, M.T., N. Noriega-Felix, S.L. Vidal-Aguilar, L.T. Findley, D.A. Lytle, O.G. Gutiérrez-Ruacho, J.A. Alvarado-Castro, and A. Varela-Romero. 2014b. Biogeography and conservation of aquatic fauna in spring-fed tropical canyons of the southern Sonoran Desert, Mexico. *Biodiversity and Conservation* 23:2705–2748.
- Bonner, T.H., C. Thomas, C.S. Williams, and J.P. Karges. 2005. Temporal assessment of a west Texas stream fish assemblage. *The Southwestern Naturalist* 50:74–78.
- Booth, M.T., N.G. Hairston, and A.S. Flecker. 2013. How mobile are fish populations? Diel movement, population turnover, and site fidelity in suckers. *Canadian Journal of Fisheries and Aquatic Sciences* 70:666–677.
- Borderplex Alliance. 2016. The Borderplex Alliance: Regional Data: Population. <http://www.borderplexalliance.org/regional-data/el-paso/overview/population>. Last Accessed January 5, 2017.
- Bosselmann, K. 2010. Ecological integrity as the core meaning of sustainability. *Phase* 31:1–6.
- Boudet, C. 2009. Chihuahua deer mouse, Chihuahua plains mouse, gentle field mouse [WWW Document]. Mammals' Planet. URL <http://www.planet-mammiferes.org/drupal/en/node/39?indice=Peromyscus+maniculatus+blandus> (accessed 8.1.15).
- Boulton, A.J., T. Datry, T. Kasahara, M. Mutz, and J.A. Stanford. 2010. Ecology and management of the hyporheic zone: stream-groundwater interactions of running waters and their floodplains. *Journal of the North American Benthological Society* 29:26–40.
- Brand, L.A., B.R. Noon, and T.D. Sisk. 2006. Predicting abundance of desert riparian birds: Validation and calibration of the effective area model. *Ecological Applications* 16:1090–1102.
- Brand, L.A., M.D. Dixon, T. Fetz, J.C. Stromberg, S. Stewart, D.C. Goodrich, D.S. Brookshire, C.D. Broadbent, and K. Benedict. 2013. Projecting avian responses to landscape management along the Middle Rio Grande, New Mexico. *The Southwestern Naturalist* 58:150–162.
- Brandt, S.A. 2000. Classification of geomorphological effects downstream of dams. *Catena* 40:375–401.
- Braun, J.K., and M.A. Mares. 1989. *Neotoma micropus*. *Mammalian Species* 1. doi:10.2307/3504233

- Breshears, D.D., and C.D. Allen. 2002. The importance of rapid disturbance-induced losses in carbon management and sequestration. *Global Ecology and Biogeography* 11:1–5.
- Breshears, D.D., N.S. Cobb, P.M. Rich, K.P. Price, C.D. Allen, R.G. Balice, W.H. Romme, J.H. Kastens, M.L. Floyd, J. Belnap, J.J. Anderson, O.B. Myers, and C.W. Meyer. 2005. Regional vegetation die-off in response to global-change-type drought. *Proceedings of the National Academy of Sciences, U.S.A* 102:15144-15148.
- Brewer, C.E., and L.A. Harveson, L.A. 2007. Diets of bighorn sheep in the Chihuahuan Desert, Texas. *Southwestern Naturalist* 52:97-103.
- Bridges, A.S., M.J. Peterson, N.J. Silvy, F.E. Smeins, X.B. Wu. 2001. Differential influence of weather on regional quail abundance in Texas. *The Journal of Wildlife Management* 65:10-18.
- Briere, P.R. 2000. Playa, playa lake, sabkha: Proposed definitions for old terms. *Journal of Arid Environments* 45:1–7. doi:10.1006/jare.2000.0633
- Bristol, S. 1992. Environmental Contaminants in Water, Sediment and Biological Samples from Playa Lakes in Southeastern New Mexico. US Fish and Wildlife Service Region 2, Albuquerque, NM.
- Bristow, K.D., S.A. Dubay, and R.A. Ockenfels. 2006. Correlation between free water availability and pronghorn recruitment. Pages 55-62. In: Cain, J.W., and P.R. Krausman (Eds.), *Managing Wildlife in the Southwest*. Presented at the *Managing Wildlife in the Southwest*, The Wildlife Society, Alpine, TX.
- Bristow, K.D., and R.A. Ockenfels. 2006. Fall and Winter Habitat Use by Scaled Quail in Southeastern Arizona. *Rangeland Ecology & Management* 59:308–313. doi:10.2111/04-117R2.1
- Brockway, D.G., R.G. Gatewood, and R.B. Paris. 2002. Restoring grassland savannas from degraded pinyon-juniper woodlands: effects of mechanical overstory reduction and slash treatment alternatives. *Journal of Environmental Management* 64:179–197.
- Brooks, M.L., and J.C. Chambers. 2011. Resistance to invasion and resilience to fire in desert shrublands of North America. *Rangeland Ecology and Management* 64:431–438.
- Brooks, M.L., C.M. D’Antonio, D.M. Richardson, J.B. Grace, J.E. Keeley, J.M. DiTomaso, R.J. Hobbs, M. Pellant, and D. Pyke. 2004. Effects of invasive alien plants on fire regimes. *BioScience* 54:677–688. doi:10.1641/0006-3568(2004)054[0677:EOIAP0]2.0.CO;2
- Brooks, M.L., and D.A. Pyke. 2001. Invasive plants and fire in the deserts of North America. Pages 1-14. In: Galley, K.E.M., and T.P. Wilson (Eds.), *Proceedings of the Invasive Species Workshop: The Role of Fire in the Control and Spread of Invasive Species*. Fire Conference, Miscellaneous Publication No. 11. Presented at the Fire Conference 2000: the First National Congress on Fire Ecology, Prevention, and Management, Tall Timbers Research Station, Tallahassee, FL.
- Brostoff, W., R. Lichvar, and S. Sprecher. 2001. Delineating Playas in the Arid Southwest: A Literature Review (No. ERDC TR-01-4). US Army Corps of Engineers, Engineer Research and Development Center, Hanover, NH.
- Brown, D.E. 1982. Semidesert grassland. *Desert Plants* 4:123–131.
- Brown, D.E. 1994. *Biotic communities: southwestern United States and northwestern Mexico*. University of Utah Press, Salt Lake City, UT, USA.
- Brown, D.E., R.A. Ockenfels. 2007. *Arizona’s Pronghorn Antelope: a conservation legacy*. David E. Brown and Antelope Foundation, Phoenix, AZ.
- Brown, D.E., D. Warnecke, and T. McKinney. 2006. Effects of midsummer drought on mortality of doe pronghorn (*Antilocapra americana*). *Southwestern Naturalist* 51: 220–225.

- Brown, D.K., A.A. Echelle, D.L. Propst, J.E. Brooks, and W.L. Fisher. 2001. Catastrophic wildfire and number of populations as factors influencing risk of extinction for Gila trout (*Oncorhynchus gilae*). *Western North American Naturalist* 61:139–148.
- Brown, J.H., and S.K.M. Ernest. 2002. Rain and rodents: complex dynamics of desert consumers. *BioScience* 52:979–87. doi:10.1641/0006-3568(2002)052[0979:RARCD0]2.0.CO;2
- Brown, J.H., and E.J. Heske. 1990. Control of a desert-grassland transition by a keystone rodent guild. *Science* 250:1705–1707. doi:10.1126/science.250.4988.1705
- Brown, J.H., and E.J. Heske. 1990. Temporal changes in a Chihuahuan desert rodent community. *Oikos* 59:290–302. doi:10.2307/3545139
- Brown, J.H., and J.C. Munger. 1985. Experimental manipulation of a desert rodent community: Food addition and species removal. *Ecology* 66:1545–1563. doi:10.2307/1938017
- Brown, J.H., T.J. Valone, and C.G. Curtin. 1997. Reorganization of an arid ecosystem in response to recent climate change. *Proceedings of the National Academy of Sciences, U.S.A* 94:9729-9733.
- Brown, J.H., and Z. Zeng. 1989. Comparative population ecology of eleven species of rodents in the Chihuahuan Desert. *Ecology* 70:1507–25. doi:10.2307/1938209
- Brown, J.R., and S. Archer. 1999. Shrub invasion of grassland: recruitment is continuous and not regulated by herbaceous biomass or density. *Ecology* 80:2385–2396.
- Brown, J.S., P.B. Kotler, R.J. Smith, W.O. Wirtz. 1988. The effects of owl predation on the foraging behavior of heteromyid rodents. *Oecologia* 76:408–415. doi:10.1007/BF00377036
- Browning, D.M., M.C. Duniway, A.S. Laliberte, and A. Rango. 2012. Hierarchical analysis of vegetation dynamics over 71 years: soil-rainfall interactions in a Chihuahuan Desert ecosystem. *Ecological Applications* 22:909–926.
- Brune, G. 1975. Major and Historical Springs of Texas, Austin, TX: Texas Water Development Board, Report 189.
- Brunelle, A., and T. Minckley. 2002. Long-Term Records of Fire, Vegetation, and Climate from Ciénega in the Region of the San Bernardino National Wildlife Refuge, Arizona and Sonora. Southwest Consortium for Environmental Research and Policy, Unpublished project report, Project Number NR-04-02, Tempe, AZ.
- Brunjes, K.J., W.B. Ballard, M.H. Humphrey, H. Fielding, N.E. McIntyre, P.R. Krausman, M.C. Wallace. 2009. Home ranges of sympatric mule deer and white-tailed deer in Texas. *Southwestern Naturalist* 54:253–260.
- Budy, J.D., and R.O. Meeuwig. 1987. Pinyon-juniper silvics and silviculture. Pages 244-248. In: Everett, R.L. (Ed.), *Proceedings of the Pinyon-Juniper Conference*. GTR-INT-215. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT, USA.
- Buechner, H.K. 1950. Life history, ecology and range use of the Pronghorn Antelope in Trans-Pecos Texas. *American Midland Naturalist* 43:257–354.
- Buffington, L.C., and C.H. Herbel. 1965. Vegetational changes on a semidesert grassland range from 1858 to 1963. *Ecological Monographs* 35:139-164.
- Bunting, S.C., H.A. Wright, and L.F. Neuenschwander. 1980. Long-term effects of fire on cactus in the southern mixed prairie of Texas. *Journal of Range Management* 33:85–88.
- Buonopane, M., L.F. Huenneke, M. Remmenga, O. Eriksson. 2005. Community response to removals of plant functional groups and species from a Chihuahuan Desert shrubland. *Oikos* 110:67–80.
- Bureau of Land Management (BLM). 1998. Notice NM-030-1220-00: Emergency Closure of the Lordsburg

Playa to Off-Highway Vehicles (OHV), Hidalgo County, NM, Federal Register Vol. 63, No. 122, Thursday, June 25 1998, Notices 34661.

- Bureau of Land Management (BLM). 1999. Southern Playas Area in Southeast New Mexico. Bureau of Land Management, New Mexico State Office, Geographic Sciences Team, Albuquerque, NM.
- Bureau of Land Management (BLM). 2000. Final Environmental Impact Statement for Riparian and Aquatic Habitat Management in the Las Cruces Field Office – New Mexico (two volumes). U.S. Department of the Interior, Bureau of Land Management, Las Cruces Field Office BLM/NM/PL-00-011-1040, Las Cruces, NM.
- Bureau of Land Management (BLM) 2001. Rangeland Health Standards -- Public. H-4180-1. https://www.blm.gov/sites/blm.gov/files/.../Media_Library_BLM_Policy_h4180-1.pdf
- Bureau of Land Management (BLM) 2008. Integrated Vegetation Management Handbook -- Public. H-1740-2. Rel. 1-1714. https://www.blm.gov/sites/blm.../Media_Library_BLM_Policy_Handbook_H-1740-2.pdf.
- Bureau of Land Management (BLM). 2015. Restore New Mexico Bureau of Land Management. URL http://www.blm.gov/nm/st/en/prog/restore_new_mexico/restore_new_mexico.html (accessed 1.12.16).
- Burgess, T.L. 1995. Desert grassland, mixed shrub savanna, shrub steppe, or semidesert scrub? Pages 31-67. In: McClaran, M.P. and T.R. Van Devender (Eds.), *The Desert Grassland*. The University of Arizona Press, Tucson, AZ, USA.
- Burke, J.H. and W.A. Dick-Peddie. 1973. Comparative production of *Larrea divaricata* Cav, on three geomorphic surfaces in southern New Mexico. *Ecology* 54:1094–1102.
- Burke, M., K. Jorde, and J.M. Buffington. 2009. Application of a hierarchical framework for assessing environmental impacts of dam operation: changes in streamflow, bed mobility and recruitment of riparian trees in a western North American river. *Journal of Environmental Management* 90:S224–S236.
- Cable, D.R. 1965. Damage to mesquite, Lehmann lovegrass, and black grama by a hot June fire. *Journal of Range Management* 18:326–329.
- Cable, D.R. 1967. Fire effects on semi-desert grasses and shrubs. *Journal of Range Management* 20:170–176.
- Cable, D.R. 1971. Lehmann lovegrass on the Santa Rita Experimental Range, 1937-1968. *Journal of Range Management* 24:17–21.
- Cable, D.R. 1973. Fire effects in southwestern semidesert grass-shrub communities. Pages 109-127. In: *Proceedings 12th Annual Tall Timbers Fire Ecology Conference: A Quest for Ecological Understanding*. Lubbock, TX, USA. Tall Timbers Research, Inc., Tallahassee, FL, USA.
- Cable, D.R. 1975. Influence of precipitation on perennial grass production in the semidesert southwest. *Ecology* 56:981–986.
- Cahalane, V. 1954. *Mammals of North America*. MacMillan Co., New York, NY, USA.
- Cain III, J.W., P.R. Krausman, S.S. Rosenstock, J.C. Turner. 2006. Mechanisms of thermoregulation and water balance in desert ungulates. *Wildlife Society Bulletin* 34:570–581.
- Calamusso, B. 2005. Fishes of Southwestern Grasslands: Ecology, Conservation, and Management. Volume 2. Pages 141-168. In: Finch, D.M. (Ed.), *Assessment of Grassland Ecosystem Conditions in the Southwestern United States: Wildlife and Fish*. U.S. Department of Agriculture, Forest Service, General Technical Report RMRS-GTR-135.

- Callaway, R.M., E.H. DeLucia, D. Moore, R. Nowak, W.H. Schlesinger. 1996. Competition and facilitation: contrasting effects of *Artemisia tridentata* on desert vs. montane pines. *Ecology* 77:2130–2141.
- Callison, J., J.D. Brotherson, and J.E. Bowns. 1985. The effects of fire on the blackbrush (*Coleogyne ramosissima*) community of southwestern Utah. *Journal of Range Management* 38:535–538.
- Cameron, G.N. 1999. Hispid cotton rat *Sigmodon hispidus*. Pages 593-595. In: Wilson, D.E., Ruff, S. (Eds.), *The Smithsonian Book of North American Mammals*. Smithsonian Institution Press, Washington, D.C.
- Cameron, G.N., and S.R. Spencer. 1981. *Sigmodon hispidus*. *Mammal Species* 158:1–9.
- Campbell, R.S. 1929. Vegetative Succession in the Prosopis Sand Dunes of Southern New Mexico. *Ecology* 10:392–398.
- Canfield, R.H. 1939. The effect of intensity and frequency of clipping on density and yield of black grama and tobosa grass. Technical Bulletin 681. US Department of Agriculture, Forest Service, Southwestern Forest and Range Experiment Station. US Government Printing Office, Washington, DC, USA.
- Canon, S.K., and F.C. Bryant. 1997. Bed-site characteristics of Pronghorn fawns. *The Journal of Wildlife Management* 61:1134. doi:10.2307/3802111
- Cantu, R., C. Richardson. 1997. Mule Deer Management in Texas. Texas Parks and Wildlife Department, Austin, TX.
- Carman, S. 2010. White Sands Pupfish Status Report 2009. New Mexico Department of Game and Fish, Conservation Services Division, Santa Fe, NM.
- Castetter, E.F. 1956. The vegetation of New Mexico. *New Mexico Quarterly* 26:257–288.
- Cayan, D.R., and R.H. Webb. 1992. El Niño/Southern Oscillation and streamflow in the western United States. Pages 29-68. In: Diaz, H.F., and V. Markgraf, (Eds.), *El Niño: Historical and Paleoclimatic Aspects of the Southern Oscillation*. Cambridge University Press, Cambridge.
- Centers for Disease Control. Diseases directly transmitted by rodents [WWW Document] 2011. Centers for Disease Control and Prevention. URL <https://www.cdc.gov/rodents/diseases/direct.html> (accessed 8.1.15).
- Cerasale, D.J., and C.G. Guglielmo. 2010. An integrative assessment of the effects of Tamarisk on stopover ecology of a long-distance migrant along the San Pedro River, Arizona. *The Auk* 127:636–646.
- Chambers, J.C., B.A. Roundy, R.R. Blank, S.E. Meyer, and A. Whittaker. 2007. What makes Great Basin sagebrush ecosystems invasible by *Bromus tectorum*? *Ecological Monographs* 77: 117–145.
- Chansler, J.F. 1964. Overwintering habits of *Ips lecontei* Sw. and *Ips confusus* (Lec.) in Arizona and New Mexico, Research Note RM 27. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO, USA.
- Cheng, E., and M.E. Ritchie. 2006. “Impacts of Simulated Livestock Grazing on Utah Prairie Dogs (*Cynomys Parvidens*) in a Low Productivity Ecosystem.” *Oecologia* 147 (3): 546–55.
- Chew, R.M. 1979. Consumers as regulators of ecosystems: an alternative to energetics. *Ohio Journal of Science* 74:359–370.
- Chew, R.M. and A.E. Chew. 1965. The primary productivity of a desert-shrub (*Larrea tridentata*) community. *Ecological Monographs* 35:355–375.
- Chihuahuan Desert Research Institute (CDRI). 2014. Chihuahuan Desert Research Institute. A nonprofit organization established in 1974. URL <http://www.cdri.org>.
- Chiggs, S.R., D.E. Hubbard, K.B. Werlin, N.J. Haugerud, K.A. Powell, J. Thompson, and T. Johnson. 2006.

Association between wetland disturbance and biological attributes in floodplain wetlands. *Wetlands* 26:497–508.

- Chowdhury, A.H., C. Ridgeway, and R.E. Mace. 2004. Origin of the waters in the San Solomon Spring system, Trans-Pecos Texas. Pages 315-344. In: R.E. Mace, E.S. Angle, and W.F. Mullican, III. (Eds.), *Aquifers of the Edwards Plateau*. Texas Water Development Board, Report 360, Austin, Texas.
- Church, M. 2006. Bed material transport and the morphology of alluvial river channels. *Annual Review of Earth and Planetary Sciences* 34:325–354.
- Christensen, K.M., and T.G. Whitham. 1993. Impact of insect herbivores on competition between birds and mammals for pinyon pine seeds. *Ecology* 74:2270–2278.
- Clarkson, R.W., P.C. Marsh, S.E. Stefferud, and J.A. Stefferud. 2005. Conflicts between native fish and nonnative sport fish management in the Southwestern United States. *Fisheries* 30:20–27.
- Clemente, F., R. Valdez, J.L. Holechek, P.J. Zwank, and M. Cardenas. 1995. Pronghorn home range relative to permanent water in southern New Mexico. *The Southwestern Naturalist* 38–41.
- Climate Change Science Program (CCSP). 2008. Preliminary Review of Adaptation Options for Climate-Sensitive Ecosystems and Resources Final Report, Synthesis and Assessment Product 4.4. Page (S. H. Julius and J. M. West, Eds.). U.S. Climate Change Science Program and the Subcommittee on Global Change Research, Washington, D.C.
- Cobb, N.S., S. Mopper, C.A. Gehring, M. Caouette, K.M. Christensen, and T.G. Whitham. 1997. Increased moth herbivory associated with environmental stress of pinyon pine at local and regional levels. *Oecologia* 109:389–397.
- Cody, B.A., and N.T. Carter. 2008. *Flood Risk Management and Levees: A Federal Primer*. Congressional Research Service, Washington, D.C.
- Coffman, J.M., B.T. Bestelmeyer, J.F. Kelly, T.F. Wright, T.F., and R.L. Schooley. 2014. Restoration Practices Have Positive Effects on Breeding Bird Species of Concern in the Chihuahuan Desert. *Restoration Ecology* 22:336–344. doi:10.1111/rec.12081
- Cole, A.T., and C. Cole. 2015. An overview of aridland ciénagas, with proposals for their classification, restoration, and preservation. *The New Mexico Botanist Special Is*:28–56.
- Collier, M., R.H. Webb, and J.C. Schmidt. 1996. *Dams and Rivers: Primer on the Downstream Effects of Dams*. U.S. Geological Survey, Circular 1126, Denver, CO.
- Collins, K., and R. Ferrari. 2000a. Elephant Butte Reservoir 1999 Reservoir Survey. U.S. Department of the Interior, Bureau of Reclamation, Technical Service Center, Denver, CO.
- Collins, K., and R. Ferrari. 2000b. Caballo Reservoir 1999 Sedimentation Survey. U.S. Department of the Interior, Bureau of Reclamation, Technical Service Center, Denver, CO.
- Comer, P., D. Faber-Langendoen, R. Evans, S. Gawler, C. Josse, G. Kittel, S. Menard, M. Pyne, M. Reid, K. Schulz, K. Snow, and J. Teague. 2003. *Ecological Systems of the United States: A Working Classification of U.S. Terrestrial Systems*. Arlington, VA: NatureServe. <http://www.natureserve.org/library/usEcologicalsystems.pdf>.
- Commission for Environmental Cooperation (CEC). 2010. *Priority Conservation Areas: Grasslands, 2010*. Commission for Environmental Cooperation. URL <http://www.cec.org/Page.asp?PageID=924&ContentID=5609>.
- Connally, W. (Ed.). 2012a. *Texas Conservation Action Plan 2012-2016: Chihuahuan Deserts and Arizona-New Mexico Mountains*. Texas Parks and Wildlife Department, Austin, TX.

- Connally, W. (Ed.) 2012b. Texas Conservation Action Plan 2012-2016: Edwards Plateau. Texas Parks and Wildlife Department, Austin, TX.
- Connolly, G.E. 1978. Predator control and coyote populations: a review of simulation models, in: Coyotes: Biology, Behavior and Management. Academic press, New York, NY, USA, pp. 327–345.
- Conroy, M.J., and J.T. Peterson. 2012. Decision Making in Natural Resource Management: A Structured, Adaptive Approach. John Wiley & Sons.
- Cook, J. 1986. The mammals of the Animas Mountains and adjacent areas, Hidalgo County, New Mexico, in: Occasional Papers of the Museum of Southwestern Biology. University of New Mexico, Albuquerque, NM.
- Cooke, G.D., E.B. Welch, S. Peterson, and S.A. Nichols. 2005. Restoration and Management of Lakes and Reservoirs. Third. CRC Press, Boca Raton, FL.
- Cooper, D.J., and D.C. Andersen. 2012. Novel plant communities limit the effects of a managed flood to restore riparian forests along a large regulated river. *River Research and Applications* 215:204–215.
- Cornelius, J.M. 1988. Fire effects on vegetation of a northern Chihuahuan desert grassland (Dissertation). New Mexico State University, Las Cruces, NM, USA.
- Cornell, J.E., M. Gutiérrez, D.A. Wait, and H.O. Rubio-Arias. 2008. Ecological characterization of a riparian corridor along the Río Conchos, Chihuahua, Mexico. *The Southwestern Naturalist* 53:96–100.
- Cosentino, B.J., R.L. Schooley, B.T. Bestelmeyer, and J.M. Coffman. 2013. Response of lizard community structure to desert grassland restoration mediated by a keystone rodent. *Biodiversity and Conservation* 22:921–935. doi:10.1007/s10531-013-0459-7
- Cosentino, B.J., R.L. Schooley, B.T. Bestelmeyer, J.F. Kelly, and J.M. Coffman. 2014. Constraints and time lags for recovery of a keystone species (*Dipodomys spectabilis*) after landscape restoration. *Landscape Ecology* 29:665–675. doi:10.1007/s10980-014-0003-5
- Cottam, W.P., and G. Stewart. 1940. Plant succession as a result of grazing and of meadow desiccation by erosion since settlement in 1862. *Journal of Forestry* 38:613–626.
- Cowley, D.E., and J.E. Sublette. 1987. Distribution of fishes in the Black River drainage, Eddy County, New Mexico. *The Southwestern Naturalist* 32:213–221.
- Cowley, D.E., P. Shirey, and C. Hohman. 2003. Agricultural Irrigation Systems and Conservation of Native Fishes: Issues in the Rio Grande Valley of New Mexico. In: R.C. Runyan (Ed.), *Efficient Irrigation for Water Conservation*. Las Cruces, NM: New Mexico State University, College of Agriculture and Home Economics, Cooperative Extension Service, Agricultural Experiment Station, Water Task Force Report 1:49–55.
- Cox, J.R., A.D. Alba-Avila, R.W. Rice, and J.N. Cox. 1993. Biological and physical factors influencing *Acacia constricta* and *Prosopis velutina* establishment in the Sonoran Desert. *Journal of Range Management* 46:43–48.
- Cox, J.R., F.A. Ibarra, and M.H. Martin 1990. Fire effects on grasses in semiarid deserts. Pages 43-49. In: Krammes, J.S. (Ed.), *Effects of Fire Management of Southwestern Natural Resources*. Proceedings. General Technical Report RM-191. US Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. Fort Collins, CO, USA.
- Cox, J.R., H.L. Morton, T.N. Johnsen, G.L. Jordan, S.C. Martin, and L.C. Fierro. 1982. Vegetation restoration in the Chihuahuan and Sonoran Deserts of North America, *Agricultural Reviews and Manuals, Western Series, No. 28*. US Department of Agriculture, Agricultural Research Service, Oakland, California, USA.
- Cox, J.R., H.L. Morton, T.N. Johnsen, G.L. Jordan, S.C. Martin, and L.C. Fierro. 1984. Vegetation restoration in

the Chihuahuan and Sonoran deserts of North America. *Rangelands* 6:112–115.

- Cox, J.R., G.B. Ruyle, and B.A. Roundy. 1990. Lehmann lovegrass in southeastern Arizona: Biomass production and disappearance. *Journal of Range Management* 43:367–372. doi:10.2307/3898933
- Crawley, M., and A.J. Gray. 1987. What makes a community invasible? Pages 429-453. In: *Colonization, Succession and Stability*. Blackwell Scientific, Oxford, England.
- Crimmins, M.A., and A.C. Comrie. 2004. Interactions between antecedent climate and wildfire variability across south-eastern Arizona. *International Journal of Wildland Fire* 13:455–466.
- Crist, P., M. Reid, H. Hamilton, G. Kittel, S. Auer, M. Harkness, D. Braun, J. Bow, C. Scott, L. Misztal, and L. Kutner. 2014. Madrean Archipelago Rapid Ecoregional Assessment Final Report. NatureServe technical report to the Bureau of Land Management. Report, appendices, and databases provided to the Bureau of Land Management. http://www.blm.gov/wo/st/en/prog/more/Landscape_Approach/reas.html.
- Curtin, C.G., D.A. Kelt, T.C. Frey, and J.H. Brown. 2000. On the role of small mammals in mediating climatically driven vegetation change. *Ecology Letters* 3:309–317. doi:10.1046/j.1461-0248.2000.00166.x
- Curtin, C.G., N.F. Sayre, and B.D. Lane. 2002. Transformations of the Chihuahuan borderlands: grazing, fragmentation, and biodiversity conservation in desert grasslands. *Environmental Science & Policy* 5:55–68. doi:10.1016/S1462-9011(02)00020-5
- Daly, C., Halbleib, M., Smith, J.I., Gibson, W.P., Doggett, M.K., Taylor, G.H., Curtis, J., Pasteris, P.P. 2008. Physiographically sensitive mapping of climatological temperature and precipitation across the conterminous United States. *International Journal of Climatology* 28:2031-2064.
- D'Antonio, C.M., and P.M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review of Ecology and Systematics* 23(1): 63-87.
- Davidson, A.D., and D.C. Lightfoot. 2007. Interactive effects of keystone rodents on the structure of desert grassland arthropod communities. *Ecography* 30:515–525. doi:10.1111/j.0906-7590.2007.05032.x
- Davidson, A.D., and D.C. Lightfoot. 2006. Keystone rodent interactions: prairie dogs and kangaroo rats structure the biotic composition of a desertified grassland. *Ecography* 29:755–765. doi:10.1111/j.2006.0906-7590.04699.x
- Davidson, A.D., D.C. Lightfoot and J.L. McIntyre. 2008. Engineering rodents create key habitat for lizards. *Journal of Arid Environments* 72:2142–2149. doi:10.1016/j.jaridenv.2008.07.006
- Davidson, A.D., E. Ponce, D.C. Lightfoot, E.L. Fredrickson, J.H. Brown, J. Cruzado, S.L. Brantley, R. Sierra-Corona, R. List, D. Toledo, and G. Ceballos. 2010. "Rapid Response of a Grassland Ecosystem to an Experimental Manipulation of a Keystone Rodent and Domestic Livestock." *Ecology* 91 (11): 3189–3200.
- Davis, D.R., and J.S. Hopkins. 1992. Lake water quality assessment surveys: playa lakes (No. NMED/SWQ-93/2). Surveillance and Standards Section, Surface Water Quality Bureau, New Mexico Environment Department, Santa Fe, NM.
- Davis, O.K. and R.M. Turner. 1986. Palynological evidence for the historic expansion of juniper and desert shrubs in Arizona. *Review of Palaeobotany and Palynology* 49:177–193.
- Davis, R., and R. Sidner. 1999. Yellow-nosed cotton rat, *Sigmodon ochrognathus*. Pages 595-596. In: Ruff, S., Wilson, D.E. (Eds.), *The Smithsonian Book of North American Mammals*. Smithsonian Institution Press, Washington D.C.
- Dean, D.J., and J.C. Schmidt. 2011. The role of feedback mechanisms in historic channel changes of the lower Rio Grande in the Big Bend region. *Geomorphology* 126:333–349.

- Deason, M.G. 1998. An historical overview of playas and other wetland/riparian areas of "Nuevo Mexico." *New Mexico Journal of Science* 38:189–218.
- DeBano, L.F. 1999. *Biodiversity and the Management of the Madrean Archipelago: The Sky Islands of Southwestern United States and Northwestern Mexico*. DIANE Publishing.
- DeBano, L.H., P.H. Ffolliott, A. Ortega-Rubio, G.J. Gottfried, R.H. Hamre, and C.B. Edminster, (Eds.), 1995. *Biodiversity and Management of the Madrean Archipelago: The Sky Islands of Southwestern United States and Northwestern Mexico, September 19-23, 1994, Tucson, AZ*. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-GTR-264, Fort Collins, CO.
- Deloach, C.J., R.I. Carruthers, J.E. Lovich, T.L. Dudley, and S.D. Smith. 2000. Ecological interactions in the biological control of saltcedar (*Tamarix* spp.) in the United States: toward a new understanding. Pages 819–873. In: Spencer, N.R., (Ed.) *Proceedings of the X International Symposium on Biological Control of Weeds*, Bozeman, Montana, USA, 4-14 July, 1999. Montana State University, Bozeman, MT.
- Desmond, M., and J.A. Montoya. 2006. Status and Distribution of Chihuahuan Desert Grasslands in the United States and Mexico. Pages 17-25. In: Basurto, X., Hadley, D. (Eds.), *Grasslands Ecosystems, Endangered Species, and Sustainable Ranching in the Mexico-U.S. Borderlands: Conference Proceedings*. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Proceedings RMRS-P-40, Fort Collins, CO.
- Detling, J.K. 1988. Grasslands and savannas: regulation of energy flow and nutrient cycling by herbivores. Pages 131-148. In: Alberts, J.J. and L.L. Pomeroy (Eds.), *Concepts of Ecosystem Ecology*. Springer-Verlag, New York
- Dettinger, M., B. Udall, and A. Georgakakos. 2015. Western water and climate change. *Ecological Applications* 25:2069–2093. <http://onlinelibrary.wiley.com/doi/10.1890/15-0938.1/full>. Accessed October 9, 2016.
- Dick-Peddie, W.A., and M.S. Alberico. 1977. *Fire ecology study of the Chisos Mountains*. The Chihuahuan Desert Research Institute, Alpine, TX.
- Dick-Peddie, W.A., M.H. William, and R. Spellenberg. 1993. *New Mexico Vegetation: Past, Present, and Future*. Albuquerque NM: University of New Mexico Press.
- Diebel, M.W., M. Fedora, S. Cogswell, and J.R. O’Hanley. 2015. Effects of road crossings on habitat connectivity for stream-resident fish. *River Research and Applications* 31:1251–1261.
- DiGennaro, B., D. Reed, C. Swanson, L. Hastings, Z. Hymanson, M. Healey, S. Siegel, S. Cantrell, and B. Herbold. 2012. Using conceptual models and decision-support tools to guide ecosystem restoration planning and adaptive management: An example from the Sacramento–San Joaquin Delta, California. *San Francisco Estuary and Watershed Science* 10(3): 1-15. <http://escholarship.org/uc/item/3j95x7vt>.
- Dinerstein, E., D. Olson, J. Atchley, C. Loucks, S. Contrera-Balderas, R. Abell, E. Iñigo, E. Enkerlin, C. Williams, and G. Castilleja (Eds.), 2001. *Ecoregion-Based Conservation in the Chihuahuan Desert: A Biological Assessment*, 2nd printing with corrections. A collaborative effort by World Wildlife Fund, Comisión Nacional para el Conocimiento y Uso de la Biodiversidad (CONABIO), The Nature Conservancy, PRONATURA Noreste, and the Instituto Tecnológico y de Estudios Superiores de Monterrey (ITESM).
- Di Tomaso, J.M. 1998. Impact, biology, and ecology of saltcedar (*Tamarix* spp.) in the southwestern United States. *Weed Technology* 12(2):326–336.
- D’Odorico, P., J.D. Fuentes, W.T. Pockman, S.L. Collins, Y. He, J.S. Medeiros, S. DeWekker, and M.E. Litvak. 2010. Positive feedback between microclimate and shrub encroachment in the northern Chihuahuan Desert. *Ecosphere* 1(6), art17. doi:10.1890/ES10-00073.1.

- Doody, T.M., P.L. Nagler, E.P. Glenn, G.W. Moore, K. Morino, K.R. Hultine, and R.G. Benyon. 2011. Potential for water salvage by removal of non-native woody vegetation from dryland river systems. *Hydrological Processes* 25:4117–4131.
- Dreesen, D., J. Harrington, T. Subirge, P. Stewart, and G. Fenchel. 2002. Riparian restoration in the Southwest: species selection, propagation, planting methods, and case studies (No. RMRS-P-24), National Proceedings: Forest and Conservation Associations -1999, 2000, and 2001. U.S. Department of Agriculture, Rocky Mountain Research Station, Ogden, UT, USA.
- Drewa, P.B., and K.M. Havstad. 2001. Effects of fire, grazing and the presence of shrubs on Chihuahuan Desert grasslands. *Journal of Arid Environments* 48:429–443.
- Drewa, P.B., and D.P. Peters. 2001. Fire, grazing, and honey mesquite invasion in black grama-dominated grasslands of the Chihuahuan Desert: a synthesis. Pages 31-39. In: *The Role of Fire in the Control and Spread of Invasive Species*. Presented at the First National Congress On Fire Ecology, Prevention, and Management, Tall Timbers Research Station, Tallahassee, FL.
- Drewa, P.B., D.P.C. Peters, and K.M. Havstad. 2006. Population and Clonal Level Responses of a Perennial Grass Following Fire in the Northern Chihuahuan Desert. *Oecologia* 150:29–39.
- Drewien, R.C., W.M. Brown, and W.L. Kendall. 1995. Recruitment in Rocky Mountain Greater sandhill cranes and comparison with other crane populations. *The Journal of Wildlife Management* 59:339–356.
- Dudley, R.K., and S.P. Platania. 2007. Flow regulation and fragmentation imperil pelagic-spawning riverine fishes. *Ecological Applications* 17:2074–2086.
- Dudley, R.K., and S.P. Platania. 2011. Rio Grande Silvery Minnow Population Monitoring Program Results from September 2009 to October 2010. Report by American Southwest Ichthyological Researchers, L.L.C., Albuquerque, NM, for the U.S. Bureau of Reclamation, Albuquerque Area Office, Albuquerque, NM.
- Dunning, J.B. Jr., R.K. Jr. Bowers, Jr., S.J. Suter, and C.E. Bock. 1999. Cassin's Sparrow (*Aimophila cassinii*). *The Birds of North America Online*. doi:10.2173/bna.471
- Dwyer, D.D., and R.D. Pieper. 1967. Fire effects on blue grama-Pinyon-Juniper rangeland in New Mexico. *Journal of Range Management* 20:359–362.
- Dye, J.L. 1998. Gemsbok and mule deer diets in southern New Mexico (Master of Science). New Mexico State University, Las Cruces, NM.
- Eads, R.B., G.C. Menzies, and V.I. Miles. 1952. Acarina taken during west Texas plague studies. *Proceedings of the Entomological Society* 54:250–253.
- East, J. 2015. Spatial and temporal variation in aquatic food-web structure in the Pecos River in New Mexico and Texas, USA. M.S. Thesis, Texas Tech University.
- Echelle, A.A., A.F. Echelle, S.C. Balderas, and M.D.L. Lozano Vilano. 2003. Pupfishes of the northern Chihuahuan desert: status and conservation. Pages 111–126. In: Garrett, G.P. and N.L. Allan (Eds.), *Aquatic Fauna of the Northern Chihuahuan Desert Contributed Papers from a Special Session, Thirty-Third Annual Symposium, Desert Fishes Council, 17 November 2001*. Museum of Texas Tech University, Special Publication Number 46, Lubbock, Texas.
- Edwards, R.J. 1997. *Ecological Profiles for Selected Stream-Dwelling Texas Freshwater Fishes*, Edinburg, TX: University of Texas-Pan American, Department of Biology, Report to the Texas Water Development Board, TWDB Contract 95-483-107.
- Edwards, R. J., G. P. Garrett, and E. Marsh-Matthews. 2002. Conservation and status of the fish communities inhabiting the Río Conchos basin and middle Rio Grande, México and U.S.A. *Reviews in Fish Biology and*

Fisheries 12:119–132.

- Edwards, R. J., G. P. Garrett, and E. Marsh-Matthews. 2003. Fish assemblages of the Rio Conchos basin, Mexico, with emphasis on their conservation and status. Pages 75–89. In: Garrett, G.P. and N.L. Allan (Eds.), *Aquatic Fauna of the Northern Chihuahuan Desert Contributed Papers from a Special Session, Thirty-Third Annual Symposium*, Desert Fishes Council, 17 November 2001. Museum of Texas Tech University, Special Publication Number 46, Lubbock, Texas.
- Edwards, R.J., G.P. Garrett, and N.L. Allan. 2004. Aquifer-Dependent Fishes of the Edwards Plateau Region. Pages 253–268. In: Mace, R.E., E.S. Angle, and W.F. Mullican II (Eds.), *Aquifers of the Edwards Plateau*. Texas Water Development Board, Report 360, Austin, Texas.
- El-Hage, A. and D.W. Moulton. 1998. Evaluation of Selected Natural Resources in Parts of Loving, Pecos, Reeves, Ward, and Winkler Counties, Texas, Austin, TX: Texas Department of Parks and Wildlife, Resource Protection Division.
- Ellis, L.M., M.C. Molles, Jr., C.S. Crawford, and F. Heinzemann. 2000. Surface-active arthropod communities in native and exotic riparian vegetation in the middle Rio Grande valley, New Mexico. *The Southwestern Naturalist* 45:456–471.
- Eng, K., D.M. Wolock, and M.D. Dettinger. 2016. Sensitivity of intermittent streams to climate variations in the USA. *River Research and Applications* 32:885–895.
- Enquist, C.A.F., E.H. Girvetz, and D.F. Gori. 2008. A climate change vulnerability assessment for biodiversity in New Mexico, Part II: Conservation Implications of Emerging Moisture Stress due to Recent Climate Changes in New Mexico. The Nature Conservancy in New Mexico, Santa Fe, NM.
- Erdman, J.A. 1970. Piñon-juniper succession after natural fires on residual soils of Mesa Verde, Colorado. *Brigham Young University Science Bulletin, Biological Series* 11:1–26.
- Ernest, K.A., E.F. Aldon, E. Muldavin. 1993. Woody debris in undisturbed pinyon-juniper woodlands of New Mexico. Pages 117-123. In: Aldon, E.F., and D.W. Shaw (Eds.), *Managing Pinyon Juniper Ecosystems for Sustainability and Social Needs*. GTR-RM-236. U.S. Department of Agriculture, Forest Service, Rocky Mountain Experiment and Range Station, Fort Collins, CO, USA.
- Esparza-Carlos, J.P., J.W. Laundre, and V.J. Sosa. 2011. Precipitation impacts on mule deer habitat use in the chihuahuan desert of Mexico. *Journal of Arid Environments* 75.
- Evans, R.A. 1988. Management of pinyon-juniper woodlands. GTR-INT-249. U.S. Department of Agriculture, Forest Service, Intermountain Research Service, Ogden, UT, USA.
- Eve, M.D., K.M. Havstadt, and W.G. Whitford. 1999. Applying satellite imagery to triage assessment of ecosystem health. *Environmental monitoring and assessment* 54:205–227.
- Everett, R.L. 1987. Plant response to fire in the pinyon-juniper zone, in: Everett, R.L. (Ed.), *Proceedings of the Pinyon-Juniper Conference*. GTR-INT-215. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT, USA, pp. 152–157.
- Faber-Langendoen, D., G. Kudray, C. Nordman, L. Sneddon, L. Vance, E. Byers, J. Rocchio, S. Gawler, G. Kittel, S. Menard, P. Comer, E. Muldavin, M. Schafale, T. Foti, C. Josse, and J. Christy. 2008. *Ecological Performance Standards for Wetland Mitigation: An Approach Based on Ecological Integrity*. NatureServe, Arlington, VA.
- Facka, A.N., G.W. Roemer, V.L. Mathis, M. Kam, M. and E. Geffen. 2010. Drought leads to collapse of black-tailed prairie dog populations reintroduced to the Chihuahuan Desert. *The Journal of Wildlife Management* 74:1752-1762.
- Fain, T.G., J.A. Montoya, M. Briggs, M. Lockwood, J. Deloach, M. Donet, and M. Davidson. 2014. Panel

- presentation: Tamarisk control and riparian restoration on the Rio Grande border. Pages 71–85. In Hoyt, C.A., and J. Karges (Eds.), *Proceedings of the Sixth Symposium on the Natural Resources of the Chihuahuan Desert Region, October 14–17, 2004*. Chihuahuan Desert Research Institute, Fort Davis, TX.
- Farley, G.H., L.M. Ellis, J. N. Stuart, and N.J. Scott, Jr. 1994. Avian species richness in different-aged stands of riparian forest along the middle Rio Grande, New Mexico. *Biological Conservation* 8:1098–1108.
- Ferrari, R. 2013. Brantley Reservoir 2013 Bathymetric Survey. U.S. Department of the Interior, Bureau of Reclamation, Technical Service Center, Technical Report No. SRH-2013-23, Denver, CO.
- Findley, J.S. 1987. *The natural history of New Mexican mammals*. University of New Mexico Press, Albuquerque, NM.
- Findley, J.S., A.H. Harris, D.E. Wilson, and C. Jones. 1975. *Mammals of New Mexico*. University of New Mexico, Albuquerque, NM.
- Finley, R.B. 1958. The wood rats of Colorado—distribution and ecology. *University of Kansas* 10:213–552.
- Fischenich, J.C. 2008. The application of conceptual models to ecosystem restoration. U.S. Army Engineers, Engineer Research and Development Center (ERDC), Ecosystem Management and Restoration Research Program (EMRRP), Technical Note ERDC/EBA TN-08-1, February 2008. Vicksburg, MS. www.wes.army.mil/el/emrrp.
- Fitzgerald, C.S., P.R. Krausman, and M.L. Morrison. 2001. Short-term impacts of prescribed fire on a rodent community in desert grasslands. *The Southwestern Naturalist* 46:332–337. doi:10.2307/3672429
- Fitzgerald, J.P., C.A. Meaney, and D.M. Armstrong. 1994. *Mammals of Colorado*. Denver Museum of Natural History. University Press of Colorado, Niwot, Colorado.
- Fitzhugh, T.W., and R.M. Vogel. 2011. The impacts of dams on flood flows in the United States. *River Research and Applications* 27:1192–1215.
- Fleishman, E., N. McDonal, R.M. Nally, D.D. Murphy, J. Walters, T. Floyd. 2003. Effects of floristics, physiognomy and non-native vegetation on riparian bird communities in a Mojave Desert watershed. *Journal of Animal Ecology* 72:484–490.
- Flesch, A.D. 2014. Distribution, abundance, habitat, and biogeography of breeding birds in the Sky Islands and adjacent Sierra Madre Occidental of northwest Mexico. University of Arizona, School of Natural Resources and the Environment, Final report to U.S. National Park Service and U.S. Fish and Wildlife Service, CESU Agreement P08AC00077/J1212080048 and FWS Cooperative Agreement F12AP00566, Tucson,
- Fletcher, T. 2000. *Gembok diets in creosote shrubland and Great Basin conifer woodland in southcentral New Mexico* (Master of Science). New Mexico State University, Las Cruces, NM.
- Floyd, M.L., M. Clifford, N.S. Cobb, D. Hanna, R. Delph, P. Ford, D. Turner. 2009. Relationship of stand characteristics to drought-induced mortality in three southwestern piñon-juniper woodlands. *Ecological Applications* 19:1223–1230.
- Floyd, M.L., W.H. Romme, and D.D. Hanna. 2000. Fire history and vegetation pattern in Mesa Verde National Park, Colorado, USA. *Ecological Applications* 10:1666–1680.
- Forstner, M.R.J., J.R. Dixon, T.M. Guerra, J.M. Winters, J.N. Stuart, and S.K. Davis. 2014. Status of U.S. populations of the Big Bend Slider (*Trachemys gaigeae*). Pages 335–367. In: Hoyt, C.A. and J. Karges (Eds.), *Proceedings of the Sixth Symposium on the Natural Resources of the Chihuahuan Desert Region, October 14–17, 2004*. Chihuahuan Desert Research Institute, Fort Davis, TX.
- Fox, K.B., Krausman, P.R. 1994. Fawning habitat of desert mule deer. *The Southwestern Naturalist* 39:269–275.

- Franssen, N.R., E.I. Gilbert, and D.L. Propst. 2015. Effects of longitudinal and lateral stream channel complexity on native and non-native fishes in an invaded desert stream. *Freshwater Biology* 60(1):16-30.
- Fredrickson, E., K.M. Havstad, R. Estell, and P. Hyder. 1998. Perspectives on desertification: southwestern United States. *Journal Arid Environments* 39(2): 191-207. <http://dx.doi.org/10.1006/jare.1998.0390>.
- Freeman, C.E. 1973. Some germination responses of lechuguilla (*Agave lechuguilla* Torr.). *Southwestern Naturalist* 18:125–134.
- Frey, J.K., and T.L. Yates. 1996. Mammalian diversity in New Mexico. *New Mexico Journal of Science* 36: 4–37.
- Friggens, M.M., K.E. Bagne, D.M. Finch, and D. Falk, D. 2013. Review and recommendations for climate change vulnerability assessment approaches with examples from the Southwest (General Technical Report No. RMRS-GTR-309). US Department of Agriculture, Fort Collins, CO, USA.
- Friggens, M.M., and C.K. Woodlief. 2014. Synthesis of Aquatic Vulnerability Assessments for the Interior West. United States Forest Service, Rocky Mountain Research Station, Annual Report to the Southern Rockies Landscape Conservation Cooperative, Albuquerque, NM.
- Friggens, M.M., D.M. Finch, K.E. Bagne, S.J. Coe, and D.L. Hawksworth. 2013a. Vulnerability of Species to Climate Change in the Southwest: Terrestrial Species of the Middle Rio Grande. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-306, Fort Collins, CO.
- Friggens, M.M., K.E. Bagne, D.M. Finch, D. Falk, J. Triepke, and A. Lynch. 2013b. Review and Recommendations for Climate Change Vulnerability Assessment Approaches with Examples from the Southwest. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-3, Fort Collins, CO.
- Fuchs, E.H. 2002. Historic increases in woody vegetation in Lincoln County, New Mexico. Vanguard Printing Company, Albuquerque, NM, USA.
- Fuller, M.R., M.W. Doyle, and D.L. Strayer. 2015. Causes and consequences of habitat fragmentation in river networks. *Annals of the New York Academy of Sciences* 1355:31–51.
- Fuller, P.L., L.G. Nico, and J.D. Williams. 1999. Nonindigenous Fishes Introduced into Inland Waters of the United States. American Fisheries Society Special Publication 27. Bethesda, MD.
- Fullerton, W., and D. Batts. 2003. Hope for a Living River-A Framework for a Restoration Vision for the Rio Grande. Alliance for Rio Grande Heritage, Albuquerque, NM.
- Furniss, R.L., and V.M. Carolin. 1977. Western forest insects. Miscellaneous Publication 1339. U.S. Department of Agriculture, Washington, D.C., USA.
- Gage, R.T., L.A. Harveson, B. Warnock, P.M. Harveson. 2011. Effects of Spike 20P on vegetation and mule deer in Trans-Pecos, Texas. In: Liley, S.G. (Ed.), Proceedings of the 9th Western States and Provinces Deer and Elk Workshop. Presented at the 9th Western States Deer and Elk Workshop, New Mexico Department of Game and Fish, Santa Ana Pueblo, NM.
- Gaines, L., M. Hemstrom, J. Kagan, and J. Salwasser. 2013. Integrated Landscape Assessment Project Final Report. The Institute for Natural Resources, Oregon State University and Portland State University, Corvallis and Portland, OR.
- Gao, Q., and J.F. Reynolds. 2003. Historical shrub-grass transitions in the northern Chihuahuan Desert: modeling the effects of shifting rainfall seasonality and event size over a landscape gradient. *Global Change Biology* 9:1475–1493. doi:10.1046/j.1365-2486.2003.00676.x

- Garrett, G.P., and R.J. Edwards. 2014. Changes in fish populations in the Lower Canyons of the Rio Grande. Pages 396–408. In: Hoyt, C.A. and J. Karges (Eds.), Proceedings of the Sixth Symposium on the Natural Resources of the Chihuahuan Desert Region, October 14–17, 2004. Chihuahuan Desert Research Institute, Fort Davis, TX.
- Garrett, M.G., and W.L. Franklin. 1988. "Behavioral Ecology of Dispersal in the Black-Tailed Prairie Dog." *Journal of Mammalogy* 69 (2): 236–50.
- Gaylord, M.L., T.E. Kolb, W.T. Pockman, J.A. Plaut, E.A. Yopez, A.K. Macalady, R.E. Pangle, and N.G. McDowell. 2013. Drought predisposes pinyon-juniper woodlands to insect attacks and mortality. *New Phytologist* 198:567-578.
- Gebow, B.S., and W.L. Halvorson. 2005. Managing Fire in the Northern Chihuahuan Desert: a Review and Analysis of the Literature. U.S. Geological Survey Open File Report 2005-1157, 35 pp. <http://pubs.usgs.gov/of/2005/1157/>. Accessed October 10, 2016.
- Gedir, J.V., J.W. Cain, G. Harris, and T.T. Turnbull. 2015. Effects of climate change on long-term population growth of pronghorn in an arid environment. *Ecosphere* 6:art189. doi:10.1890/ES15-00266.1
- Geiger, E.L., and G.R. McPherson. 2005. Response of semi-desert grasslands invaded by non-native grasses to altered disturbance regimes. *Journal of Biogeography* 32:895–902.
- Gentry, H.S. 1972. The agave family in Sonora. Agriculture Handbook 399. US Department of Agriculture, Agricultural Research Service, US Government Printing Office, Washington, DC, USA.
- George, D.B., C.T. Webb, K.M. Pepin, L.T. Savage, and M.F. Antolini. 2013. "Persistence of Black-Tailed Prairie-Dog Populations Affected by Plague in Northern Colorado, USA." *Ecology* 94 (7): 1572–83.
- George, P., R.E. Mace, and W.F. Mullican III. 2005. The Hydrogeology of Hudspeth County, Texas, Austin, TX: Texas Water Development Board Report 364.
- George, P.G., R.E. Mace, and R. Petrossian. 2011. Aquifers of Texas. Texas Water Development Board, Report 380, Austin, TX. Available at: www.twdb.texas.gov.
- Ghil, M., and R. Vautgard. 1991. Interdecadal oscillations and the warming trend in global temperature time series. *Nature* 350:324–327.
- Gibbens, R.P., and R.F. Beck. 1988. Changes in grass basal area and forb densities over a 64-year period on grassland types of the Jornada Experimental Range. *Journal of Range Management* 41:186–192.
- Gibbens, R.P., and J.M. Lenz. 2001. Root systems of some Chihuahuan Desert plants. *Journal of Arid Environments* 49:221–263.
- Gibbens, R.P., R.P. McNeely, K.M. Havstad, R.F. Beck, and B. Nolen. 2005. Vegetation changes in the Jornada Basin from 1858 to 1998. *Journal of Arid Environments* 61:651-668.
- Gibson, P.P., and J.D. Olden. 2014. Ecology, management, and conservation implications of North American beaver (*Castor canadensis*) in dryland streams. *Aquatic Conservation: Marine and Freshwater Ecosystems* 24:391–409.
- Gibson, P.P., J.D. Olden, and M.W. O'Neill. 2014. Beaver dams shift desert fish assemblages toward dominance by non-native species (Verde River, Arizona, USA). *Ecology of Freshwater Fish* 2014:1–18.
- Gido, K.B., and D.L. Propst. 1999. Habitat use and association of native and nonnative fishes in the San Juan River, New Mexico and Utah. *Copeia* 1999:321–332.
- Gido, K.B., and D.L. Propst. 2012. Long-Term Dynamics of Native and Nonnative Fishes in the San Juan River, New Mexico and Utah, under a Partially Managed Flow Regime. *Transactions of the American Fisheries Society* 141:645–659.

- Gido, K. B., D. L. Propst, J. D. Olden, and K. R. Bestgen. 2013. Multidecadal responses of native and introduced fishes to natural and altered flow regimes in the American Southwest. *Canadian Journal of Fisheries and Aquatic Sciences* 70:554–564.
- Gill, A.M. 1977. Plants' traits adaptive to fires in the Mediterranean land ecosystems. Pages 17-26. In: Mooney, H.A., and C.E. Conrad (Eds.), *The Environmental Consequences of Fire and Fuel Management in Mediterranean Ecosystems*. Proceedings. General Technical Report WO-3. US Department of Agriculture, Forest Service, US Government Printing Office, Washington, DC, USA.
- Gitlin, A.R., C.M. Sthultz, M.A. Bowker, S. Stumpf, K.L. Paxton, K. Kennedy, A. Munoz, J.K. Bailey, T.G. Whitham. 2006. Mortality gradients within and among dominant plant populations as barometers of ecosystem change during extreme drought. 20 1477–1486.
- Glass, G.E., Yates, T.L., Fine, J.B., Shields, T.M., Kendall, J.B., Hope, A.G., Parmenter, C.A., Peters, C.J., Ksiazek, T.G., Li, C.-S., Patz, J.A., Mills, J.N. 2002. Satellite imagery characterizes local animal reservoir populations of Sin Nombre virus in the southwestern United States. *Proceedings of the National Academy of Sciences* 99:16817–16822. doi:10.1073/pnas.252617999
- Glendening, G.E., and H.A. Paulsen, Jr. 1955. Reproduction and establishment of velvet mesquite as related to invasion of semidesert grasslands. Technical Bulletin 1127. US Department of Agriculture, Forest Service, US Government Printing Office, Washington, DC, USA.
- Glinski, R.L., and D.E. Brown. 1982. Mesquite (*Prosopis juliflora*) response to severe freezing in southeastern Arizona. *Journal of the Arizona-Nevada Academy of Science* 17:15–18.
- Goodrich, S., and B. Barber. 1999. Return interval for pinyon juniper following fire in the Green River corridor, near Dutch John, Utah. Pages 391-393. In: Monsen, S.B., and Stevens, R. (Eds.), *Proceedings: Ecology and Management of Pinyon Juniper Communities within the Interior West*. RMRS-P-9. U.S. Department of Agriculture, Forest Service, Ogden, UT, USA.
- Goolsby, J.A., P.J. Moran, A.E. Racelis, K.R. Summy, M.M. Jimenez, R.D. Lacewell, A. Perez de Leon, and A.A. Kirk. 2016. Impact of the biological control agent *Tetramesa romana* (Hymenoptera: Eurytomidae) on *Arundo donax* (Poaceae: Arundinoideae) along the Rio Grande River in Texas. *Biocontrol Science and Technology* 26:47–60.
- Goertz, J.W. 1964. The influence of habitat quality upon density of cotton rat populations. *Ecological Monographs* 34, 359–81. doi:10.2307/2937068
- Gordon, C.E. 2000. Movement Patterns of Wintering Grassland Sparrows in Arizona. *The Auk* 117:748–759. doi:10.1642/0004-8038(2000)117[0748:MPOWGS]2.0.CO;2
- Gordon, E., and R.K. Meentemeyer. 2006. Effects of dam operation and land use on stream channel morphology and riparian vegetation. *Geomorphology* 82:412–429.
- Gori, D., and J. Bate. 2007. Historical range of variation and state and transition modeling of historical and current landscape conditions for pinyon-juniper of the southwestern U.S. The Nature Conservancy, Tucson, AZ, USA.
- Gosz, J.R., D.L. Moore, G.A. Shore, and H.D. Grover. 1995. Lightning estimates of precipitation location and quantity on the Sevilleta LTER, New Mexico. *Ecological Applications* 5:1141–1150.
- Gosz, R.J., and J.R. Gosz. 1996. Species interactions on the biome transition zone in New Mexico: response of blue grama (*Bouteloua gracilis*) and black grama (*Bouteloua eriopoda*) to fire and herbivory. *Journal of Arid Environments* 34:101–114.
- Gottfried, G.J. 1987. Regeneration of Pinyon. Pages 249-254. In: Everett, R.L. (Ed.), *Proceedings of the Pinyon-Juniper Conference*. GTR-INT-215. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT, USA.

- Gottfried, G.J., and P.F. Ffolliott. 1995. Stand dynamics on upper elevation pinon-juniper watershed at Beaver Creek, Arizona. Pages 38-45. In: Shaw, D.W., E.F. Aldon, and C. LoSapio (Eds.), *Desired Future Conditions for Piñon-Juniper Ecosystems*. Flagstaff, AZ, USA. GTR-RM-258. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Research Station, Fort Collins, CO, USA.
- Gottfried, G.J., and K.E. Severson. 1993. Distribution and multiresource management of pinon-juniper woodlands in the southwestern United States. Pages 108-116. In: Aldon, E.F., and D.W. Shaw (Eds.), *Managing Pinyon Juniper Ecosystems for Sustainability and Social Needs*. GTR-RM-236. U.S. Department of Agriculture, Forest Service, Rocky Mountain Experiment and Range Station, Fort Collins, CO, USA.
- Gottfried, G.J., T.W. Swetnam, C.D. Allen, J.L. Betancourt, A.L. Chung-MacCoubrey. 1995. Pinyon-Juniper Woodlands. Pages 95-132. In: Finch, D.M., and J.A. Tainter, J.A. (Eds.), *Ecology, Diversity, and Sustainability of the Middle Rio Grande Basin*. GTR-RM-268. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Research Station, Fort Collins, CO, USA.
- Graf, W.L. 1999. Dam nation: A geographic census of American dams and their large-scale hydrologic impacts. *Water Resources Research* 35:1305–1311.
- Graf, W.L. 2005. Geomorphology and American dams: The scientific, social, and economic context. *Geomorphology* 71:3–26.
- Graf, W.L. 2006. Downstream hydrologic and geomorphic effects of large dams on American rivers. *Geomorphology* 79:336–360.
- Grant, G.E., J.C. Schmidt, and S.L. Lewis. 2003. A geological framework for interpreting downstream effects of dams on rivers. Pages 209–226. In O’Connor, J.E. and G.E. Grant, (Eds.), *A Peculiar River, Geology, Geomorphology, and Hydrology of the Deschutes River, Oregon*. American Geophysical Union Water Science and Application 7, Washington, D.C.
- Graves, G.N., W.C. Bennett, J.R. Wheeler, B.E. Miller, and D.L. Forcum. 1974. Sylvatic plague studies in southeast New Mexico. I. Flea-host relationships from six years’ study. *Journal of Medical Entomology*. 11:488–498.
- Graves, H.S. 1917. The pine trees of the Rocky Mountain region. Bulletin 460. U.S. Department of Agriculture, Forest Service, Washington, DC, USA.
- Green, M.T., P.E. Lowther, S.L. Jones, S.K. Davis, and B.C. Dale. 2002. Baird’s Sparrow (*Ammodramus bairdii*). *The Birds of North America Online*. doi:10.2173/bna.638
- Gregg, M.A., M. Bray, K.M. Kilbride, M.R. Dunbar. 2001. Birth synchrony and survival of Pronghorn fawns. *The Journal of Wildlife Management* 65:19. doi:10.2307/3803271
- Gregory, L., and W. Hatler. 2008. *A Watershed Protection Plan for the Pecos River in Texas*. Texas A&M University, Texas Water Resources Institute and Texas AgriLife Extension Service, College Station, TX.
- Gresswell, R.E. 1999. Fire and Aquatic Ecosystems in Forested Biomes of North America. *Transactions of the American Fisheries Society* 128:193–221.
- Griffith, G. E., J. M. Omernik, M.M. McCraw, G. Z. Jacobi, C.M. Canavan, T. S. Schrader, D. Mercer, R. Hill, and B.C. Moran. 2006. *Ecoregions of New Mexico*. U.S. Geological Survey, Reston, Virginia.
- Griffith, G.E., S.A. Bryce, J.M. Omernik, J.A. Comstock, A.C. Rogers, B. Harrison, S.L. Hatch, and D. Bezanson. 2004. *Ecoregions of Texas*. U.S. Environmental Protection Agency, Corvallis, OR.
- Griffiths, D.A. 1910. A protected stock range in Arizona. Bulletin 177. US Department of Agriculture, Bureau of Plant Industry, Office of Farm Management, US Government Printing Office, Washington, DC, USA.
- Grover, H.D., and H.B. Musick. 1990. Shrubland encroachment in southern New Mexico, USA: An analysis of

desertification processes in the American southwest. *Climatic Change* 17:305–330.
doi:10.1007/BF00138373

- Grunstra, M. and O.W. Van Auken. 2007. Factors that influence the distribution and cover of *Helianthus paradoxus* in a west Texas salt marsh. *Phytologia* 89:24–42.
- Guay, J., and C.M. Senger. 1962. The occurrence of *Catenotaenia* sp. in *Dipodomys spectabilis* in Arizona. *The Journal of Parasitology* 48:451. doi:10.2307/3275213
- Guo, Q. 2004. Slow recovery in desert perennial vegetation following prolonged human disturbance. *Journal of Vegetation Science* 15, 757–762.
- Guo, Y., H. Zhao, X. Zuo, S. Drake, and X. Zhao. 2008. Biological soil crust development and its topsoil properties in the process of dune stabilization, Inner Mongolia, China. *Environmental Geology* 54(3):653-662.
- Gutiérrez, M., and E. Johnson. 2014. Salinity trend in the Lower Rio Conchos, Chihuahua. Pages 60–70. In Hoyt, C.A. and J. Karges (Eds.), *Proceedings of the Sixth Symposium on the Natural Resources of the Chihuahuan Desert Region, October 14–17, 2004*. Chihuahuan Desert Research Institute, Fort Davis, TX.
- Gutzwiller, K.J., Jr., W.C.B. 2008. Desert Bird Associations with Broad-Scale Boundary Length: Applications in Avian Conservation. *Journal of Applied Ecology* 45, 873–882.
- Haase, P., D. Hering, S.C. Jähnig, A.W. Lorenz, and A. Sundermann. 2012. The impact of hydromorphological restoration on river ecological status: a comparison of fish, benthic invertebrates, and macrophytes. *Hydrobiologia* 704:475–488.
- Hafner, D.J., and E. Yensen. 1998. North American rodents: status survey and conservation action plan. IUCN, Gland, Switzerland.
- Hagen, E.M., and J.L. Sabo. 2012. Influence of river drying and insect availability on bat activity along the San Pedro River, Arizona (USA). *Journal of Arid Environments* 84:1–8.
- Hagen, E.M., and J.L. Sabo. 2014. Temporal variability in insectivorous bat activity along two desert streams with contrasting patterns of prey availability. *Journal of Arid Environments* 102:104–112.
- Hagle, S.K., Gibson, K.E., and Tunnock, S.T. 2003. Field guide to diseases and insect pests of northern and central Rocky Mountain conifers. Forest Health Protection Report R1-03-08. U.S. Department of Agriculture, Forest Service, Missoula, MT, USA.
- Hall, E.R. 1981. *The mammals of North America*, 2nd ed. John Wiley and Sons, New York, NY, USA.
- Hallett, J.G. 1982. Habitat selection and the community matrix of a desert small-mammal fauna. *Ecology* 63:1400–1410. doi:10.2307/1938867
- Hancock, P.J., A.J. Boulton, and W.F. Humphreys. 2005. Aquifers and hyporheic zones: Towards an ecological understanding of groundwater. *Hydrogeology Journal* 13:98–111.
- Hanley, T.A., and J.L. Page. 1982. Differential effects of livestock use on habitat structure and rodent populations in Great Basin communities. *California Fish and Game* 68:1–160.
- Hanna, A.H., K.W. Conway, E.W. Carson, G.P. Garrett, and J.R. Gold. 2013. Conservation genetics of an undescribed species of *Dionda* (Teleostei: Cyprinidae) in the Rio Grande Drainage in western Texas. *The Southwestern Naturalist* 58:35–40.
- Harings, N.M., Boeing, W.J. 2014. Desert anuran occurrence and detection in artificial breeding habitats. *Herpetologica* 70:123–134.
- Harper, K.T., and Davis, J.N. 1999. Biotic, edaphic, and other factors influencing pinyon-juniper distribution in the Great Basin. Pages 51-54. In: Monsen, S.B., and Stevens, R. (Eds.), *Proceedings: Ecology and*

Management of Pinyon Juniper Communities within the Interior West. RMRS-P-9. U.S. Department of Agriculture, Forest Service, Ogden, UT, USA.

- Harveson, L.A., S. Gray, B. Tarrant, J. Weaver, J. Hoffman. 2012a. Trans-Pecos pronghorn restoration and research report 2012. Borderlands Research Institute for Natural Resource Management, Alpine, TX.
- Harveson, L.A., A. James, S. Gray, R. Gate, J. Hoffman, J. Martinez de laLuz, A. Ortega-Sanchez, and R. Walser. 2012b. A report on mule deer research in the Chihuahuan Desert. Borderlands Research Institute for Natural Resource Management, Alpine, TX.
- Hastings, J.R., and Turner, R.M. 1965. The changing mile: an ecological study of vegetation change with time in the lower mile of an arid and semiarid region. University of Arizona Press, Tucson, AZ, USA.
- Hatfield, J., K. Boote, P. Fay, L. Hahn, C. Izaurralde, B.A. Kimball, T. Meder, J. Morgan, D. Ort, W. Polley, A. Thompson, and D. Wolfe. 2008. Agriculture, The effects of climate change on agriculture, land resources, water resources and biodiversity. U.S. Climate Change Science Program and Subcommittee on Global Change Research, Washington D.C.
- Hatler, W.L., and C.R. Hart. 2009. Water loss and salvage in saltcedar (*Tamarix* spp.) stands on the Pecos River, Texas. *Invasive Plant Science and Management* 2:309–317.
- Hauer, F.R., B.J. Cook, M.C. Gilbert, E.J. Clairain, and R.D. Smith. 2002. A Regional Guidebook for Applying the Hydrogeomorphic Approach to Assessing Wetland Functions of Riverine Floodplains in the Northern Rocky Mountains Environmental Laboratory. U.S. Army Corps of Engineers, Engineer Research and Development Center Environmental Laboratory, ERDC/EL TR-02-21, Vicksburg, MS.
- Haukos, D.A., and L.M. Smith. 1992. Ecology of Playa Lakes (No. Leaflet 13.3.7), Waterfowl Management Handbook. USFWS, Washington, D.C.
- Havstad, K.M., and R. Beck. 1996. Research in the Jornada Basin of southern New Mexico: a field tour (United States Department of Agriculture Forest Service General Technical Report No. INT-GTR-338). Las Cruces, NM.
- Havstad, K.M., and D.P. Coffin Peters. 1999. People and rangeland biodiversity—North America, in: *The Sixth International Rangeland Congress*. Townsville, Australia, pp. 634–638.
- Havstad, K.M., E.L. Fredrickson, and L.F. Huenneke. 2006. Grazing livestock management in an arid ecosystem. Pages 266-277. In: Havstad, K.M., L.F. Huenneke, and W.H. Schlesinger (Eds.), *Structure and Function of a Chihuahuan Desert Ecosystem: The Jornada Basin Long-Term Ecological Research Site*. Oxford University Press, New York, NY.
- Havstad, K.M., and W.H. Schlesinger. 2006. Introduction. Pages 3-14. In: Havstad, K.M., L.F. Huenneke, and W.H. Schlesinger (Eds.), *Structure and Function of a Chihuahuan Desert Ecosystem: the Jornada Basin Long-Term Ecological Research Site*, edited by New York: Oxford University Press.
- Hawkins, L.K. 1996. Burrows of kangaroo rats are hotspots for desert soil fungi. *Journal of Arid Environments* 32:239–249. doi:10.1006/jare.1996.0020
- Hawley, J.W. 1993. Geomorphic Setting and Late Quaternary History of Pluvial Lake Basins in the Southern New Mexico Region, Socorro, NM: New Mexico Bureau of Mines and Mineral Resources, Open-file Report 391.
- Haynes, C.V., Jr. 1991. Geoaerchological and paleohydmlgical evidence for a Clovis-age drought in North America and its bearing on extinction. *Quaternary Research* 35:438–450.
- Heard, T.C., J.S. Perkin, and T.H. Bonner. 2012. Intra-annual variation in fish communities and habitat associations in a Chihuahua desert reach of the Rio Grande/Rio Bravo del Norte. *Western North American Naturalist* 72:1–15.

- Hedden, S.C., K.B. Gido, and J.E. Whitney. 2016. Introduced flathead catfish consumptive demand on native fishes of the upper Gila River, New Mexico. *North American Journal of Fisheries Management* 36:55–61.
- Heffelfinger, J. 2006. *Deer of the Southwest: A Complete Guide to the Natural History, Biology, and Management of Southwestern Mule Deer and White-Tailed Deer*, First. ed. Texas A & M University Press, College Station, TX.
- Heffelfinger, J.R., C. Brewer, C.H. Alcala-Galvan, B. Hale, D.L. Weybright, B.E. Wakeling, L.H. Carpenter, N.L. Dodd. 2006. Habitat guidelines for mule deer: southwest deserts ecoregion. Mule Deer Working Group, Western Association of Fish and Wildlife Agencies, Cheyenne, WY.
- Heerwagen, A. 1956. Mixed prairie in New Mexico, in: *Grasslands of the Great Plains: Their Nature and Use*. Johnsen Publishing Company, Lincoln, NE, USA, p. 395.
- Heirman, A.L., and H.A. Wright. 1973. Fire in medium fuels of west Texas. *Journal of Range Management* 26:331–335.
- Heitmuller, F.T. 2006. Texas Springs Inventory, Flow, and Water Quality, in: Texas Groundwater Protection Committee Meeting February 2, 2006 Presentations. Texas Groundwater Protection Committee, Austin, Texas.
- Heitmuller, F.T., and B.D. Reece. 2003. Database of Historically Documented Springs and Spring Flow Measurements in Texas. U.S. Geological Survey, Open-File Report 03-315, Reston, VA.
- Heitmuller, F.T. and I.P. Williams. 2006. Compilation of Historical Water-Quality Data for Selected Springs in Texas by Ecoregion, Reston, VA: U.S. Geological Survey, Data Series 230. Available at: <http://pubs.usgs.gov/ds/2006/230/>.
- Hem, J.D. 1992. *Study and Interpretation of the Chemical Characteristics of Natural Water*. U.S. Geological Survey, Water-Supply Paper 2254, Third Edition, Reston, VA.
- Hemstrom, M.A. 2014. Integrating social, economic, and ecological values across large landscapes (No. PNW-GTR-896). U.S. Department of Agriculture, U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon.
- Henderson, C.W. 1971. Comparative temperature and moisture responses in Gambel and scaled quail. *The Condor* 73:430-436.
- Hendrickson, J. 1979. Saline habitats and halophytic vegetation of the Chihuahuan Desert region. Pages 289–314. In: Wauer, R.H. and D.H. Riskind (Eds.), *Transactions of the Symposium on the Biological Resources of the Chihuahuan Desert Region, United States and Mexico, 17–18 October 1974*. Alpine Texas: Sul Ross State University. <https://babel.hathitrust.org/cgi/pt?id=umn.31951002827525u;view=1up;seq=1>. Accessed October 8, 2016.
- Hennessy, J.T., R.P. Gibbens, J.M. Tromble, and M. Cardenas, M. 1983. Vegetation changes from 1935 to 1980 in mesquite dunelands and former grasslands of southern New Mexico. *Journal of Range Management* 36:370-374.
- Henrickson, J., and M.C. Johnston. 1986. Vegetation and community types of the Chihuahuan Desert. Pages 21-39. In: Barlow, J.C., A.M. Powell, and B.N. Timmermann (Eds.), *Proceedings of the 2nd Symposium on Resources of the Chihuahuan Desert Region, United States and Mexico*. Chihuahuan Desert Research Institute, Sul Ross State University, Alpine, TX.
- Herbel, C.H., F.N. Ares, and R.A. Wright. 1972. Drought effects on a semidesert grassland range. *Ecology* 53:1084–1093.

- Herbst, D.B., and S.D. Cooper. 2010. Before and after the deluge: rain-on-snow flooding effects on aquatic invertebrate communities of small streams in the Sierra Nevada, California. *Journal of the North American Benthological Society* 29:1354–1366.
- Hernandez, H.M., and R.T. Barcenas. 1995. Endangered cacti in the Chihuahuan Desert: I. distribution patterns. *Conservation Biology* 9:1176–1188.
- Herrick, J.E., K.M. Havstad, A. Rango. 2006. Remediation research in the Jornada Basin: past and future. Pages 278-304. In: Havstad, K.M., L.F. Huenneke, and W.S. Schlesinger (Eds.), *Structure and Function of a Chihuahuan Desert Ecosystem: The Jornada Basin Long-Term Ecological Research Site*. Oxford University Press, Inc., New York, NY, USA.
- Hershler, R., H.P. Liu, and J. Howard. 2014. Springsnails: A new conservation focus in western North America. *BioScience* 64:693–700.
- Hershler, R., Liu, H.-P., Landye, J.J. 2011. New species and records of springsnails (Caenogastropoda: Cochliopidae: Tryonia) from the Chihuahuan Desert (Mexico and United States), an imperiled biodiversity hotspot. *Zootaxa* 3001:1–32.
- Heske, E.J., J.H. Brown, and S. Mistry. 1994. Long-term experimental study of a Chihuahuan Desert rodent community: 13 years of competition. *Ecology* 75:438–445. doi:10.2307/1939547
- Heske, E.J., and M. Campbell. 1991. Effects of an 11-year livestock enclosure on rodent and ant numbers in the Chihuahuan desert, southeastern Arizona. *The Southwestern Naturalist* 36:89–93. doi:10.2307/3672121
- Hibbs, B.J., M.M. Lee, J.W. Hawley, and J.F. Kennedy. 2000. Some notes on the hydrogeology and ground-water quality of the Animas Basin system, southwestern New Mexico. Pages 227-234. In: *New Mexico Geological Society 51st Annual Fall Field Conference Guidebook*. Presented at the Southwest Passage: A trip through the Phanerozoic, New Mexico Geological Society.
- Hibler, C.P. 1981. Diseases. Pages 129-156. In: Wallmo, O.C. (Ed.), *Mule and Black-Tailed Deer of North America*. University of Nebraska Press, Lincoln, NE.
- Higgins, C.L., and R.E. Strauss. 2008. Modeling stream fish assemblages with niche apportionment models: Patterns, processes, and scale dependence. *Transactions of the American Fisheries Society* 137:696–706.
- Hill, A. 1990. Ecology and classification of the pinyon-juniper woodlands in western New Mexico (Dissertation). New Mexico State University, Las Cruces, NM, USA.
- Hill, C.A. 2000. Overview of the geologic history of cave development in the Guadalupe Mountains, New Mexico. *Journal of Cave and Karst Studies*, 62(2):60–71.
- Hill, D.P., and L.K. Gould. Chestnut-collared Longspur (*Calcarius ornatus*). *The Birds of North America Online*. doi:10.2173/bna.288
- Hill, M.E., Jr. and V.T. Holliday. 2011. Paleoindian and later occupations along ancient shorelines of the San Agustín Plains, New Mexico. *Journal of Field Archaeology*, 36(1):3–20.
- Hinojosa-Huerta, O., S. DeStefano, Y. Carrillo-Guerrero, W.W. Shaw, C. Valdés-Casillas. 2004. Waterbird communities and associated wetlands of the Colorado River Delta, México. *Studies in Avian Biology* 27:52–60.
- Hoagstrom, C.W. 2003. Historical and recent fish fauna of the lower Pecos River. Pages 92–110. In: Garrett, G.P., and N.L. Allan (Eds.), *Aquatic Fauna of the Northern Chihuahuan Desert Contributed Papers from a Special Session, Thirty-Third Annual Symposium, Desert Fishes Council, 17 November 2001*. Museum of Texas Tech University, Special Publication Number 46, Lubbock, TX.

- Hoagstrom, C.W. 2009. Causes and impacts of salinization in the lower Pecos River. *Great Plains Research* 19:27–44.
- Hoagstrom, C.W. and J.E. Brooks. 1999. Distribution, status, and abundance of the Pecos Pupfish, *Cyprinodon Pecosensis*, Santa Fe, NM: New Mexico Department of Game and Fish, Technical Report No. 2.
- Hoagstrom, C.W., J.E. Brooks, and S.R. Davenport. 2008. Recent habitat association and the historical decline of *Notropis simus pecosensis*. *River Research and Applications* 24:789–803.
- Hoagstrom, C.W., N.D. Zymonas, S. Davenport, D.L. Propst, and J.E. Brooks. 2010. Rapid species replacements between fishes of the North American plains: a case history from the Pecos River. *Aquatic Invasions* 5:141–153.
- Hobbs, N.T., D.S. Schimel, C.E. Owensby, and D.S. Ojima. 1991. Fire and grazing the tallgrass prairie: Contingent effects on nitrogen budgets. *Ecology* 72(4): 1374-1384. DOI: 10.2307/1941109.
- Hoffmeister, D.F. 1986. *Mammals of Arizona*. University of Arizona Press, Tucson, Arizona.
- Hogan, J.F. 2013. Water quantity and quality challenges from Elephant Butte to Amistad. *Ecosphere*, 4(1), p. Article 9. Available at: <http://www.esajournals.org/doi/abs/10.1890/ES12-00302.1>.
- Holdenried, R., and S.F. Quan. 1956. Susceptibility of New Mexico rodents to experimental plague. *Public Health Rep* 71, 979–984.
- Holechek, J., and D. Galt. 2000. Grazing intensity guidelines. *Rangelands* 22:11–14.
- Holechek, J., D. Galt, J. Joseph, J. Navarro, G. Kumalo, F. Molinar, and M. Thomas. 2003. Moderate and light cattle grazing effects on Chihuahuan Desert rangelands. *Journal of Range Management* 56: 133–139. doi:10.2307/4003896
- Holechek, J.L. 1991. Chihuahuan desert rangelands, livestock grazing and sustainability. *Rangelands* 13:115–120.
- Holechek, J.L., A. Tembo, A. Daniel, M.J. Fusco, and M. Cardenas. 1994. Long-term grazing influences on Chihuahuan Desert rangeland. *Southwestern Naturalist* 342–349.
- Hooper, E.T. 1968. Classification. Pages 27-74. In: King, J.A. (Ed.), *Biology of Peromyscus (Rodentia)*. The American Society of Mammalogists, Stillwater, OK.
- Hope, A.G., and R.R. Parmenter. 2007. Food habits of rodents inhabiting arid and semi-arid ecosystems of central New Mexico, Special publication. Museum of Southwestern Biology, Albuquerque, New Mexico.
- Howard, J.L. 1995. *Antilocapra americana*, Fire Effects Information System [Online]. U.S. Forest Service, Rocky Mountain Research Station.
- Howell, J. 1941. Pinon and juniper woodlands of the Southwest. *Journal of Forestry* 39, 542–545.
- Howells, R.G. 2003. Declining status of freshwater mussels in the Rio Grande, with comments on other bivalves. Pages 60-73. In: G.P. Garrett and N.L. Allan (Eds.), *Aquatic Fauna of the Northern Chihuahuan Desert Contributed Papers from a Special Session, Thirty-Third Annual Symposium, Desert Fishes Council, 17 November 2001*. Lubbock, TX: Museum of Texas Tech University, Special Publication Number 46. Available at: http://www.desertfishes.org/dfc/proceed/2001/Chih_desert_symp/AFNCD_index.html.
- Hoyt, Cathryn A. 2002. The Chihuahuan Desert: Diversity at risk” *Endangered Species Bulletin* XXVII (2): 16–17.
- Hubbs, C. 2003. Spring-endemic gambusia of the Chihuahuan desert. Pages 127–133. In Garrett, G.P., and N.L. Allan (Eds.), *Aquatic Fauna of the Northern Chihuahuan Desert Contributed Papers from a Special*

Session, Thirty-Third Annual Symposium, Desert Fishes Council, 17 November 2001. Museum of Texas Tech University, Special Publication Number 46, Lubbock, TX.

- Hubbs, C. 2014. Differences in spring vs stream fish assemblages. Pages 376–395. In Hoyt, C.A. and J. Karges (Eds.), *Proceedings of the Sixth Symposium on the Natural Resources of the Chihuahuan Desert Region*, October 14–17, 2004. Chihuahuan Desert Research Institute, Fort Davis, Texas.
- Hubbs, C., R.J. Edwards, and G.P. Garrett. 2008. *An Annotated Checklist of the Freshwater Fishes of Texas, with Keys to Identification of Species*, Second Edition. Texas Academy of Science, Edinburg, TX.
- Hubbs, C., R.R. Miller, R.J. Edwards, K.W. Thompson, E. Marsh, G.P. Garrett, G.L. Powell, D.J. Morris, and R.W. Zerr. 1977. Fishes inhabiting the Rio Grande, Texas and Mexico, between El Paso and the Pecos confluence. Pages 91–97. *Symposium on the Importance, Preservation and the Management of the Riparian Habitat*. U.S. Department of Agriculture, Forest Service, General Service Technical Report RM-43, Fort Collins, CO.
- Huber, A., S. Goodrich, and K. Anderson. 1999. Diversity with successional status in the pinyon–juniper/mountain mahogany/bluebunch wheatgrass community type near Dutch John, Utah. Pages 114–117. In: Monsen, S.B., and R. Stevens (Eds.), *Proceedings: Ecology and Management of Pinyon Juniper Communities within the Interior West*. RMRS-P-9. U.S. Department of Agriculture, Forest Service, Ogden, UT, USA.
- Hudnall, W. and J. Boxell. 2010. Pedogenesis of gypsum soils from gypseous materials. In *Proceedings of the 19th World Congress of Soil Science, Soil Solutions for a Changing World*, 1 – 6 August 2010, Brisbane, Australia.
- Huenneke, L.F. 1995. Ecological impacts of plant invasions in rangeland ecosystems. Abstracts, 48th Annual Meeting, Society for Range Management, Phoenix, AZ.
- Huenneke, L.F., J.P. Anderson, M. Remmenga, and W.H. Schlesinger. 2002. Desertification alters patterns of aboveground net primary production in Chihuahuan ecosystems. *Global Change Biology* 8:247–264.
- Huff, G.F. 2004a. Overview of the Hydrogeology of Saline Ground Water in New Mexico. In *Water Desalination and Reuse Strategies for New Mexico*, September 2004. Las Cruces, NM: New Mexico Water Resources Research Institute, pp. 21–34.
- Huff, G.F. 2004b. Review of Knowledge on the Occurrence, Chemical Composition, and Potential Use for Desalination of Saline Ground Water in Arizona, New Mexico, and Texas with a Discussion of Potential Future Study Needs, Reston, VA: U.S. Department of the Interior, U.S. Geological Survey, Open-File Report 2004-1197.
- Huff, G.F., and D.A. Chace. 2006. Knowledge and Understanding of the Hydrogeology of the Salt Basin in South-central New Mexico and Future Study Needs. U.S. Geological Survey, Open-File Report 2006-1358, Reston, VA.
- Huffman, D.W., M.T. Stoddard, C.M. McGlone, P.Z. Fule, W.W. Covington. 2006. *Ecosystem Restoration*. Final Report. Ecological Restoration Institute, Flagstaff, AZ, USA.
- Hulbert, J., T. H. Bonner, J. N. Fries, G. P. Garrett, and D. R. Pendergrass. 2007. Early development of the Devils River minnow, *Dionda diaboli* (Cyprinidae). *The Southwestern Naturalist* 52:378–385.
- Hultine, K.R., S.E. Bush, A.G. West, and J.R. Ehleringer. 2007. Population structure, physiology and ecohydrological impacts of dioecious riparian tree species of western North America. *Oecologia* 154:85–93. doi:10.1007/s00442-007-0813-0
- Humphrey, N. 1994. History, status and management of Lehmann Lovegrass. *Rangelands* 16:205–206.
- Humphrey, R.R. 1949. Fire as a means of controlling velvet mesquite, burroweed, and cholla on southern

- Arizona ranges. *Journal of Range Management* 2:175–182.
- Humphrey, R.R. 1953. The desert grassland, past and present. *Journal of Range Management* 6:159–164.
- Humphrey, R.R. 1958. The desert grassland: a history of vegetational change and an analysis of causes. *Botanical Review* 24:193–252.
- Humphrey, R.R. 1987. 90 years and 535 miles: vegetation changes along the Mexican Border. University of New Mexico Press, Albuquerque, NM, USA.
- Hunter, W.C., B.W. Anderson, and R.D. Ohmart. 1985. Summer avian community composition of Tamarix habitats in three southwestern desert riparian systems. Pages 128–134. In Johnson, R.R., C.D. Ziebell, D.R. Patron, P.F. Folliott, and R.H. Hamre (Eds.), *Riparian Ecosystems and Their Management: Reconciling Conflicting Uses*. U.S. Department of Agriculture, Rocky Mountain Research Station, General Technical Report RM-120, Fort Collins, CO.
- Hupy, C.M., W.G. Whitford, and E.C. Jackson. 2004. The effect of dominance by an alien grass species, Lehmann lovegrass, *Eragrostis lehmanniana*, on faunal pedoturbation patterns in North American Desert grasslands. *Journal of Arid Environments* 58:321–334. doi:10.1016/j.jaridenv.2003.08.004
- Hussain, M. and J.K. Warren. 1989. Nodular and enterolithic gypsum: the “sabkha-tization” of Salt Flat playa, west Texas. *Sedimentary Geology*, 64(1–3):13–24.
- Innes, R. 2013. *Odocoileus hemionus*, Fire Effects Information System [Online]. U.S. Department of Agriculture, Rocky Mountain Research Station, Fire Sciences Laboratory, Missoula, MT.
- Integrated Landscape Assessment Project (ILAP) State-and-Transition Model (STM). 2012. Documentation Arizona and New Mexico (Region 3) Arid Lands (non-forests). The Institute for Natural Resources, Oregon State University and Portland State University, Corvallis and Portland, OR, USA.
- Integrated Landscape Assessment Project (ILAP,) 2013. Integrated Landscape Assessment Project Final Report. The Institute for Natural Resources, Oregon State University and Portland State University, Corvallis and Portland, OR, USA.
- International Boundary and Water Commission (IBWC). 2013. 2013 Rio Grande Basin Summary Report. International Boundary and Water Commission, United States Section, Texas Clean Rivers Program, El Paso, TX.
- Israël, N.M.D., M.M. VanLandeghem, S. Denny, J. Ingle, and R. Patiño. 2014. Golden alga presence and abundance are inversely related to salinity in a high-salinity river ecosystem, Pecos River, USA. *Harmful Algae* 39:81–91.
- Jackson, I.I., J.R. McAuliffe, and B.A. Roundy. 1991. Desert restoration. *Restoration and Management Notes* 9:71–80.
- Jacobs, B.F., R.G. Gatewood, and C.D. Allen. 2002. Watershed restoration in degraded piñon-juniper woodlands: a paired watershed study 1996-1998, Final Report. Project 9706. U.S. Geological Survey, Biological Resources Division, Research/ NPS-Natural Resource Preservation Program, Los Alamos, NM, USA.
- Jaeger, K.L., J.D. Olden, and N.A. Pelland. 2014. Climate change poised to threaten hydrologic connectivity and endemic fishes in dryland streams. *Proceedings of the National Academy of Sciences of the United States of America* 111:1–6.
- Jameson, D.A. 1965. Arrangement and growth of pinyon and one-seed juniper trees. *Plateau* 37:121–127.
- Januchowski-Hartley, S.R., M. Diebel, P.J. Doran, and P.B. McIntyre. 2014. Predicting road culvert passability for migratory fishes. *Diversity and Distributions* 20:1414–1424.

- Januchowski-Hartley, S.R., P.B. McIntyre, M. Diebel, P.J. Doran, D.M. Infante, C. Joseph, and J.D. Allan. 2013. Restoring aquatic ecosystem connectivity requires expanding inventories of both dams and road crossings. *Frontiers in Ecology and the Environment* 11:211–217.
- Johnsen, T.N. 1962. One-seed juniper invasion of northern Arizona grasslands. *Ecological Monographs* 32:187–207.
- Johnson, K.A. 2002. *Juniperus monosperma* [WWW Document]. Fire Effects Information System. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. URL <http://www.fs.fed.us/database/feis/plants/tree/junmon/all.html> (accessed 11.18.15).
- Johnson, M.S., and D.H. Rappuhn. 2002. Hydrogeology and Preliminary Simulation of Ground-Water Flow in the Lower Animas and Lordsburg Basins, Grant and Hidalgo Counties, New Mexico. New Mexico Office of the State Engineer, Technical Division, Hydrology Report 02-06, Santa Fe, NM.
- Johnson, P.D., Bogan, A.E., Brown, K.M., Burkhead, N.M., Cordeiro, J.R., Garner, J.T., Hartfield, P.D., Lepitzki, D.A.W., Mackie, G.L., Pip, E., Tarpley, T.A., Tiemann, J.S., Whelan, N. V., Strong, E.E., 2013. Conservation status of freshwater gastropods of Canada and the United States. *Fisheries* 38:247–282. doi:10.1080/03632415.2013.785396
- Johnson, R.R., and L.T. Haight. 1985. Avian use of xeroriparian ecosystems in the North American Warm Deserts. Pages 156–160. In Johnson, R.R., C.D. Ziebell, D.R. Patron, P.F. Folliort, and R.H. Hamre (Eds.), *Riparian Ecosystems and Their Management: Reconciling Conflicting Uses*. U.S. Department of Agriculture, Rocky Mountain Research Station, General Technical Report RM-120, Fort Collins, CO.
- Johnston, C.A., J.B. Zedler, M.G. Tulbure, C.B. Frieswyk, B.L. Bedford, and L. Vaccaro. 2009. A unifying approach for evaluating the condition of wetland plant communities and identifying related stressors. *Ecological Applications* 19:1739–57.
- Johnston, M.C. 1977. Brief resume of botanical, including vegetational, features of the Chihuahuan Desert Region with special emphasis on their uniqueness. Pages 335-362. In: Wauer, R.H., and D.H. Riskind (Eds.), *Transactions of the Symposium on the Biological Resources of the Chihuahuan Desert Region, U.S. and Mexico*. National Park Service Transactions and Proceedings, Series 3. Alpine, Texas, USA, 1974. US Department of Interior, National Park Service, Washington, D.C., USA.
- Jones, A. 2000. Effects of cattle grazing on North American arid ecosystems: a quantitative review. *Western North American Naturalist* 60:155–164.
- Jones, J., and J.O. Woods (Eds.). 2013. *A Statewide Assessment of Arizona's Streams, July 2007-June 2010*. Arizona Department of Environmental Quality, ADEQ Water Quality Division, Surface Water Section, Monitoring Unit, Open File Report 13-01, Phoenix, AZ.
- Jones, K.B., E.T. Slonecker, M.S. Nash, A.C. Neale, T.G. Wade, and S. Hamann. 2010. Riparian habitat changes across the continental United States (1972–2003) and potential implications for sustaining ecosystem services. *Landscape Ecology* 25:1261–1275.
- Jones, W.T. 1986. Survivorship in philopatric and dispersing kangaroo rats (*Dipodomys Spectabilis*). *Ecology* 67:202. doi:10.2307/1938519
- Jones, W.T. 1984. Natal philopatry in banner-tailed kangaroo rats. *Behavioral Ecology and Sociobiology* 15, 151–155. doi:10.1007/BF00299383
- Jones, W.T. P.M. Waser, L.F. Elliott, N.E. Link, and B.B. Bush. 1988. Philopatry, dispersal, and habitat saturation in the banner-tailed kangaroo rat, *Dipodomys Spectabilis*. *Ecology* 69:1466. doi:10.2307/1941643
- Jones, Z.F., C.E. Bock, and J.H. Bock. 2003. Rodent communities in a grazed and ungrazed Arizona grassland, and a model of habitat relationships among rodents in southwestern grass/shrublands. *The American*

Midland Naturalist 149:384–394. doi:10.1674/0003-0031(2003)149[0384:RCIAGA]2.0.CO;2

- Joule, J., and G.N. Cameron. 1975. Species removal studies. I. dispersal strategies of sympatric *Sigmodon hispidus* and *Reithrodontomys fulvescens* populations. *Journal of Mammalogy* 56:378–396. doi:10.2307/1379368
- Jurwitz, L.R., and P.C. Kangieser. 1978. Climates of the states, Arizona. Pages 48-67. In: *Climates of the States: National Oceanic and Atmospheric Administration Narrative Summaries, Tables, and Maps for Each State with Current Tables of Normals, 1941-1970, Means and Extremes to 1975, Overview of State Climatological Programs, Vol. 1.* Gale Research, Detroit, MI, USA.
- Kapuscinski, A.R., and T.J. Patronski. 2005. *Genetic Methods for Biological Control of Non-Native Fish in the Gila River Basin.* University of Minnesota, Institute for Social, Economic and Ecological Sustainability, Minnesota Sea Grant Publication F 20, St. Paul, MN.
- Karatayev, A.Y., T.D. Miller, and L.E. Burlakova. 2012. Long-term changes in unionid assemblages in the Rio Grande, one of the World's top 10 rivers at risk. *Aquatic Conservation: Marine and Freshwater Ecosystems* 22:206–219.
- Karges, J. 2007. Diamond Y Spring and marshland: A rare desert aquatic system and refuge of biodiversity. *Chihuahuan Desert Discovery* 53:19–32.
- Karr, J.R., and E.W. Chu. 1999. *Restoring Life in Running Waters: Better Biological Monitoring.* Island Press, Washington, D.C.
- Katz, G.L., J.C. Stromberg, and M.W. Denslow. 2009. Streamside herbaceous vegetation response to hydrologic restoration on the San Pedro River, Arizona. *Ecohydrology* 2:213–225.
- KellerLynn, K. 2003. *Geoinicators Scoping Report for White Sands National Monument, Strategic Planning Goal Ib4.* National Park Service, Geologic Resources Division.
- KellerLynn, K. 2008. *Geologic Resource Evaluation Scoping Summary: White Sands National Monument.* National Park Service, Geologic Resources Division.
- KellerLynn, K. 2012. *White Sands National Monument: Geologic Resources Inventory Report.* U.S. Department of the Interior, National Park Service, Natural Resource Program Center, Natural Resource Report NPS/NRSS/GRD/NRR–2012/585. Fort Collins, CO.
- Kelley, V.C. 1971. *Geology of the Pecos Country, Southeastern New Mexico, Socorro, NM.* New Mexico Bureau of Mines and Mineral Resources, Memoir 24, Socorro, NM.
- Kelly, M.E. 2001. *The Río Conchos: A Preliminary Overview.* Texas Center for Policy Studies, Austin, TX.
- Kelt, D.A., and T.J. Valone. 1995. Effects of grazing on the abundance and diversity of annual plants in Chihuahuan Desert scrub habitat. *Oecologia* 103:191–195.
- Kemp, P.R. 1983. Phenological patterns of Chihuahuan desert plants in relation to the timing of water availability. *Journal of Ecology* 71:427–436.
- Kennedy, J. R., and B. Gungle. 2010. *Quantity and Sources of Base Flow in the San Pedro River near Tombstone, Arizona.* U.S. Geological Survey, Scientific Investigations Report 2010–5200, Reston, VA.
- Kennedy, K.L., W.H. Moir, and L. Hendzel. 1983. A habitat type classification of the pinyon-juniper woodlands of the Lincoln National Forest, New Mexico, in: *Proceedings of the Workshop on Southwestern Habitat Types; Albuquerque, NM.* U.S. Department of Agriculture, Forest Service, Albuquerque, NM, USA, pp. 54–61.
- Kennedy, T.A., and S.E. Hobbie. 2004. Saltcedar (*Tamarix ramosissima*) invasion alters organic matter dynamics in a desert stream. *Freshwater Biology* 49:65–76.

- Kerley, G.I.H., and W.G. Whitford. 2000. Impact of grazing and desertification in the Chihuahuan Desert: plant communities, granivores and granivory. *The American Midland Naturalist* 144:78–91. doi:10.1674/0003-0031(2000)144[0078:IOGADI]2.O.CO;2
- Kerley, G.I.H., W.G. Whitford, and F.R. Kay. 1997. Mechanisms for the keystone status of kangaroo rats: graminivory rather than granivory? *Oecologia* 111:422–428. doi:10.1007/s004420050254
- Khumalo, G., J. Holechek, M. Thomas, and F. Molinar, F. 2007. Long-term vegetation productivity and trend under two stocking levels on Chihuahuan Desert rangeland. *Rangeland Ecology and Management* 60:165–171.
- Kilgore, A., E. Jackson, and W.G. Whitford. 2009. Fire in Chihuahuan Desert grassland: Short-term effects on vegetation, small mammal populations, and faunal pedoturbation. *Journal of Arid Environments* 73:1029–1034. doi:10.1016/j.jaridenv.2009.04.016
- King, A.J., P. Humphries, and P.S. Lake. 2003. Fish recruitment on floodplains: the roles of patterns of flooding and life history characteristics. *Canadian Journal of Fisheries and Aquatic Sciences* 60:773–786.
- Kinzli, K.D., and C.A. Myrick. 2010. Bendway weirs: Could they create habitat for the endangered Rio Grande silvery minnow. *River Research and Applications* 26:806–822.
- Kittams, W.H. 1973. Effects of fire on vegetation of the Chihuahuan Desert region, in: *Proceedings, Annual Tall Timbers Fire Ecology Conference: A Quest for Ecological Understanding, No. 12*. Lubbock, TX. Tall Timbers Research Station, Tallahassee, Florida, USA, pp. 427–444.
- Klise, G.T., V.C. Tidwell, M.D. Reno, B.D. Moreland, K.M Zemlick, and J. Macknick. 2013. *Water Use and Supply Concerns for Utility-Scale Solar Projects in the Southwestern United States*, Albuquerque, NM: Sandia National Laboratories, Report SAND2013-5238.
- Kocurek, G., M. Carr, R. Ewing, K.G. Havholm, Y.C. Nagar, and A.K. Singhvi. 2007. White Sands Dune Field, New Mexico: age, dune dynamics and recent accumulations. *Sedimentary Geology* 197:313–331.
- Kodric-Brown, A., C. Wilcox, J.G. Bragg, and J.H. Brown. 2007. Dynamics of fish in Australian desert springs: Role of large-mammal disturbance. *Diversity and Distributions* 13:789–798.
- Kodric-Brown, A., and J.H. Brown. 2007. Native fishes, exotic mammals, and the conservation of desert springs. *Frontiers in Ecology and the Environment* 5:549–553.
- Kolivras, K.N., and A.C. Comrie. 2004. Climate and infectious disease in the southwestern United States. *Progress in Physical Geography* 28:387–398. doi:10.1191/0309133304pp417ra
- Kominoski, J.S., and A.D. Rosemond. 2012. Conservation from the bottom up: forecasting effects of global change on dynamics of organic matter and management needs for river networks. *Freshwater Science* 31:51–68.
- Kondolf, G.M. 1997. Hungry water: Effects of dams and gravel mining on river channels. *Environmental Management* 21:533–51.
- Kondolf, G.M., J.G. Williams, T.C. Horner, and D. Milan. 2008. Assessing physical quality of spawning habitat. Pages 249–274. In Sear, D.A. and P. DeVries (Eds.), *Salmonid Spawning Habitat in Rivers: Physical Controls, Biological Responses, and Approaches*. American Fisheries Society Symposium 65. American Fisheries Society, Bethesda, MD.
- Koniak, S. 1985. Succession in pinyon-juniper woodlands following wildfire in the Great Basin. *Great Basin Naturalist* 45:556–566.
- Konieczki, A.D. 2006. Investigation of the Hydrologic Monitoring Network of the Willcox and Douglas Basins of Southeastern Arizona: A Project of the Rural Watershed Initiative (Fact Sheet No. 2006--3055). US

Department of the Interior, US Geological Survey.

- Kotliar, N.B., B.W. Baker, A.D. Whicker, and G. Plumb. 1999. "A Critical Review of Assumptions About the Prairie Dog as a Keystone Species." *Environmental Management* 24 (2): 177–92.
- Kozma, J.M., and N.E. Mathews. 1997. Breeding bird communities and nest plant selection in Chihuahuan desert habitats in south-central New Mexico. *Wilson Bulletin* 109:424–436.
- Kraft, J.P., and P. Stapp. 2013. "Movements and Burrow Use by Northern Grasshopper Mice as a Possible Mechanism of Plague Spread in Prairie Dog Colonies." *Journal of Mammalogy* 94 (5): 1087–93.
- Krammes, J.S. 1990. Effects of fire management of southwestern natural resources (General Technical Report RM-191). US Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Fort Collins, CO, USA.
- Krausman, P.R. 1978. Forage Relationships between Two Deer Species in Big Bend National Park, Texas. *The Journal of Wildlife Management* 42:101. doi:10.2307/3800695
- Krausman, P.R., A.J. Kuenzi, R.C. Etchberger, K.R. Rautenstrauch, L.L. Ordway, and J.J. Herver, J.J. 1997. Diets of desert mule deer. *Journal of Range Management* 50:513–522.
- Krausman, P.R., B.D. Leopold. 1988. Influence of Large Herbivores on Chihuahuan Desert Vegetation. *The Southwestern Naturalist* 33:107. doi:10.2307/3672100
- Krausman, P.R., B.D. Leopold, R.F. Seegmiller, S.G. Torres. 1989. Relationship between desert bighorn sheep and habitat in western Arizona. *Wildlife Monographs* 102:3–66.
- Krausman, P.R., K.R. Rautenstrauch, and B.D. Leopold. 1985. Xeroriparian systems used by mule deer in Texas and Arizona (No. USDA Forest Service General Technical Report TM-120). Tucson, Arizona.
- Krogh, S.N., M.S. Zeisset, E. Jackson, and W.G. Whitford. 2002. Presence/absence of a keystone species as an indicator of rangeland health. *Journal of Arid Environments* 50:513–519. doi:10.1006/jare.2001.0900
- Krueper, D., J. Bart, and T.D. Rich. 2003. Response of vegetation and breeding birds to the removal of cattle on the San Pedro River, Arizona (U.S.A.). *Conservation Biology* 17:607–615.
- Kunkel, K.E., L.E. Stevens, S.E. Stevens, L. Sun, E. Janssen, D. Wuebbles, K.T. Redmond, J.G. Dobson. 2013a. Regional Climate Trends and Scenarios for the U.S. National Climate Assessment: Part 5. Climate of the Southwest U.S. National Environmental Satellite, Data, and Information Service, National Oceanic and Atmospheric Administration, Washington, D.C.
- Kunkel, K.E., L.E. Stevens, S.E. Stevens, L. Sun, E. Janssen, D. Wuebbles, M.C. Kruk, D.P. Thomas, M.D. Shulski, N.A. Umphlett, K.G. Hubbard, K. Robbins, L. Romolo, A. Akyuz, T.B. Pathak, T.R. Bergantino, and J.G. Dobson. 2013b. Regional Climate Trends and Scenarios for the U.S. National Climate Assessment: Part 4. Climate of the U.S. Great Plains. National Environmental Satellite, Data, and Information Service, National Oceanic and Atmospheric Administration, Washington, D.C.
- Kupferberg, S.J. 1997. The role of larval diet in anuran metamorphosis. *American Zoology* 159:146–159.
- Ladwig, L.M. 2014. Abiotic Drivers of Chihuahuan Desert Plant Communities. Dissertation. University of New Mexico, Albuquerque, NM.
- Land, L. 2013. Evaporite Karst in the Permian Basin Region of West Texas and Southeastern New Mexico: the Human Impact. In National Cave and Karst Research Institute 13th Sinkhole Conference, Symposium 2. Carlsbad, NM: National Cave and Karst Research Institute, pp. 113–121.
- Land, L. and G.F. Huff. 2010. Multi-tracer investigation of groundwater residence time in a karstic aquifer: Bitter Lakes National Wildlife Refuge, New Mexico, USA. *Hydrogeology Journal*, 18(2):455–472.

- Land, L. and G. Veni. 2012. Electrical resistivity surveys of anthropogenic karst phenomena, southeastern New Mexico. *New Mexico Geology*, 34(4):17–125.
- LANDFIRE. 2014. Landfire program. US Department of Agriculture and US Department of Interior. URL <http://www.landfire.gov>.
- Landis, A.G., and J.D. Bailey. 2005. Reconstruction of age structure and spatial arrangement of pinyon-juniper woodlands and savannas of Anderson Mesa, Arizona. *Forest Ecology and Management* 204: 221–236.
- Lang, B. and D.C. Rogers. 2002. Biodiversity Survey of Large Branchiopod Crustacea in New Mexico, Santa Fe, NM: Bureau of Land Management, New Mexico State Office, Completion Report for Assistance Agreement No. GDA000013, Task Order No. 001.
- Lang, B.K., V. Gervasio, D.J. Berg, S.I. Guttman, N.L. Allan, M.E. Gordon, and G. Warrick. 2003. Gammarid amphipods of northern Chihuahuan desert spring systems: An imperiled fauna. Pages 47-57. In: G.P. Garrett and N.L. Allan(Eds.), *Aquatic Fauna of the Northern Chihuahuan Desert Contributed Papers from a Special Session, Thirty-Third Annual Symposium, Desert Fishes Council, 17 November 2001*. Lubbock, TX: Museum of Texas Tech University, Special Publication Number 46. Available at: http://www.desertfishes.org/dfc/proceed/2001/Chih_desert_symp/AFNCD_index.html.
- Langford, R.P. 2000. Nabkha (coppice dunes) fields of south-central New Mexico, USA. *Journal of Arid Environments* 46:25–31.
- Langford, R.P. 2003. The Holocene history of the White Sands dune field and influences on eolian deflation and playa lakes. *Quaternary International*, 104:31–39.
- Larrison, E.J., and D.R. Johnson. 1973. Density changes and habitat affinities of rodents of shadscale and sagebrush associations. *The Great Basin Naturalist* 255–264.
- Larson, F.R. 1980. Pinyon juniper 239. Pages 116-117. In: Eyre, F.H. (Ed.), *Forest Cover Types of the United States and Canada*. Society of America Foresters, Washington, D.C., USA.
- Lawrence, R.K. 1995. Population dynamics and habitat use of desert mule deer in the Trans-Pecos region of Texas (Dissertation). Texas Tech University, Lubbock, TX.
- Lawrence, R.K., S. Demarais, R.A. Relyea, S.P. Haskell, W.B. Ballard, and T.L. Clark. 2004. Desert mule deer survival in southwest Texas. *Journal of Wildlife Management* 68:561–569.
- Leenhouts, J.M., J.C. Stromberg, R.L. Scott (Eds.), 2006. *Hydrologic Requirements of and Consumptive Ground-Water Use by Riparian Vegetation along the San Pedro River, Arizona*. U.S. Geological Survey Scientific Investigations Report 2005–5163, Reston, VA.
- Leopold, A. 1924. Grass, brush, timber and fire in southern Arizona. *Journal of Forestry* 22:1–10.
- Leopold, B.D., P.R. Krausman. 1991. Factors influencing desert mule deer distribution and productivity in southwestern Texas. *The Southwestern Naturalist* 36:67–74.
- Levick, L., J. Fonseca, D. Goodrich, M. Hernandez, D. Semmens, J. Stromberg, R. Leidy, M. Scianni, D.P. Guertin, M. Tluczek, and W. Kepner. 2008. *The Ecological and Hydrological Significance of Ephemeral and Intermittent Streams in the Arid and Semi-arid American Southwest*. Environmental Protection Agency, EPA/600/R-08/134, Washington, D.C.
- Levings, G.W., D.F. Healy, S.F. Richey, and L.F. Carter. 1998. *Water Quality in the Rio Grande Valley, Colorado, New Mexico, and Texas, 1992-95*. U.S. Geological Survey, Circular 1162, Reston, VA.
- Ligon, J.D. 1978. Reproductive interdependence of pinon jays and pinon pines. *Ecological Monographs* 48:111–126.

- Lingle, S. 2002. Coyote Predation and Habitat Segregation of White-Tailed Deer and Mule Deer. *Ecology* 83:2037–2048. doi:10.2307/3071785
- Linsdale, J.M. 1946. The California ground squirrel: a record of observations made on the Hastings Natural History Reservation. University of California Press, Berkeley, California.
- Linzey, A., and R. Timm. 2008. *Dipodomys spectabilis* (banner-tailed kangaroo rat) [WWW Document]. The IUCN Red List of Threatened Species. URL <http://www.iucnredlist.org/details/6693/0> (accessed 7.4.15).
- Lite, S.J., and J.C. Stromberg. 2005. Surface water and ground-water thresholds for maintaining Populus–Salix forests, San Pedro River, Arizona. *Biological Conservation* 125:153–167.
- Lloyd, N., A. Moehrensclager, D.H.V. Smith, and D. Bender, D. 2013. Food limitation at species range limits: Impacts of food availability on the density and colony expansion of prairie dog populations at their northern periphery. *Biological Conservation* 161:110–117. doi:10.1016/j.biocon.2013.03.008
- Loeser, M.R., S.D. Mezulis, T.D. Sisk, T.C. Theimer, T.C. 2005. Vegetation cover and forb responses to cattle exclusion: implications for pronghorn. *Rangeland Ecology & Management* 58:234–238.
- Lohmiller, R.G. 1963. Drought and its effect on condition and production of a desert grassland range (Thesis). New Mexico State University, Las Cruces, NM, USA.
- Lomas, L.A., and L.C. Bender. 2007. Survival and Cause-Specific Mortality of Neonatal Mule Deer Fawns, North-Central New Mexico. *Journal of Wildlife Management* 71:884–894. doi:10.2193/2006-203
- Lopez-Gonzalez, C., and D.F. Garcia-Mendoza. 2012. A checklist of the mammals (Mammalia) of Chihuahua, Mexico. *Check List* 8:1122–33.
- Loring, S.J., MacKay, W.P., Whitford, W.G. 1988. Ecology of small desert playas, Small Water Impoundments in Semi-arid Regions. University of New Mexico Press, Albuquerque, New Mexico.
- Lougheed, V.L., and R. Rodriguez. 2008. Creation of a Chihuahuan Desert Bi-National Wetland: A Feasibility Assessment. New Mexico State University, Unpublished Report, Las Cruces, NM.
- Lucas, R.W., T.T. Baker, M.K. Wood, C.D. Allison, and D.M. VanLeeuwen. 2009. Streambank morphology and cattle grazing in two montane riparian areas in western New Mexico. *Journal of Soil and Water Conservation* 64:183–189.
- Luce, C., P. Morgan, K. Dwire, D. Isaak, Z. Holden, and B. Rieman. 2012. Climate Change, Forests, Fire, Water, and Fish: Building Resilient Landscapes, Streams, and Managers. U.S. Department of Agriculture, Forest Service, Rocky Mountain Field Station, General Technical Report RMRS-GTR-290, Fort Collins, CO.
- Lymbery, G.A., and R.D. Pieper. 1983. Ecology of pinyon-juniper vegetation in the northern Sacramento Mountains. Bulletin 698. New Mexico State University, Agricultural Experiment Station, Las Cruces, NM, USA.
- Lyon, L.J., R.G. Hooper, E.S. Telfer, D.S., Schreiner. 2000. Fire Effects on Wildlife Foods (No. RMRS-GTR-42-volume 1), Wildland Fire in Ecosystems: Effects of Fire on Fauna. U.S. Department of Agriculture, U.S. Forest Service, Rocky Mountain Research Station, Ogden, UT, USA.
- Mace, R.E., W.F. Mullican III, and E.S. Angle (Eds.) 2001. Aquifers of West Texas, Austin, TX: Texas Water Development Board Report 356. Available at: https://www.twdb.texas.gov/publications/reports/numbered_reports/doc/R356/356_AquifersofWestTexas.pdf.
- Macedo, R.H., and M.A. Mares. 1988. *Neotoma albigula*. *Mammalian Species* 310:1–7.
- Macías-Duarte, A., A.B. Montoya, C.E. Méndez-González, J.R. Rodríguez-Salazar, W.G. Hunt, and P.G.

- Krannitz. 2009. Factors Influencing Habitat Use by Migratory Grassland Birds in the State of Chihuahua, Mexico. *The Auk* 126:896–905. doi:10.1525/auk.2009.08251
- Macías-Duarte, A., and A.O. Panjabi. 2013. Association of Habitat Characteristics with Winter Survival of a Declining Grassland Bird in Chihuahuan Desert Grasslands of Mexico. *The Auk* 130:141–149. doi:10.1525/auk.2012.12047
- MacKay, W.P., S.J. Loring, T.M. Frost, and W.G. Whitford. 1990. Population dynamics of a playa community in the Chihuahuan Desert. *The Southwestern Naturalist* 35:393–402. doi:10.2307/3672036
- MacKay, W.P., J. Zak, and W.G. Whitford. 1992. Litter decomposition in a Chihuahuan Desert Playa. *American Midland Naturalist* 128:89–94. doi:10.2307/2426416
- MacRae, R.K., J.D. Lusk, and W. Radke. 2001. Investigation of the Role of Environmental Contaminants upon Ecological Sentinel Species and Their Habitats at Bitter Lake National Wildlife Refuge, New Mexico, Albuquerque, NM: U.S. Department of the Interior, Fish and Wildlife Service, Region 2, Project ID#2N27 9620001A.
- Mack, R.N. 1985. Invading plants: their potential contribution to population biology. Pages 127-142. In: White, J. (Ed.), *Studies on Plant Demography: A Festschrift for John L. Harper*. London.
- Mack, R.N., and J.N. Thompson. 1982. Evolution in steppe with few large, hooved mammals. *American Naturalist* 119(6): 757-773.
- Magaña, H.A. 2009. Feeding preference of the Rio Grande silvery minnow (*Hybognathus amarus*). *Reviews in Fisheries Science* 17:468–477.
- Magaña, H.A. 2013. Flood pulse trophic dynamics of larval fishes in a restored arid-land, river-floodplain, Middle Rio Grande, Los Lunas, New Mexico. *Reviews in Fish Biology and Fisheries* 23:507–521.
- Magle, S.B., and K.R. Crooks. 2009. “Investigating the Distribution of Prairie Dogs in an Urban Landscape.” *Animal Conservation* 12 (3): 192–203.
- Magle, S.B., and L.M. Angeloni. 2011. “Effects of Urbanization on the Behaviour of a Keystone Species.” *Behaviour* 148 (1): 31–54.
- Magoulick, D.D., and R.M. Kobza. 2003. The role of refugia for fishes during drought: a review and synthesis. *Freshwater Biology* 48:1186–1198.
- Magruder, I.A., W.W. Woessner, and S.W. Running. 2009. Ecohydrologic process modeling of mountain block groundwater recharge. *Ground Water* 47:774–785.
- Mainston, C.P., and W. Parr. 2002. Phosphorus in rivers-ecology and management. *The Science of the Total Environment* 282–283:25–47.
- Maker, H.J., H.E. Dregne, V.G. Link, and J.U. Anderson. 1974. *Soils of New Mexico*. Las Cruces NM: New Mexico State University Agricultural Experiment Station Research Report 285. 132 pp.
- Makings, E. 2005. Flora of the San Pedro Riparian National Conservation Area, Cochise County, Arizona. Pages 92–99. In Gottfried, G.J., B.S. Gebow, L.G. Eskew, and C.B. Edminster, (Eds.), *Connecting Mountain Islands and Desert Seas: Biodiversity and Management of the Madrean Archipelago II*, May 11-15, 2004, Tucson, AZ. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Proceedings 36, Fort Collins, CO.
- Malcom, J.W., and W.R. Radke. 2008. Effects of riparian and wetland restoration on an avian community in southeast Arizona, USA. *The Open Conservation Biology Journal* 2:30–36.
- Manahan, S.E. 1991. *Environmental Chemistry*. Lewis Publishers, Inc., Chelsea, MI.
- Marquez, M.J. 2008. The foraging ecology of an introduced species *Oryx gazella*: using stable isotopes of

carbon and nitrogen (Master of Science). New Mexico State University, Las Cruces, NM.

- Marquez, M.J., and W.J. Boecklen. 2010. Isotopic shift in an introduced population of gemsbok (*Oryx gazella*). *Journal of Arid Environments* 74:928–932. doi:10.1016/j.jaridenv.2010.01.004
- Marshall, J.P., V.C. Bleich, N.G. Andrew, and P.R. Krausman, P.R. 2004. Seasonal forage use by desert mule deer in southeastern California. *The Southwestern Naturalist* 49:501–505.
- Marshall, J.P., V.C. Bleich, P.R. Krausman, M.L. Reed, and N.G. Andrew, N.G. 2006. Factors affecting habitat use and distribution of desert mule deer in an arid environment. *Wildlife Society Bulletin* 34:609–619.
- Marshall, J.P., P.R. Krausman, and V.C. Bleich. 2005. Rainfall, temperature, and forage dynamics affect nutritional quality of desert mule deer forage. *Rangeland Ecology & Management* 58:360–365. doi:10.2111/1551-5028(2005)058[0360:RTAFDA]2.0.CO;2
- Marshall, J.T., Jr. 1957. Birds of pine-oak woodland in southern Arizona and adjacent Mexico (No. 32). Cooper Ornithological Society, Pacific Coast Avifauna, Berkeley, California.
- Martens, S.N., Breshears, D.D., Barnes, F.J. 2001. Development of species dominance along an elevational gradient: population dynamics of *Pinus edulis* and *Juniperus monosperma*. *International Journal of Plant Sciences* 162:777–783.
- Martin, A.C., H.S. Zim, A.L. Nelson. 1961. American wildlife and plants: a guide to wildlife food habits: the use of trees, shrubs, weeds, and herbs by birds and mammals of the United States. Dover Publications, New York, NY, USA.
- Martin, S.C. 1975. Ecology and management of southwestern semidesert grass-shrub ranges: the status of our knowledge. Research Paper RM-156, US Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experimental Station, Fort Collins, CO, USA.
- Martinez, P.J. 2012. Invasive crayfish in a high desert river: Implications of concurrent invaders and climate change. *Aquatic Invasions* 7:219–234.
- Maser, C., and Z. Maser. 1987. Notes on mycophagy in four species of mice in the genus *Peromyscus*. *The Great Basin Naturalist*. Pages 308–313.
- Mason, L.C., M.J. Desmond, and M.S. Agudelo. 2005. Influence of grassland type, nest type, and shrub encroachment on predation of artificial nests in Chihuahuan Desert grasslands. *Western North American Naturalist* 65:196–201.
- Mata-González, R., B. Figueroa-Sandoval, F. Clemente, and M. Manzano. 2007. Vegetation changes after livestock grazing exclusion and shrub control in the southern Chihuahuan desert. *Western North American Naturalist* 67:63–70. doi:10.3398/1527-0904(2007)67[63:VCALGE]2.0.CO;2
- Mathis, V.L., W.G. Whitford, F.R. Kay, and P.U. Alkon. 2006. Effects of grazing and shrub removal on small mammal populations in southern New Mexico, USA. *Journal of Arid Environments* 66:76–86. doi:10.1016/j.jaridenv.2005.10.015
- Maupin, M.A., J.F. Kenny, S.S. Hutson, J.K. Lovelace, N.L. Barber, and K.S. Linsey. 2014. Estimated Use of Water in the United States in 2010. U.S. Geological Survey, Circular 1405. Reston, VA.
- McAuliffe, J.R. 1994. Landscape evolution, soil formation and ecological patterns and processes in Sonoran Desert bajadas. *Ecological Monographs* 64:111–148.
- McClaran, M.P. 1995. Desert grasslands and grasses. Pages 1-103. In: McClaran, M.P., and T.R. Van Devender (Eds.), *The Desert Grassland*. The University of Arizona Press, Tucson, AZ, USA.
- McClaran, M.P., and M.E. Anable. 1992. Spread of introduced Lehmann lovegrass among a grazing intensity gradient. *The Journal of Applied Ecology* 29:92–98. doi:10.2307/2404352

- McCluney, K.E., and J.L. Sabo. 2012. River drying lowers the diversity and alters the composition of an assemblage of desert riparian arthropods. *Freshwater Biology* 57:91–103.
- McCluney, K.E., and J.L. Sabo. 2014. Sensitivity and tolerance of riparian arthropod communities to altered water resources along a drying river. *PLoS one* 9:e109276.
- McCraw, D.J. 2008. Preliminary Geologic Map of the South Spring Quadrangle, Chaves County, New Mexico, Socorro, NM: New Mexico Bureau of Geology and Mineral Resources, Open-file Digital Geologic Map OF-GM 171.
- McCraw, D.J., G. Rawling, and L.A. Land. 2007. Geologic Map of the Bitter Lake Quadrangle, Chaves County, New Mexico, Socorro, NM: New Mexico Bureau of Geology and Mineral Resources, Open-file Digital Geologic Map OF-GM 15.
- McCulloch, C.Y. 1969. Some effects of wildfire on deer habitat in pinyon-juniper woodland. *Journal of Wildlife Management* 33:778–784.
- McDonald, A. 2012. Mechanical brush management in Trans-Pecos, Texas, in: Proceedings of the Trans-Pecos Wildlife Conference-2012. Presented at the August 2-3, 2012 Trans-Pecos Wildlife Conference, Alpine, TX.
- McDonald, C., and G. McPherson, G. 2011. Absence of a grass/fire cycle in a semiarid grassland: response to prescribed fire and grazing. *Rangeland Ecology and Management* 64:384–393.
- McFarland, D.G., L.S. Nelson, M.J. Grodowitz, R.M. Smart, and C.S. Owens. 2004. *Salvinia molesta* D. S. Mitchell (Giant Salvinia) in the United States: A Review of Species Ecology and Approaches to Management. U.S. Army Engineer Research and Development Center, Environmental Laboratory, Aquatic Plant Control Research Program. ERDC/EL SR-04-2, Vicksburg, MS.
- McGlone, C.M. 2013. No long-term effects of prescribed fire on Lehmann lovegrass (*Eragrostis lehmanniana*)–Invaded Desert Grassland. *Invasive Plant Science and Management* 6:449–456. doi:10.1614/IPSM-D-12-00059.1
- McGlone, C.M., and L.F. Huenneke. 2004a. The impact of a prescribed burn on introduced Lehmann lovegrass versus native vegetation in the northern Chihuahuan Desert. *Journal of Arid Environments* 57:297–310. doi:10.1016/S0140-1963(03)00109-5
- McGlone, C.M., L.F. Huenneke. 2004b. The impact of a prescribed burn on introduced Lehmann lovegrass versus native vegetation in the northern Chihuahuan Desert. *Journal of Arid Environments* 57:297–310. doi:10.1016/S0140-1963(03)00109-5
- McKinney, T., D.E. Brown, L. Allison. 2008. Winter precipitation and recruitment of pronghorns in Arizona. *The Southwestern Naturalist* 53:319–325.
- McLemore, V.T. 1999. New Mexico State Park Series, Bottomless Lakes. *New Mexico Geology* 21:51–55.
- McPherson, G.R. 2006. Fire ecology and management in grasslands of the American Southwest (No. RMRS-P-40), Grasslands ecosystems, endangered species, and sustainable ranching in the Mexico-U.S. borderlands. U.S. Forest Service, Fort Collins, CO, USA.
- McPherson, G.R. 1995. The role of fire in the desert grasslands. Pages 130-151. In: McClaran, M.P., and T.R. Van Devender (Eds.), *The Desert Grassland*. The University of Arizona Press, Tucson, AZ, USA.
- McPherson, G.R. 1997. Ecology and management of North American savannas. University of Arizona Press, Tucson, AZ, USA.
- McPherson, G.R., and J.F. Weltzin, J.F. 2000. Disturbance and climate change in United States/Mexico borderland plant communities: A state-of-the-knowledge review. (Rocky Mountain General Technical Report No. RMRS-GTR-50). USDA Forest Service, Collins, CO.

- Meagher, G.S. 1943. Reaction of pinon and juniper seedlings to artificial shade and supplemental watering. *Journal of Forestry* 41:480–482.
- Medina, A.L., and S.C. Martin. 1988. Stream channel and vegetation changes in sections of McKnight Creek, New Mexico. *Great Basin Naturalist* 48:373–381.
- Medley, C.N., and P.D. Shirey. 2013. Review and reinterpretation of Rio Grande silvery minnow reproductive ecology using egg biology, life history, hydrology, and geomorphology information. *Ecohydrology* 6:491–505.
- Meeuwig, R.O., and R.L. Bassett. 1983. Pinyon-juniper. Pages 84-86. In: Burns, R.M. (Ed.), *Silvicultural Systems for the Major Forest Types of the United States*. Agriculture Handbook 445. U.S. Department of Agriculture, Forest Service, Washington, D.C., USA.
- Meixner, T., A.H. Manning, D.A. Stonestrom, D.M. Allen, H. Ajami, K.W. Blasch, A.E. Brookfield, C.L. Castro, J.F. Clark, D.J. Gochis, A.L. Flint, K.L. Neff, R. Niraula, M. Rodell, B.R. Scanlon, K. Singha, and M.A. Walvoord. 2016. Implications of projected climate change for groundwater recharge in the western United States. *Journal of Hydrology* 534:124–138.
- Melillo, J.M., T.C. Richmond, and G.W. Yohe (Eds.) 2014. *Climate Change Impacts in the United States: The Third National Climate Assessment*. U.S. Global Change Research Program, Washington, D.C.
- Merkle, J. 1952. An analysis of a pinyon-juniper community at Grand Canyon, Arizona. *Ecology* 33:375–384.
- Merritt, D.M., and H.L. Bateman. 2012. Linking stream flow and groundwater to avian habitat in a desert riparian system. *Ecological Applications* 22:1973–1988.
- Merritt, D.M., and N.L.R. Poff. 2010. Shifting dominance of riparian *Populus* and *Tamarix* along gradients of flow alteration in western North American rivers. *Ecological Applications* 20:135–152.
- Merritt, D.M., M.L. Scott, N.L. Poff, G.T. Auble, and D.A. Lytle. 2010. Theory, methods and tools for determining environmental flows for riparian vegetation: riparian vegetation-flow response guilds. *Freshwater Biology* 55:206–225.
- Mesilla Valley Economic Development Alliance (MVEDA). Undated. NM Borderplex Regional Profile. <http://www.mveda.com/data-center/regional-profile/>. Accessed September 26, 2016.
- Metzler, E.H. 2014. The remarkable endemism of moths at White Sands National Monument in New Mexico, USA, with special emphasis on Gelechioidea (Lepidoptera). *Journal of Asia-Pacific Biodiversity* 7(1):e1-e5.
- Meyer, J.E., M.R. Wise, and S. Kalaswad. 2012. Pecos Valley Aquifer, West Texas: Structure and Brackish Groundwater. Texas Water Development Board Report 382, Austin, TX.
- Meyer, J.L., D.L. Strayer, J.B. Wallace, S.L. Eggert, G.S. Helfman, and N.E. Leonard. 2007. The contribution of headwater streams to biodiversity in river networks. *Journal of the American Water Resources Association* 43:86–103.
- Milholland, M.T., J.P. Shumate, T.R. Simpson, and R.W. Manning. 2010. Nutria (*Myocastor coypus*) in Big Bend National Park; A non-native species in desert wetlands. *Texas Journal of Science* 62:205–222.
- Miller, B., G. Ceballos, R. Reading. 1994. The prairie dog and biotic diversity. *Conservation Biology* 8: 677–681. doi:10.1046/j.1523-1739.1994.08030677.x
- Miller, B.J., R.P. Reading, D.E. Biggins, J.K. Detling, S.C. Forrest, J.L. Hoogland, J. Javersak, S.D. Miller, J. Proctor, J. Truett, and D.W. Uresk. 2007. “Prairie Dogs: An Ecological Review and Current Biopolitics.” *The Journal of Wildlife Management* 71 (8): 2801–10.
- Miller, D.M., S.P. Finn, A. Woodward, A. Torregrosa, M.E. Miller, D.R. Bedford, A.M. Brasher. 2010.

Conceptual Ecological Models to Guide Integrated Landscape Monitoring of the Great Basin. U.S. Geological Survey Scientific Investigations Report 2010-5133, 134 p. Reston, VA.

- Miller, M.E. 1999. Use of historic aerial photography to study vegetation change in the Negrito Creek watershed, southwestern New Mexico. *The Southwestern Naturalist* 44:121–137.
- Miller, R.E., L.F. Huenneke. 2000. The relationship between density and demographic variation within a population of *Larrea tridentata*. *Southwestern Naturalist* 45:313–321.
- Miller, R.F., J.D. Bates, T.J. Svejcar, F.B. Pierson, and L.E. Eddleman. 2005. Biology, ecology, and management of western juniper (*Juniperus occidentalis*), OSU Agricultural Experiment Station Technical Bulletin 152. Oregon State University, Corvallis, Oregon, USA.
- Miller, R.F., T.J. Svejcar, J.A. Rose. 2000. Impacts of western juniper on plant community composition and structure. *Journal of Range Management* 53: 574–585.
- Miller, R.F. and P.E. Wigand. 1994. Holocene changes in semiarid pinyon-juniper woodlands. *BioScience* 44:465–474.
- Miller, R.R. 1977. Composition and derivation of the native fish fauna of the Chihuahuan Desert region. Pages 365-382. In: Wauer, R. and D.H. Riskind (Eds.), *Transactions of the Symposium on the Biological Resources of the Chihuahuan Desert Region, United States and Mexico*. Washington, D.C.: U.S. Department of the Interior, National Park Service Transactions and Proceedings Series No. 3.
- Mills, K.W. 2005. Updated Evaluation for the Trans-Pecos Priority Groundwater Management Study Area, Austin, TX: Texas Commission on Environmental Quality, Priority Groundwater Management Area File Report March 2005.
- Milne-Laux, S., and R.A. Sweitzer. 2006. “Experimentally Induced Colony Expansion by Black-Tailed Prairie Dogs (*Cynomys Ludovicianus*) and Implications for Conservation.” *Journal of Mammalogy* 87 (2): 296–303.
- Milstead, W.W. 1960. Relict species of the Chihuahuan Desert. *The Southwestern Naturalist* 5:75-88.
- Minckley, T.A., D.S. Turner, and S.R. Weinstein. 2013. The relevance of wetland conservation in arid regions: A re-examination of vanishing communities in the American Southwest. *Journal of Arid Environments* 88:213–221.
- Minnick, T.J., Coffin, D.P. 1999. Geographic patterns of simulated establishment of two *Bouteloua* species: implications for distributions of dominants and ecotones. *Journal of Vegetation Science* 10, 343–356.
- Misztal, L.W., N. Deyo, and C.F. Campbell. 2013. Springs in the Sky Island Region: Inventory, Assessment, and Management Planning Project. Sky Island Alliance, Tucson, AZ.
- Mitsch, W.J., and J.G. Gosselink. 2007. *Wetlands*. 4th edition. John Wiley & Sons, Hoboken, NJ.
- Miyamoto, S., F. Yuan, S. Anand, W. Hatler, A. McDonald, G. Anaya, and W. Belzer. 2005. Reconnaissance Survey of Salt Sources and Loading into the Pecos River Report Sub., College Station, TX: Texas A&M University.
- Miyamoto, S., F. Yuan, and S. Anand. 2007. Water Balance, Salt Loading, and Salinity Control Options of Red Bluff Reservoir, Texas. Texas Water Resources Institute, Texas A&M University, Technical Report TR – 298, College Station, TX.
- Miyazono, S. 2014. Fish Metacommunity Dynamics and Threatened Species Ecology in a Desert River System. Ph.D. Dissertation, Texas Tech University, Wildlife, Aquatic, and Wildlands Science and Management, Lubbock, TX.
- Mohlhenrich, J.S. 1961. Distribution and ecology of the hispid and least cotton rats in New Mexico. *Journal*

of Mammalogy 42:13. doi:10.2307/1377236

- Molinar, F., J. Holechek, D. Galt, M. Thomas. 2002. Soil depth effects on Chihuahuan Desert vegetation. *Western North American Naturalist* 62:300–306.
- Molinar, F., J. Navarro, J. Holechek, D. Galt, and M. Thomas, M. 2011. Long-term vegetation trends on grazed and ungrazed Chihuahuan Desert rangelands. *Rangeland Ecology & Management* 64:104–108.
- Molles, M.C., and C.N. Dahm. 1990. A perspective on El Nino and La Nina: Global implications for stream ecology. *Journal of the North American Benthological Society* 9:68. doi:10.2307/1467935
- Monger, H.C. 1993. Soil-Geomorphic and Paleoclimatic Characteristics of the Fort Bliss Maneuver Areas, Southern New Mexico and Western Texas, Fort Bliss, TX: United States Army Air Defense Artillery Center, Fort Bliss, Texas, Directorate of Environment, Environmental Management Division, Cultural Resources Branch, Historic and Natural Resources Report No. 10.
- Monger, H.C., G.H. Mack, B.A. Nolen, and L.H. Gile. 2006. Regional setting of the Jornada Basin. Pages 15–43. In: Havstad, K.M., L.F. Huenneke, and W.H., Schlesinger (Eds.), *Structure and Function of a Chihuahuan Desert Ecosystem: the Jornada Basin Long-Term Ecological Research Site*. New York: Oxford University Press. <http://jornada.nmsu.edu/files/bibliography/06-053.pdf>. Accessed September 30, 2016.
- Monson, G. 1943. Food habits of the banner-tailed kangaroo rat in Arizona. *The Journal of Wildlife Management* 7:98. doi:10.2307/3795784
- Moody, E.K., and J.L. Sabo. 2013. Crayfish impact desert river ecosystem function and litter-dwelling invertebrate communities through association with novel detrital resources. *PLoS ONE* 8:e63274.
- Moore, M.J. 2015. The Origin and Evolution of Gypsum Endemic Plants. Oberlin College website, Biology Department. <http://www.oberlin.edu/faculty/mmoore/gypsophily.html>. Accessed October 17, 2016.
- Moore, M.J. and R.K. Jansen. 2007. Origins and Biogeography of Gypsophily in the Chihuahuan Desert Plant Group *Tiquilia* subg. *Eddya* (Boraginaceae). *Systematic Botany*, 32(2):392–414.
- Moore, M.J., J.F. Mota, N.A. Douglas, H.F. Olvera, and H. Ochoterena. 2015. The Ecology, Assembly and Evolution of Gypsophile Floras. Pages 97-128. In: Rajakaruna, N., R. Boyd, and T. Harris (Eds.), *Plant Ecology and Evolution in Harsh Environments*. Hauppauge, NY: Nova Science Publishers.
- Moorhead, D.L., F.M. Fisher, and W.G. Whitford. 1988. Cover of spring annuals on nitrogen-rich kangaroo rat mounds in a Chihuahuan desert grassland. *American Midland Naturalist* 120:443. doi:10.2307/2426018
- Mora, M., R. Skiles, B. McKinney, M. Paredes, D. Buckler, D. Papoulias, and D. Klein. 2002. Environmental contaminants in prey and tissues of the peregrine falcon in the Big Bend Region, Texas, USA. *Environmental Pollution* 116:169–176.
- Morafka, D.J. 1977. Is there a Chihuahuan Desert? A quantitative evaluation through a herpetofaunal perspective. Pages 437-454. In: Wauer, R.H., and D.H. Riskind (Eds.), *Transactions of the Symposium on the Biological Resources of the Chihuahuan Desert Region, U.S. and Mexico*. National Park Service Transactions and Proceedings, Series 3. Alpine, Texas, USA, 1974. US Department of Interior, National Park Service, Washington, D.C., USA.
- Moroka, N., R.F. Beck, and R.D. Pieper. 1982. Impact of burrowing activity of the bannertail kangaroo rat on southern New Mexico Desert rangelands. *Journal of Range Management* 35:707. doi:10.2307/3898244
- Morris, D.W. 2003. Toward an ecological synthesis: a case for habitat selection. *Oecologia* 136:1–13.
- Morton, H.L., and A. Melgoza. 1991. Vegetation Changes following Brush Control in Creosotebush Communities. *Journal of Range Management* 44:133–139.

- Moses, M.R., J.K. Frey, and G.W. Roemer. 2012. Elevated surface temperature depresses survival of banner-tailed kangaroo rats: will climate change cook a desert icon? *Oecologia* 168:257–268. doi:10.1007/s00442-011-2073-2
- Mosher, K.R., and H.L. Bateman. 2016. The effects of riparian restoration following saltcedar (*Tamarix* spp.) biocontrol on habitat and herpetofauna along a desert stream. *Restoration Ecology* 24:71–80.
- Moyle, P.B., and J.F. Mount. 2007. Homogenous rivers, homogenous faunas. *Proceedings of the National Academy of Sciences of the United States of America* 104:5711–2.
- Mueller, R.C., C.M. Scudder, M.E. Porter, R.T.I. Trotter, C.A. Gehring, and T.G. Whitham. 2005. Differential tree mortality in response to severe drought: evidence for long-term vegetation shifts. *Journal of Ecology* 93:1085–1093.
- Muldavin, E., P. Durkin, M. Bradley, M. Stuever, P. Mehlhop. 2000. *Handbook of Wetland Vegetation Communities of New Mexico Volume I: Classification and Community Descriptions*. New Mexico Natural Heritage Program, Albuquerque, New Mexico.
- Mungall, E.C. 2000. Exotics. Pages 736–764. In: *Ecology and Management of Large Mammals of North America*. Prentice-Hall, Upper Saddle River, NJ.
- Munson, S.M., E.H. Muldavin, J. Belnap, D.P.C. Peters, J.P. Anderson, M.H. Reiser, K. Gallo, A. Melegoza-Castillo, J.E. Herrick, T.A. Christiansen. 2013. Regional signatures of plant response to drought and elevated temperature across a desert ecosystem. *Ecology* 94:2030–2041.
- Murphy, A.L., A. Pavlova, R. Thompson, J. Davis, and P. Sunnucks. 2015. Swimming through sand: Connectivity of aquatic fauna in deserts. *Ecology and Evolution* 5:5252–5264.
- Nagler, P.L., T. Brown, K.R. Hultine, C. van Riper, D.W. Bean, P.E. Dennison, R.S. Murray, and E.P. Glenn. 2012. Regional scale impacts of Tamarix leaf beetles (*Diorhabda carinulata*) on the water availability of western U.S. rivers as determined by multi-scale remote sensing methods. *Remote Sensing of Environment* 118:227–240.
- Nagler, P.L., E.P. Glenn, C.S. Jarnevich, and P.B. Shafroth. 2011. Distribution and abundance of saltcedar and Russian olive in the western United States. *Critical Reviews in Plant Sciences* 30:508–523.
- Naranjo, L.G. and R.J. Raitt. 1993. Breeding Bird Distribution in Chihuahuan Desert Habitats. *The Southwestern Naturalist* 38:43–51. doi:10.2307/3671643
- Nash, M.H., Whitford, W.G. 1995. Subterranean termites: regulators of soil organic matter in the Chihuahuan Desert. *Biology and Fertility of Soils* 19:15–18.
- National Agricultural Statistics Service (NASS). 2013. 2012 Census of Agriculture: New Mexico, County Summary Highlights 2012. https://www.agcensus.usda.gov/Publications/2012/Full_Report/Volume_1,_Chapter_2_County_Level/New_Mexico/st35_2_001_001.pdf. Accessed September 29, 2016.
- National Centers for Environmental Information (NCEI). 2016. Normals Monthly Station Details: Rio Grande Village, TX. <https://www.ncdc.noaa.gov/cdo-web/>. Accessed November 2, 2016.
- National Parks of Canada (NPC). 2014. Ecological Integrity URL <http://www.pc.gc.ca/eng/progs/np-pn/ie-ei.aspx>.
- National Park Service (NPS). 2005. Geology Fieldnotes: White Sands National Monument. <http://www.nature.nps.gov/geology/parks/whsa/index.cfm>. Accessed October 7, 2016.
- National Park Service (NPS). 2016. White Sands National Monument Nature & Science. <https://www.nps.gov/whsa/learn/nature/index.htm>.

- National Park Service (NPS), Chihuahuan Desert Inventory and Monitoring Network (CDIMN). 2010. Chihuahuan Desert Network Vital Signs Monitoring Plan. Natural Resources Report NPS/CHDN/NRR—2010/188. National Park Service, Fort Collins, CO. Available at: https://irma.nps.gov/App/Reference/DownloadDigitalFile?code=152721&file=CHDN_MonPlan_FINAL_Publication.pdf.
- National Park Service (NPS). 2010. Chihuahuan Desert Network vital signs monitoring plan (Natural Resource Report No. Published Report-664523), NPS/Chihuahuan DesertN/NRR—2010/188. National Park Service Chihuahuan Desert Inventory and Monitoring Network, Fort Collins, CO.
- Natural Heritage New Mexico. 2015. NMBiotics Database. Museum of Southwestern Biology. URL <http://nhnm.unm.edu>. (accessed 8.5.15).
- NatureServe. 2015. NatureServe Explorer: An Online Encyclopedia of Life [web application] V7.1. URL <http://explorer.natureserve.org>
- NatureServe. 2014. NatureServe Explorer: An Online Encyclopedia of Life. NatureServe, Arlington, VA, USA. URL <http://explorer.natureserve.org>. Accessed: August 11, 2014.
- NatureServe. 2009. NatureServe Web Service. Arlington, VA. <http://services.natureserve.org>. Accessed: September 20, 2016.
- Navarro, J.M., D. Galt, J. Holechek, J. McCormick, and F. Molinar. 2002. Long-term impacts of livestock grazing on Chihuahuan Desert rangelands. *Journal of Range Management* 55:400. doi:10.2307/4003478
- Naylor, J.N. 1964. Plant distribution of the Sandia Mountains Area, New Mexico (Thesis). University of New Mexico, Albuquerque, NM, USA.
- Neal, J.T. 1975. Playas and dried lakes. Dowden, Hutchinson & Ross, New York, NY, USA.
- Negron, J.F. 1995. Cone and seed insects associated with piñon pine. Pages 97-106. In: Shaw, D.W., E.F. Aldon, and C. LoSapio (Eds.), *Desired Future Conditions for Piñon-Juniper Ecosystems*. Flagstaff, AZ, USA. GTR-RM-258. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Research Station, Fort Collins, CO, USA.
- Negron, J.F., and Wilson, J.L. 2003. Attributes associated with probability of infestation by the piñon ips, *Ips confusus* (Coleoptera: Scolytidae), in piñon pine, *Pinus edulis*. *Western North American Naturalist* 63:440–451.
- Neilson, R.P. 1986. High-resolution climatic analysis and southwest biogeography. *Science* 232:27–34.
- Nelle, S. 2006. Food habits of pronghorn in the Trans-Pecos. Pshrd 12-19. In: *Pronghorn Symposium*. Presented at the Pronghorn Symposium, Alpine, TX.
- Nelson, E.W. 1934. The influence of precipitation and grazing upon black grama grass range. Technical Bulletin 409. US Department of Agriculture, Forest Service, Division of Range Research. US Government Printing Office, Washington, DC, USA.
- Nelson, T., J. Holechek, and R. Valdez. 1999. Wildlife plant community preference in the Chihuahuan Desert. *Rangelands Archives* 21:9–11.
- New Mexico Demographics. 2016. New Mexico Counties by Population. http://www.newmexico-demographics.com/counties_by_population. Accessed September 27, 2106.
- New Mexico Department of Game and Fish (NMDGF). 2006. Comprehensive Wildlife Conservation Strategy (CWCS) for New Mexico. New Mexico Department of Game and Fish. Santa Fe, NM. 526 pp + appendices. http://www.wildlife.state.nm.us/conservation/comp_wildlife_cons_strategy/.

- New Mexico Department of Game and Fish (NMDGF). 2016. 2016 Statewide Fisheries Management Plan. New Mexico Department of Game and Fish (NMDGF), Santa Fe, NM.
- New Mexico Energy Forum. 2016. Hydraulic Fracturing. Available at: <http://www.nmenergyforum.com/topics/hydraulic-fracturing>.
- New Mexico Energy, Minerals and Natural Resources Department (NMEMNRD). 2016. "Oil and Gas Education." New Mexico Oil Conservation Division. Available at: <http://www.emnrd.state.nm.us/OCD/education.html#WOG2>.
- New Mexico Office of the State Engineer (NMOSE). 2013. Working Toward Solutions: Integrating Our Water and Our Economy - State Water Plan 2013 Review. New Mexico Office of the State Engineer, Santa Fe, NM.
- New Mexico Office of the State Engineer/Interstate Stream Commission (NMOSE). 2016a. Basins and Interstate Compacts: Rio Grande. <http://www.ose.state.nm.us/Basins/RioGrande/index.php>. Accessed January 4, 2017.
- New Mexico Office of the State Engineer/Interstate Stream Commission (NMOSE). 2016b. Basins and Interstate Compacts: Pecos River. http://www.ose.state.nm.us/Basins/Pecos/isc_Pecos.php. Accessed January 4, 2017.
- New Mexico Oil and Gas Association (NMOGA). 2012. Hydraulic Fracturing: Oil and Gas in New Mexico, Santa Fe, NM: New Mexico Oil and Gas Association. Available at: www.nmoga.org.
- New Mexico State University (NMSU). 1975. Nogal blackgrama (Circular No. 460), Cooperative Extension Service. New Mexico State University, Las Cruces, NM, USA.
- Noble, D.L. 1990. Rocky Mountain juniper. Pages 116-126. In: Burns, R.M., and B.H. Honkala (Eds.), *Silvics of North America: Conifers*. Agricultural Handbook 654. U.S. Department of Agriculture, Forest Service, Washington, D.C., USA.
- Norris, J.J. 1950. Effect of rodents, rabbits, and cattle on two vegetation types in semidesert range land (No. Bulletin 353). New Mexico State University Agricultural Experimental Station, Las Cruces, NM, USA.
- Nowak, R.M. 1999. *Walker's mammals of the world*, 6th ed. Johns Hopkins University Press, Baltimore, MD.
- Ockenfels, R.A. 1994. Mountain lion predation on pronghorn in Central Arizona. *The Southwestern Naturalist* 39:305.
- O'Gara, B.W. 2004. Diseases and Parasites. Pages 299-336. In: O'Gara, B.W., and J.D. Yoakum (Eds.), *Pronghorn Ecology and Management*. University Press of Colorado, Boulder, CO.
- O'Gara, B.W., and C.M. Janis. 2004. Scientific Classification. Pages 4-25. In: O'Gara, B.W., and J.D. Yoakum (Eds.), *Pronghorn Ecology and Management*. University Press of Colorado, Boulder.
- Ogle, K., T.G. Whitham, and N.S. Cobb. 2000. Tree-ring variation in pinyon predicts likelihood of death following severe drought. *Ecology* 81:3237–3243.
- Oldemeyer, J.L. and L.R. Allen-Johnson. 1988. Cattle grazing and small mammals on the Sheldon National Wildlife Refuge, Nevada. Pages 391-398. In: *Management of Amphibians, Reptiles, and Small Mammals in North America*. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Flagstaff, AZ, USA.
- Opperman, J.J. 2008. Sacramento-San Joaquin Delta Regional Ecosystem Restoration Implementation Plan, Ecosystem Conceptual Model, Floodplain. Sacramento-San Joaquin Delta Regional Ecosystem Restoration Implementation Plan, Sacramento, CA.
- Opperman, J.J. 2012. A conceptual model for floodplains in the Sacramento–San Joaquin Delta. San

- Ordway, L.L., and P.R. Krausman. 1986. Habitat Use by Desert Mule Deer. *The Journal of Wildlife Management* 50:677. doi:10.2307/3800980
- Oring, L.W., L. Neel, and K.E. Oring. 2013. Intermountain West Regional Shorebird Plan. Rev. 01/13. Manomet Center for Conservation Sciences, Manomet, MA.
- O'Rourke, J.T., and P.R. Ogden. 1969. Vegetative response following pinyon-juniper control in Arizona. *Journal of Range Management* 22:416–418.
- Ortega-Sanchez, A. 2013. Evaluation of a Translocated Population of Desert Mule Deer in the Chihuahuan Desert of Northern Coahuila, Mexico. Texas A&M University.
- O'Shaughnessy, R., T. Garrison, J. French, L.A. Harveson, and S. Gray. 2014. Restoration and management of pronghorn in Trans-Pecos, Texas. Borderlands Research Institute for Natural Resource Management, Alpine, TX.
- Padien, D.J., and K. Lajtha. 1992. Plant spatial pattern and nutrient distribution in pinyon-juniper woodlands along an elevational gradient in northern New Mexico. *International Journal of Plant Sciences* 153:425–433.
- Parcher, J.W., D.G. Woodward, and R.A. Durall. 2010. A Descriptive Overview of the Rio Grande-Rio Bravo Watershed. *Journal of Transboundary Water Resources* 1:159–177.
- Parker, K. 2002. Fire in the pre-European lowlands of the American southwest, in: Vale, T.R. (Ed.), *Fire, Native Peoples, and the Natural Landscape*. Island Press, Washington, DC, pp. 101–141.
- Parker, K.W., and S.C. Martin. 1952. The mesquite problem on southern Arizona ranges. Circular 908. US Department of Agriculture, Forest Service. US Government Printing Office, Washington, DC, USA.
- Parmenter, R.R. 2008. Long-term effects of a summer fire on desert grassland plant demographics in New Mexico. *Rangeland Ecology and Management* 61:156–168.
- Parmenter, R.R., and T.R. Van Devender. 1995. Diversity, spatial variability, and functional roles of vertebrates in the desert grassland. Pages 196-229. In: Van Devender, T.R. (Ed.), *The Desert Grassland*. Tucson, Arizona.
- Parmenter, R.R., E.P. Yadav, C.A. Parmenter, P. Ettestad, and K.L. Gage. 1999. Incidence of plague associated with increased winter-spring precipitation in New Mexico. *American Journal of Tropical Medicine and Hygiene* 61:814–821.
- Paroz, Y., J. Monzingo, and D. Propst. 2009. Inventory of the East, Middle and West Forks of the Gila River, 2005-2008. New Mexico Department of Game and Fish and U.S. Forest Service Gila National Forest, Santa Fe, NM.
- Partey, F.K., L. Land, B. Frey, E. Premo, and Crossey, L. 2011. Final Report on Geochemistry of Bitter Lake National Wildlife Refuge, Roswell, New Mexico, Socorro, NM: New Mexico Bureau of Geology and Mineral Resources, Open-file Report 526.
- Patrick, G., W. Reid, and S. Helfert. 1977. White Sands National Monument Studies: Preliminary survey of Lake Lucero and Playa Crustaceans and Protozoans (Research Report No. 3). University of Texas, El Paso Laboratory for Environmental Biology, El Paso, TX.
- Paulsen, H.A., Jr., F.N. Ares. 1962. Grazing values and management of black grama and tobosa grasslands and associated shrub ranges of the Southwest. Technical Bulletin 1270. US Department of Agriculture, Forest Service. US Government Printing Office, Washington, DC, USA.
- Paysen, T.E., R.J. Ansley, J.K. Brown, G.J. Gottfried, S.M. Haase, M.G. Harrington, M.G. Narog, S.S. Sackett,

- and R.C. Wilson. 2000. Fire in western shrubland, woodland and grassland ecosystems (General Technical Report No. RMRS-GTR-42-vol. 2), Wildland Fire in Ecosystems. U.S. Department of Agriculture, U.S. Forest Service, Rocky Mountain Research Station, Ogden, UT, USA.
- Patten, D.T., L. Rouse, and J.C. Stromberg. 2008. Isolated spring wetlands in the Great Basin and Mojave deserts, USA: Potential response of vegetation to groundwater withdrawal. *Environmental Management* 41:398–413.
- Pease, A.A., J.J. Davis, M.S. Edwards, and T.F. Turner. 2006. Habitat and resource use by larval and juvenile fishes in an arid-land river (Rio Grande, New Mexico). *Freshwater Biology* 51:475–486.
- Pennington, D.D., and S.L. Collins. 2007. Response of an aridland ecosystem to interannual climate variability and prolonged drought. *Landscape Ecology* 22:897–910. doi:10.1007/s10980-006-9071-5
- Perkin, J.S., K.B. Gido, E. Johnson, and V.M. Tabor. 2010. Consequences of Stream Fragmentation and Climate Change for Rare Great Plains Fishes. Kansas State University, Great Plains Landscape Conservation Cooperative Program, Final Report to U.S. Fish & Wildlife Service, Manhattan, KS.
- Perkins, S.R., McDaniel, K.C., Ulery, A.L. 2006. Vegetation and soil change following creosotebush (*Larrea tridentata*) control in the Chihuahuan Desert. *Journal of Arid Environments* 64:152–173. doi:10.1016/j.jaridenv.2005.04.002
- Peters, D.P.C. 2000. Climatic variation and simulated patterns in seedling establishment of two dominant grasses at a semiarid-arid grassland ecotone. *Journal of Vegetation Science* 11:493–504.
- Peters, D.P.C. 2002. Recruitment potential of two perennial grasses with different growth forms at a semiarid-arid transition zone. *American Journal of Botany* 89:1616–1623.
- Peters, D.P.C. and R.P. Gibbens. 2006. Plant communities in the Jornada Basin: The dynamic landscape. In: Havstad, K.M., L.F. Huenneke, and W.H. Schlesinger (Eds.), *Structure and Function of a Chihuahuan Desert Ecosystem: the Jornada Basin Long-Term Ecological Research Site*. New York: Oxford University Press. <http://jornada.nmsu.edu/files/bibliography/06-061.pdf>. Accessed September 30, 2016.
- Peterson, R.C., and C.S. Boyd. 1998. Ecology and management of sand shinnery communities: a literature review (General Technical Report No. RMRS-GTR-16). U.S. Department of Agriculture, Fort Collins, CO, USA.
- Phillips, A., J. Marshall, and G. Monson. 1978. *The Birds of Arizona*. University of Arizona Press, Tucson, AZ, USA.
- Pieper, R.D. 1977. The southwestern pinyon-juniper ecosystem. Pages 1-5. In: Aldon, E., and T. Loring (Eds.), *Ecology, Uses, and Management of Pinyon-Juniper Woodlands*. GTR-RM-39. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO, USA.
- Pieper, R.D., J.R. Montoya, and V.L. Groce. 1971. Site characteristics on pinyon-juniper and blue grama in south-central New Mexico. Bulletin 573. New Mexico State University, Agricultural Experiment Station, Las Cruces, NM, USA.
- Pidgeon, A.M., N.E. Mathews, R. Benoit, E.V. Nordheim. 2001. Response of Avian Communities to Historic Habitat Change in the Northern Chihuahuan Desert. *Conservation Biology* 15:1772–1788.
- Pilger, T.J., K.B. Gido, and D.L. Propst. 2010. Diet and trophic niche overlap of native and nonnative fishes in the Gila River, USA: implications for native fish conservation. *Ecology of Freshwater Fish* 19:300–321.
- Pilger, T.J., K.B. Gido, D.L. Propst, J.E. Whitney, and T.F. Turner. 2015. Comparative conservation genetics of protected endemic fishes in an arid-land riverscape. *Conservation Genetics* 16:875–888.
- Poff, B., K.A. Koestner, D.G. Neary, and V. Henderson. 2011. Threats to riparian ecosystems in Western

- North America: An analysis of existing literature. *Journal of the American Water Resources Association* 47:1241–1254.
- Poff, N.L., and D.D. Hart. 2002. How dams vary and why it matters for the emerging science of dam removal. *BioScience* 52:659–668.
- Pool, D.B., A. Macias-Duarte, A.O. Panjabi, G. Levandoski, and E. Youngberg. 2012. Chihuahuan Desert Grassland Bird Conservation Plan.
- Pool, D.B., A.O. Panjabi, A. Macias-Duarte, and D.M. Solhjem. 2014. Rapid expansion of croplands in Chihuahua, Mexico threatens declining North American grassland bird species. *Biological Conservation* 170:274–281. doi:10.1016/j.biocon.2013.12.019
- Porter, S.D., R.A. Barker, R.M. Slade, Jr., and G. Longley. 2009. Historical Perspective of Surface Water and Groundwater Resources in the Chihuahuan Desert Network, National Park Service. Texas State University, Edwards Aquifer Research and Data Center (EARDC), Report R1-09, Lubbock, TX.
- Potter, L.D. 1957. Phytosociological study of San Augustin Plains, New Mexico. *Ecological Monographs* 27: 113–136.
- Powell, A.M. and B.L. Turner. 1979. Aspects of the plant biology of the gypsum outcrops of the Chihuahuan Desert. Pages 315–325. In: Wauer, R.H. and D.H. Riskind (Eds.), *Transactions of the Symposium on the Biological Resources of the Chihuahuan Desert Region, United States and Mexico, 17–18 October 1974*. Alpine Texas: Sul Ross State University.
<https://babel.hathitrust.org/cgi/pt?id=umn.31951002827525u;view=1up;seq=1>. Accessed October 8, 2016.
- Price, J., C.H. Galbraith, M. Dixon, J. Stromberg, T. Root, D. MacMykowski, T. Maddock, and K. Baird. 2005. Potential Impacts of Climate Change on Ecological Resources and Biodiversity in the San Pedro Riparian National Conservation Area, Arizona: A Report to U.S. EPA. American Bird Conservancy, The Plains, VA.
- Pringle, C. 2003. What is hydrologic connectivity and why is it ecologically important? *Hydrological Processes* 17:2685–2689.
- PronaturaNoroeste. 2004. *Ecoregional Conservation Assessment of the Chihuahuan Desert*, 2nd Ed. ed. Pronatura Noroeste, Ensenada, Baja California, Mexico.
- Propst, D.L. 1999. *Threatened and Endangered Fishes of New Mexico*, Santa Fe, NM: New Mexico Department of Game and Fish, Technical Report No. 1.
- Propst, D.L. 2016. Personal communication. David L. Propst, Department of Biology and Museum of Southwestern Biology, University of New Mexico, Albuquerque, NM, February 18, 2016.
- Propst, D.L., and K.B. Gido. 2004. Responses of native and nonnative fishes to natural flow regime mimicry in the San Juan River. *Transactions of the American Fisheries Society* 133:922–931.
- Propst, D.L., K.B. Gido, and J.A. Stefferud. 2008. Natural flow regimes, nonnative fishes, and native fish persistence in arid-land river systems. *Ecological Applications* 18:1236–52.
- Propst, D.L., Y.M. Paroz, S.M. Carman, and N.D. Zymonas. 2009. *Systematic Investigations of Warmwater Fish Communities*. New Mexico Department of Game and Fish, Conservation Services Division, Santa Fe, NM.
- Pyne, S.J., P.L. Andrews, and R.D. Laven. 1996. *Introduction to Wildland Fire*. John Wiley & Sons, New York, NY.
- Radford, C.D. 1953. Four new species of “harvest mite” or “chigger” and a new fur-mite *Acarina: Trombiculidae* and *Listrophoridae*. *Parasitology* 43:210. doi:10.1017/S0031182000018564

- Ragotzkie, K.E., and J.A. Bailey. 1991. Desert Mule Deer Use of Grazed and Ungrazed Habitats. *Journal of Range Management* 44:487. doi:10.2307/4002750
- Raison, R.J. 1979. Modification of the soil environment by vegetation fires, with particular reference to nitrogen transformations: a review. *Plant and Soil* 51:73–108.
- Ramirez, J.E., and G.S. Keller. 2010. “Effects of Landscape on Behavior of Black-Tailed Prairie Dogs (*Cynomys ludovicianus*) in Rural and Urban Habitats.” *The Southwestern Naturalist* 55 (2): 167–71.
- Raitt, R.J., and S.L. Pimm. 1976. Dynamics of Bird Communities in the Chihuahuan Desert, New Mexico. *The Condor* 78:427. doi:10.2307/1367091
- Randall, J.A. 1987. Field observations of male competition and mating in Merriam’s and bannertail kangaroo rats. *American Midland Naturalist* 117:211. doi:10.2307/2425723
- Rango, A. 2006. Snow-The real water supply for the Rio Grande basin. *New Mexico Journal of Science* 44:99–118.
- Rango, A., S. Goslee, J. Herrick, M. Chopping, K. Havstad, L. Huenneke, R. Gibbens, R. Beck, R. McNeely. 2002. Remote sensing documentation of historic rangeland remediation treatments in southern New Mexico. *Journal of Arid Environments* 50:549–572.
- Rango, A., S.L. Tartowski, A. Laliberte, J. Wainwright, A. Parsons. 2006. Islands of hydrologically enhanced biotic productivity in natural and managed arid ecosystems. *Journal of Arid Environments* 65:235–252. doi:10.1016/j.jaridenv.2005.09.002
- Rasmussen, D.I. 1941. Biotic communities of Kaibab Plateau, Arizona. *Ecological Monographs* 11:229–275.
- Raun, G.G. 1966. A population of woodrats (*Neotoma micropus*) in southern Texas. Texas Memorial Museum.
- Rautenstrauch, K.R., and P.R. Krausman. 1989. Influence of water availability and rainfall on movements of desert mule deer. *Journal of Mammalogy* 70:197–201.
- Refsnider, J.M., D.A. Warner, and F.J. Janzen. 2013. Does shade cover availability limit nest-site choice in two populations of a turtle with temperature-dependent sex determination? *Journal of Thermal Biology* 38:152–158.
- Reichardt, K.L. 1982. Succession of abandoned fields on the shortgrass prairie, northeastern Colorado. *The Southwestern Naturalist* 299–304.
- Reid, W.H., and G.R. Patrick. 1983. Gemsbok (*Oryx gazella*) in White Sands National Monument. *The Southwestern Naturalist* 28:97. doi:10.2307/3670599
- Relyea, R.A., and S. Demarais. 1994. Activity of Desert Mule Deer during the Breeding Season. *Journal of Mammalogy* 75:940–949. doi:10.2307/1382475
- Relyea, R.A., R.K. Lawrence, and S. Demarais. 2000. Home Range of Desert Mule Deer: Testing the Body-Size and Habitat-Productivity Hypotheses. *The Journal of Wildlife Management* 64:146–153. doi:10.2307/3802984
- Restore New Mexico. 2015. U.S. Department of the Interior Bureau of Land Management. URL <http://www.blm.gov/wo/st/en.html>
- Reynolds, H.G. 1950. Relation of Merriam kangaroo rats to range vegetation in southern Arizona. *Ecology* 31:456–463. doi:10.2307/1931498
- Reynolds, H.G., and J.W. Bohning. 1956. Effects of burning on a desert grass-shrub range in southern Arizona. *Ecology* 37:769–777.

- Reynolds, L.V., D.J. Cooper, and N.T. Hobbs. 2014. Drivers of riparian tree invasion on a desert stream. *River Research and Applications* 30:60–70.
- Rice, P.M., G.R. McPherson, and L.J. Rew. 2008. Fire and nonnative invasive plants in the interior west bioregion (General Technical Report No. RMRS-GTR-42 Volume 6), *Wildland Fire in Ecosystems: Fire and Nonnative Invasive Plants*. U.S. Forest Service, Rocky Mountain Research Station, Fort Collins, CO, USA.
- Richardson, C. 2006. Pronghorn habitat requirements. Pages 5-12. In: Cearley, K.A., and S. Nelle (Eds.), *Pronghorn Symposium*. Presented at the Pronghorn Symposium, Alpine, TX.
- Richter, B.D., and G.A. Thomas. 2007. Restoring environmental flows by modifying dam operations. *Ecology and Society* 12:12.
- Rinne, J.N., and W.L. Minckley. 1991. Native Fishes of Arid Lands: A Dwindling Resource of the Desert Southwest. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-206, Fort Collins, CO.
- Rogalski, M.A., and D.K. Skelly. 2012. Positive effects of nonnative invasive *Phragmites australis* on larval bullfrogs. *PLoS ONE* 7:e44420.
- Rogers, T.J. 1993. Insect and disease associates of the pinon-juniper woodlands. Pages 124-125. In: Aldon, E.F., and D.W. Shaw (Eds.), *Managing Pinyon Juniper Ecosystems for Sustainability and Social Needs*. GTR-RM-236. U.S. Department of Agriculture, Forest Service, Rocky Mountain Experiment and Range Station, Fort Collins, CO, USA.
- Rogers, T.J. 1995. Insect and disease associates of the pinon-juniper woodlands. Pages 107-108. In: Shaw, D.W., E.F. Aldon, and C. LoSapio (Eds.), *Desired Future Conditions for Piñon-Juniper Ecosystems*. Flagstaff, AZ, USA. GTR-RM-258. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Research Station, Fort Collins, CO, USA.
- Rogstad, A., T.M. Bean, A. Olsson, G.M. Casady. 2009. Fire and invasive species management in hot deserts: resources, strategies, tactics and response. *Rangelands* 31:6–13.
- Rollins, D., and J.P. Carroll. 2001. Impacts of Predation on Northern Bobwhite and Scaled Quail. *Wildlife Society Bulletin* 29:39–51.
- Rolls, R. J., I. O. Gowns, T. A. Khan, G. G. Wilson, T. L. Ellison, A. Prior, and C. C. Waring. 2013. Fish recruitment in rivers with modified discharge depends on the interacting effects of flow and thermal regimes. *Freshwater Biology* 58:1804–1819.
- Romme, W., C. Allen, J. Bailey, W. Baker, B. Bestelmeyer, P. Brown, K. Eisenhart, L. Floyd-Hanna, D. Huffman, B. Jacobs, R. Miller, E. Muldavin, T. Swetnam, R. Tausch, P. Weisberg. 2007. Historical and modern disturbance regimes of piñon-juniper vegetation in the western U.S. Colorado Forest Restoration Institute and The Nature Conservancy.
- Romme, W.H., L. Floyd-Hanna, D. D. Hanna. 2003. Ancient pinon-juniper forests of Mesa Verde and the west: a cautionary note for forest restoration programs. Pages 335-350. In: Omi, P.N., Joyce, L.A. (Eds.), *Fire, Fuel Treatments, and Ecological Restoration*. RMRS-P-29. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.
- Ronco Jr., F.P. 1990. *Pinus edulis* Engelm, Pages. 327-337. In Burns R.M. and B.H. Honkala (Eds.), *Silvics of North America, Volume 1. Agricultural Handbook 654*. U.S. Department of Agriculture, Forest Service, Washington, DC. Available online: https://www.na.fs.fed.us/spfo/pubs/silvics_manual/Volume_1/pinus/edulis.htm.
- Rooney, R.C., and S.E. Bayley. 2010. Quantifying a stress gradient: An objective approach to variable selection, standardization and weighting in ecosystem assessment. *Ecological Indicators* 10:1174–1183.

- Rosen, M.R. 1994. The importance of groundwater in playas: A review of playa classifications and the sedimentology and hydrology of playas, in: Geological Society of America Special Papers. Geological Society of America. Pages 1–18.
- Rosen, P.C. 2005. Lowland riparian herpetofaunas: The San Pedro River in Southeastern Arizona. Pages 106–111. In: G.J. Gottfried, B.S. Gebow, L.G. Eskew, and C.B. Edminster (Eds.) *Connecting Mountain Islands and Desert Seas: Biodiversity and Management of the Madrean Archipelago II*, 2004 May 11-15, Tucson, AZ. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Proceedings RMRS-P-36, Fort Collins, CO.
- Rosen, P.C., and D.J. Caldwell. 2004. Aquatic and Riparian Herpetofauna of Las Cienegas National Conservation Area, Empire-Cienega Ranch, Pima County, Arizona. University of Arizona, School of Natural Resources, Final Report to the Bureau of Land Management for Cooperative Agreement AAA000011 Task Order No. 6 (BLM, AZ State Office – CESU-FLPMA, University of Arizona) AZ068-1120-BV-SSSS-25-2B, Tucson, AZ.
- Rosen, P.C., C.R. Schwalbe, D.A. Parizek, P.A. Holm, and C.H. Lowe. 1994. Introduced aquatic vertebrates in the Chiricahua Region: Effects on declining native Ranid frogs. Pages 251–261. In: L.H. DeBano, P.H. Ffolliott, A. Ortega-Rubio, G.J. Gottfried, R.H. Hamre, and C.B. Edminster (Eds.) *Biodiversity and Management of the Madrean Archipelago: The Sky Islands of Southwestern United States and Northwestern Mexico*, Sept. 19-23, 1994, Tucson, Arizona. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-GTR-264, Fort Collins, CO.
- Rosen, P.C., W.R. Radke, and D.J. Caldwell. 2005. Herpetofauna of lowland bottomlands of southeastern Arizona: A comparison of sites. Pages 112–117. In: G.J. Gottfried, B.S. Gebow, L.G. Eskew, and C.B. Edminster (Eds.) *Connecting Mountain Islands and Desert Seas: Biodiversity and Management of the Madrean Archipelago II*, 2004 May 11-15, Tucson, AZ. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Proceedings RMRS-P-36, Fort Collins, CO.
- Rosenzweig, M.L., and J. Winakur. 1969. Population ecology of desert rodent communities: Habitats and environmental complexity. *Ecology* 50:558–72. doi:10.2307/1936246
- Roundy, B.A., and S.H. Biedenbender. 1995. Revegetation in desert grassland. Pages 265-303. In: Mitchell, P., McClaran, M., Van Devender, T.R. (Eds.), *The Desert Grasslands*. University of Arizona Press, Tucson, Arizona.
- Ruhlman, J., L. Gass, and B. Middleton. 2012. Chihuahuan Deserts Ecoregion. Pages 275-284. In: Sleeter, B.M., T.S. Wilson, and W. Acevedo (Eds.) *Status and Trends of Land Change in the Western United States—1973 to 2000*. Reston, VA: U.S. Geological Survey, Professional Paper 1794–A, Available at: <http://pubs.usgs.gov/pp/1794/a/>.
- Ruth, J.M., R.K. Felix, and R.H. Diehl. 2010. Bird Migration Patterns in the Arid Southwest—Final Report. U.S. Geological Survey, Open-File Report 2010–1271, Reston, VA.
- Ruyle, G.G., B.A. Roundy, and J.R. Cox. 1988. Effects of burning on germinability of Lehmann lovegrass. *Journal of Range Management* 41, 404–406.
- Ryan, M., R. Archer, C. Birdsey, L. Dahm, J. Hicke, D. Hollinger, T. Huxman, G. Okin, R. Oren, J. Randerson, and W. Schlesinger. 2008. Land Resources, The effects of climate change on agriculture, land resources, water resources and biodiversity. U.S. Climate Change Science Program and Subcommittee on Global Change Research, Washington D.C.
- Sabo, J.L. 2014. Predicting the river’s blue line for fish conservation. *Proceedings of the National Academy of Sciences* 111:13686–13687.
- Sabo, J.L., K. Bestgen, W. Graf, T. Sinha, and E.E. Wohl. 2012. Dams in the Cadillac Desert: downstream

- effects in a geomorphic context. *Annals of the New York Academy of Sciences* 1249:227–46.
- Sackett, L.C., T.B. Cross, R.T. Jones, W.C. Johnson, K. Ballare, C. Ray, S.K. Collinge, and A.P. Martin. 2012. “Connectivity of Prairie Dog Colonies in an Altered Landscape: Inferences from Analysis of Microsatellite DNA Variation.” *Conservation Genetics* 13 (2): 407–18.
- Saiwana, L., J.L. Holechek, A. Tembo, R. Valdez, and M. Cardenas. 1998. Scaled Quail Use of Different Seral Stages in the Chihuahuan Desert. *The Journal of Wildlife Management* 62:550–556. doi:10.2307/3802328
- Sallach, B.K. 1986. Vegetation changes in New Mexico documented by repeat photography (Thesis). New Mexico State University, Las Cruces, NM, USA.
- Salomonson, M.G. 1978. Adaptations for animal dispersal of one-seed juniper seeds. *Oecologia* 32, 333–339.
- Samuels, M.L., J.L. Betancourt. 1982. Modeling the long-term effects of fuelwood harvests on piñon-juniper woodlands. *Environmental Management* 6, 505–515.
- Sanchez-Rojas, G., and Gallina, S. 2000a. Mule deer (*Odocoileus hemionus*) density in a landscape element of the Chihuahuan Desert, Mexico. *Journal of Arid Environments* 44:357–368.
- Sanchez-Rojas, G., Gallina, S. 2000b. Factors affecting habitat use by mule deer (*Odocoileus hemionus*) in the central part of the Chihuahuan Desert, Mexico: an assessment of univariate and multivariate methods. *Ethology Ecology and Evolution* 12:405–417.
- Sandoval-Solis, S., and D.C. McKinney. 2014. Integrated water management for environmental flows in the Rio Grande. *Journal of Water Resources Planning and Management* 140:355–364.
- Sarmiento, G., and M. Monasterio. 1983. Life forms and phenology. In: Bourliere, F. (Ed.), *Tropical Savannas. Ecosystems of the World* 13. Elsevier, New York, NY, USA, pp. 79–108.
- Sauer, J.R., J.E. Hines, J.E. Fallon, K.L. Pardieck, D.J. Ziolkowski, and W.A. Link, n.d. Breeding Bird Survey, Results and Analysis 1966-2012. Version 02.19.2014 USGS Patuxent Wildlife Research Center, Laurel, MD [WWW Document]. URL <http://www.mbr-pwrc.usgs.gov/bbs/> (accessed 9.18.14).
- Savage, L.T., R.M. Reich, L.M. Hartley, P. Stapp, and M.F. Antolin. 2011. Climate, soils, and connectivity predict plague epizootics in black-tailed prairie dogs (*Cynomys ludovicianus*). *Ecological Applications* 21:2933–2943. doi:10.1890/10-1946.1
- Scanlon, B.R., R.C. Reedy, D.A. Stonestrom, D.E. Prudic, and K.F. Dennehy. 2005. Impact of land use and land cover change on groundwater recharge and quality in the southwestern US. *Global Change Biology* 11:1577–1593.
- Scarbrough, D.L., P.R. Krausman, P.R. 1988. Sexual Segregation by Desert Mule Deer. *The Southwestern Naturalist* 33:157–165. doi:10.2307/3671890
- Schemnitz, S.D. 1994. Scaled Quail (*Callipepla squamata*). *The Birds of North America Online*. doi:10.2173/bna.106
- Schwertner, T.W. 2002. Non-native ungulates in the Trans-Pecos Region of Texas. Pages 53-60. In: Harveson, L.A., P.M. Harveson, and C. Richardson (Eds.), *Proceedings of the Trans-Pecos Wildlife Conference*. Presented at the Trans-Pecos Wildlife Conference, Alpine, TX.
- Schlesinger, W.H., J.F. Reynolds, G.L. Cunningham, L.F. Huenneke, W.M. Jarrell, R.A. Virginia, W.G. Whitford. 1990. Biological feedbacks in global desertification. *Science* 247: 1043–1048. doi:10.1126/science.247.4946.1043
- Schlesinger, W.H., S.L. Tartowski, and S.M. Schmidt. 2006. Nutrient cycling within an arid ecosystem. In:

Havstad, K.M., L.F. Huenneke, and W.H. Schlesinger (Eds.), Structure and Function of a Chihuahuan Desert Ecosystem: the Jornada Basin Long-Term Ecological Research Site. New York: Oxford University Press. <http://jornada.nmsu.edu/files/bibliography/06-057.pdf>. Accessed September 30, 2016

Schmidly, D.J. 1977. Factors governing the distribution of mammals in the Chihuahuan Desert region. Pages 1-658. In: Wauer, R.H., Riskind, D.H. (Eds.), Transactions of the Symposium on the Biological Resources of the Chihuahuan Desert Region United States and Mexico, National Park Service Transactions and Proceedings Series. United States Department of the Interior.

Schmidly, D.J., and R.B. Ditton. 1978. Relating human activities and biological resources in riparian habitats of western Texas. Pages 107–116. In: National Symposium on Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. U.S. Forest Service, General Technical Report WO-12, Flagstaff, AZ.

Schmidt, J.C., and P.R. Wilcock. 2008. Metrics for assessing the downstream effects of dams. *Water Resources Research* 44:W04404.

Schmidt, J.C., B.L. Everitt, and G.A. Richard. 2003. Hydrology and geomorphology of the Rio Grande and implications for river rehabilitation. Pages 25–45. In: G.P. Garrett and N.L. Allan (Eds.), Aquatic Fauna of the Northern Chihuahuan Desert Contributed Papers from a Special Session, Thirty-Third Annual Symposium, Desert Fishes Council, 17 November 2001. Museum of Texas Tech University, Special Publication Number 46, Lubbock, TX.

Schmidt, R.H. 1986. Chihuahuan climate. Pages 40–63. In: Barlow, J.C., A.M. Powell, and B.N. Timmermann, (Eds.), Invited Papers from the Second Symposium on Resources of the Chihuahuan Desert Region, United States and Mexico, 20–21 October 1983. Alpine Texas: Sul Ross State University, Chihuahuan Desert Research Institute.

Schmitt, C.J., J.E. Hinck, V.S. Blazer, N.D. Denslow, G.M. Dethloff, T.M. Bartish, J.J. Coyle, and D.E. Tillitt. 2005. Environmental contaminants and biomarker responses in fish from the Rio Grande and its U.S. tributaries: spatial and temporal trends. *The Science of the Total Environment* 350:161–93.

Schmidt-Nielsen, K. 1964. Desert animals: Physiological problems of heat and water. Oxford at the Clarendon Press, Mishawaka, IN.

Schmutz, E.M. E.L. Smith, P.R. Ogden, M.L. Cox, J.O. Klemmedson, J.J. Norris, and L.C. Fierro. 1992. Desert grasslands. Pages 337-362. In: Coupland, R.T. (Ed.), Ecosystems of the World 8A, Natural Grasslands, Introduction and Western Hemisphere. Elsevier Science, Amsterdam.

Schmutz, E.M. M.K. Sourabie, and D.A. Smith. 1985. The Page Ranch story: its vegetative history and management implications. *Desert Plants* 7:13–21.

Schott, M.R. 1984. Pinyon-juniper ecology with emphasis on secondary succession (Dissertation). New Mexico State University, Las Cruces, NM, USA.

Schott, M.R. and R.D. Pieper. 1986. Succession in pinyon-juniper vegetation in New Mexico. *Rangelands* 8:126–128.

Schott, M.R., and R.D. Pieper. 1987. Succession of pinyon-juniper communities after mechanical disturbance in southcentral New Mexico. *Journal of Range Management* 40:88–94.

Schroder, G.D. 1979. Foraging behavior and home range utilization of the banner-tail kangaroo at (*Dipodomys Spectabilis*). *Ecology* 60:657. doi:10.2307/1936601

Schulman, E. 1956. Dendroclimatic changes in semi-arid America. University of Arizona Press, Tucson, AZ, USA.

Schumm, S.A. 1985. Patterns of alluvial rivers. *Annual Review of Earth and Planetary Sciences* 13:5–27.

- Scott, R.L., E.A. Edwards, W.J. Shuttleworth, T.E. Huxman, C. Watts, and D.C. Goodrich. 2004. Interannual and seasonal variation in fluxes of water and carbon dioxide from a riparian woodland ecosystem. *Agricultural and Forest Meteorology* 122:65–84.
- Scott, R.L., W.L. Cable, T.E. Huxman, P.L. Nagler, M. Hernandez, and D.C. Goodrich. 2008. Multiyear riparian evapotranspiration and groundwater use for a semiarid watershed. *Journal of Arid Environments* 72:1232–1246.
- Scott, R.L., W.J. Shuttleworth, D.C. Goodrich, and T. Maddock III. 2000. The water use of two dominant vegetation communities in a semiarid riparian ecosystem. *Agricultural and Forest Meteorology* 105:241–256.
- Scott, V.E., Boeker, E.L. 1977. Responses of Merriam’s turkey to pinyon-juniper control. *Journal of Range Management* 30, 220–223.
- Scurlock, D. 1993. Pinyon-juniper in southwest history: an overview of historical use of, impact on, and manage through time. Manuscript prepared for National Science Foundation Grant Study, Biology Department, University of New Mexico, Albuquerque, NM, USA.
- Seaber, P.R., F.P. Kapinos, and G.L. Knapp. 1987. Hydrologic Unit Maps. U.S. Geological Survey, Water-Supply Paper 2294:63 p. Denver, Colorado.
- Seager, R., M. Ting, I. Held, Y. Kushnir, J. Lu, G. Vecchi, H.-P. Huang, N. Harnik, A. Leetmaa, N. -C. Lau, C. Li, J. Velez, and N. Naik. 2007. Model Projections of an Imminent Transition to a More Arid Climate in Southwestern North America. *Science* 316:1181–1184. doi:10.1126/science.1139601
- Seiler, R.L., J.P. Skorupa, D.L. Naftz, and B.T. Nolan. 2003. Irrigation-Induced Contamination of Water, Sediment, and Biota in the Western United States-Synthesis of Data from the National Irrigation Water Quality Program. U.S. Geological Survey, Professional Paper 1655., Denver, CO.
- Serrat-Capdevila, A., J.B. Valdés, J.G. Pérez, K. Baird, L.J. Mata, and T. Maddock III. 2007. Modeling climate change impacts – and uncertainty – on the hydrology of a riparian system: The San Pedro Basin (Arizona/Sonora). *Journal of Hydrology* 347:48–66.
- Shafroth, P.B., J.C. Stromberg, and D.T. Patten. 2000. Woody riparian vegetation response to different alluvial water table regimes. *Western North American Naturalist* 60:66–76.
- Shafroth, P.B., A.C. Wilcox, D.A. Lytle, J.T. Hickey, D.C. Andersen, V.B. Beauchamp, A. Hautzinger, L.E. McMullen, and A. Warner. 2010. Ecosystem effects of environmental flows: modelling and experimental floods in a dryland river. *Freshwater Biology* 55:68–85.
- Shafroth, P.B., and V.B. Beauchamp. 2006. Defining Ecosystem Flow Requirements for the Bill Williams River, Arizona. U.S. Geological Survey Open File Report 2006-1314, Reston, VA.
- Shafroth, P.B., J.R. Cleverly, T.L. Dudley, J.P. Taylor, C. van Riper, E.P. Weeks, and J.N. Stuart. 2005. Control of Tamarix in the Western United States: implications for water salvage, wildlife use, and riparian restoration. *Environmental Management* 35:231–46.
- Shafroth, P.B., V.B. Beauchamp, M.K. Briggs, K. Lair, M.L. Scott, and A.A. Sher. 2008. Planning riparian restoration in the context of Tamarix control in western North America. *Restoration Ecology* 16:97–112.
- Sharp, J.M. Jr., R. Boghici, M.M. Uliana. 2003. Groundwater systems feeding the springs of west Texas, Pages 1-11. In: Garrett, G.P., and N.L. Allan (Eds.), *Aquatic Fauna of the Northern Chihuahuan Desert Contributed Papers from a Special Session, Thirty-Third Annual Symposium, Desert Fishes Council, 17 November 2001*. Museum of Texas Tech University, Special Publication Number 46, Lubbock, Texas.
- Shaw, H.G. 2006. Wood plenty, grass good, water none: Vegetation changes in Arizona’s upper Verde River

watershed from 1850 to 1997. RMRS-GTR-177. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO, USA.

- Shaw, J.D., B.E. Steed, and L.T. DeBlander. 2005. Forest Inventory and Analysis (FIA) Annual inventory answers the question: what is happening to pinyon-juniper woodlands? *Journal of Forestry* 103:280–285.
- Sheaffer, J.D., and E.R. Reiter. 1985. Influence of Pacific Sea surface temperatures on seasonal precipitation over the western plateau of the United States. *Archives for Meteorology, Geophysics and Bioclimatology* 34:111–130.
- Sheldon, F., and M.C. Thoms. 2006. In-channel geomorphic complexity: The key to the dynamics of organic matter in large dryland rivers? *Geomorphology* 77:270–285.
- Sheley, R.L., J.J. James, J.M. Rinella, D. Blumenthal, J.M. DiTomaso. 2011. Invasive plant management on anticipated conservation benefits: a scientific assessment, *Conservation Benefits of Rangelands Practices: Assessment, Recommendations and Knowledge Gaps*. U.S. Department of Agriculture, Natural Resources Conservation Service, Lawrence, KS.
- Sheng, Z. 2013. Impacts of groundwater pumping and climate variability on groundwater availability in the Rio Grande Basin. *Ecosphere* 4:Article 5.
- Short, H.L. 1977. Food habits of mule deer in a semidesert grass-shrub habitat. *Journal of Range Management* 30:206. doi:10.2307/3897471
- Short, H.L., D.E. Medin, and A.E. Anderson. 1965. Ruminoreticular characteristics of mule deer. *Journal of Mammalogy* 46:196. doi:10.2307/1377838
- Short, H.L., W. Evans, E.L. Boeker. 1977. The use of natural and modified pinyon-juniper woodlands by deer and elk. *Journal of Wildlife Management* 41: 543–559.
- Shump, K.A. 1999. Yellow-nosed cotton rat, *Sigmodon fulviventer*. Pages 592-593. In: Wilson, D.E., and S. Ruff (Eds.), *The Smithsonian Book of North American Mammals*. Smithsonian Institution Press, Washington, DC.
- Sickerman, S.L., and J.K. Wangberg. 1983. Behavioral responses of the cactus bug, *Chelinidea vittiger* Uhler, to fire damaged host plants. *Southwestern Entomologist* 8: 263–267.
- Siegel, M.D., S.J. Lambert and K.L. Robinson. 1991. Hydrogeochemical Studies of the Rustler Formation and Related Rocks in the Waste Isolation Pilot Plant Area, Southeastern New Mexico, Albuquerque, NM: Sandia National Laboratories, Sandia Report SAND88-0196.
- Sigstedt, S.C., F.M. Phillips, and A.B.O. Ritchie. 2016. Groundwater flow in an “underfit” carbonate aquifer in a semiarid climate: application of environmental tracers to the Salt Basin, New Mexico (USA). *Hydrogeology Journal*:24(4):841–863.
- Simpson, D.C., L.A. Harveson, C.E. Brewer, R.E. Walser, A.R. Sides. 2007. Influence of precipitation on pronghorn demography in Texas. *Journal of Wildlife Management* 71:906–910.
- Sims, P.L., and J.S. Singh. 1978. The structure and function of ten western North American Grasslands. II. Intra-seasonal dynamics in primary producer compartments. *Journal of Ecology* 66(2): 547-572.
- Skagen, S.K., C.P. Melcher, W.H. Howe, and F.L. Knopf. 1998. Comparative use of riparian corridors and oases by migrating birds in southeast Arizona. *Conservation Biology* 12:896–909.
- Skagen, S.K., J.F. Kelly, C. van Riper III, R.L. Hutto, D.M. Finch, D.J. Krueper, and C.P. Melcher. 2005. Geography of spring landbird migration through riparian habitats in southwestern North America. *The Condor* 107:212–227.

- Skyvarla, J.L., J.D. Nichols, J.E. Hines, and P. Waser. 2004. Modeling interpopulation dispersal by banner-tailed kangaroo rats. *Ecology* 85:2737–2746.
- Smakhtin, V.U. 2001. Low flow hydrology: a review. *Journal of Hydrology* 240:147–186.
- Small, M.F., T.H. Bonner, and J.T. Baccus. 2009. Hydrologic alteration of the lower Rio Grande terminus: A quantitative assessment. *River Research and Applications* 25:241–252.
- Smith, C., R. Valdez, J.L. Holechek, P.J. Zwank, and M Cardenas. 1998a. Diets of native and non-native ungulates in southcentral New Mexico. *The Southwestern Naturalist* 163–169.
- Smith, C., R. Valdez, J.L. Holechek, P.J. Zwank, and M. Cardenas. 1998b. Diets of native and non-native ungulates in southcentral New Mexico. *The Southwestern Naturalist* 163–169.
- Smith, D.M., and D.M. Finch. 2014. Use of native and nonnative nest plants by riparian-nesting birds along two streams in New Mexico. *River Research and Applications* 30:1134–1145.
- Smith, D.M., D.M. Finch, and D.L. Hawksworth. 2009. Black-chinned hummingbird nest-site selection and nest survival in response to fuel reduction in a southwestern riparian forest. *The Condor* 111:641–652.
- Smith, D. M., J. F. Kelly, and D. M. Finch. 2006. Influences of disturbance and vegetation on abundance of native and exotic detritivores in a southwestern riparian forest. *Environmental Entomology* 35:1525–1531.
- Smith, F.A., and J.L. Betancourt. 2006. Predicting woodrat (*Neotoma*) responses to anthropogenic warming from studies of the palaeomidden record. *Journal of Biogeography* 33:2061–2076. doi:10.1111/j.1365-2699.2006.01631.x
- Smith, F.A., H. Browning, and U.L. Shepherd. 1998. The influence of climate change on the body mass of woodrats *Neotoma* in an arid region of New Mexico, USA. *Ecography* 21:140–148. doi:10.1111/j.1600-0587.1998.tb00668.x
- Smith, R.H., and A. Lecount. 1979. Some factors affecting survival of desert mule deer fawns. *Journal of Wildlife Management* 43:657–665.
- Sogge, M.K., S.J. Sferra, and E.H. Paxton. 2008. Tamarix as habitat for birds: implications for riparian restoration in the southwestern United States. *Restoration Ecology* 16:146–154.
- Southeast New Mexico Economic Development District/Council of Governments (SENMEDD/COG). 2010. Comprehensive Economic Development Strategy, 2011-2015. <http://snmedd.com/wp-content/uploads/2013/12/CEDs-Revised-Narrative-10-20-12.pdf>. Accessed September 25, 2016.
- Sparks, R.E. 1995. Need for ecosystem management of large rivers and their floodplains. *BioScience* 45:168–182.
- Sperry, J.S., and M.T. Tyree. 1988. Mechanism of water stress-induced xylem embolism. *Plant Physiology* 88: 581–587.
- Springfield, H.W. 1976. Characteristics and management of southwestern pinyon-juniper ranges: the status of our knowledge. RP-RM-160. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO, USA.
- Springer, A.E. and L.E. Stevens. 2008. Spheres of discharge of springs. *Hydrogeology Journal* 17:83–93. doi:10.1007/s10040-008-0341-y
- Stacey, C.A., and D.M. Post. 2009. Effects of disturbance by humans on small mammals in a Chihuahuan desert ecosystem. *The Southwestern Naturalist* 54:272–278. doi:10.1894/PS-29.1
- Stafford, K.W. 2013. Evaporite Karst and Hydrogeology of the Castile Formation: Culberson County, Texas and Eddy County, New Mexico. In National Cave and Karst Research Institute 13th Sinkhole

Conference, Symposium 2. Carlsbad, NM: National Cave and Karst Research Institute, pp. 123–132. Available at: <http://scholarworks.sfasu.edu/geology/3> =.

- Stafford, K.W., A.B. Klimchouk, L. Land, and M. O. Gary. 2009. The Pecos River Hypogene Speleogenetic Province: a Basin-Scale Karst Paradigm for Eastern New Mexico and West Texas, USA. Pages 121-135. In: K. Stafford, L. Land, and G. Veni, eds. *Advances in Hypogene Karst Studies*. Carlsbad, NM: National Cave and Karst Research Institute, Symposium, 1.
- Stafford, K.W., D. Ulmer-Scholle, and L. Rosales-Lagarde. 2008. Hypogene calcitization: Evaporite diagenesis in the western Delaware Basin. *Carbonates and Evaporites*, 23(2):89–103. Available at: <http://www.springerlink.com/index/10.1007/BF03176155>.
- Stafford, K.W., L. Land, and A. Klimchouk. 2008. Hypogenic speleogenesis within Seven Rivers Evaporites: Coffee Cave, Eddy County, New Mexico. *Journal of Cave and Karst Studies*, 70(1):47–61.
- Stafford, K.W., R. Nance, L. Rosales-Lagarde, and P.J. Boston. 2008. Epigene and hypogene gypsum karst manifestations of the Castile formation: Eddy County, New Mexico and Culberson County, Texas, USA. *International Journal of Speleology*, 37(2):83–98.
- Stefferdud, J.A., K.B. Gido, and D.L. Propst. 2011. Spatially variable response of native fish assemblages to discharge, predators and habitat characteristics in an arid-land river. *Freshwater Biology* 56:1403–1416.
- Steidl, R.J., A.R. Litt, W.J. Matter. 2013. Effects of plant invasions on wildlife in desert grasslands: Plant Invasions and Wildlife in Desert Grasslands. *Wildlife Society Bulletin* 37:527–536. doi:10.1002/wsb.308
- Stoddard, J.L., D. Peck, A.R. Olsen, D.P. Larsen, J. Van Sickle, C.P. Hawkins, R.M. Hughes, T.R. Whittier, G. Lomnický, A.T. Herlihy, P.R. Kaufmann, S.A. Peterson, P.L. Ringold, S.G. Paulsen, and R. Blair. 2005. Environmental Monitoring and Assessment Program (EMAP), Western Streams and Rivers Statistical Summary. United States Environmental Protection Agency, Office of Research and Development, EPA 620/R-05/006, Washington, D.C.
- Stonestrom, D.A., J. Constantz, T.P.A. Ferré, and S.A. Leake (Eds.) 2007. *Ground-Water Recharge in the Arid and Semiarid Southwestern United States*. U.S. Geological Survey, Professional Paper 1703, Reston, VA.
- Stromberg, J.C. 1998. Dynamics of Fremont cottonwood (*Populus fremontii*) and saltcedar (*Tamarix chinensis*) populations along the San Pedro River, Arizona. *Journal of Arid Environments* 40:133–155.
- Stromberg, J.C. 2001. Restoration of riparian vegetation in the south-western United States: importance of flow regimes and fluvial dynamism. *Journal of Arid Environments* 49:17–34.
- Stromberg, J.C., D.C. Andersen, and M.L. Scott. 2012. Riparian floodplain wetlands of the arid and semiarid Southwest. Pages 343–356. In: Batzer, D.P. and A.H. Baldwin (Eds.), *Wetland Habitats of North America: Ecology and Conservation Concerns*. University of California Press, Berkeley, CA.
- Stromberg, J.C., K.J. Bagstad, J.M. Leenhouts, S.J. Lite, and E. Makings. 2005. Effects of stream flow intermittency on riparian vegetation of a semiarid region river (San Pedro River, Arizona). *River Research and Applications* 21:925–938.
- Stromberg, J.C., T.J. Rychener, and M.D. Dixon. 2009a. Return of fire to a free-flowing desert river: effects on vegetation. *Restoration Ecology* 17:327–338.
- Stromberg, J.C., M.K. Chew, P.L. Nagler, and E.P. Glenn. 2009b. Changing perceptions of change: The role of scientists in tamarix and river management. *Restoration Ecology* 17:177–186.
- Stromberg, J.C., S.J. Lite, R. Marler, C. Paradzick, P.B. Shafroth, D. Shorrocks, J.M. White, and M.S. White. 2007. Altered stream-flow regimes and invasive plant species: the Tamarix case. *Global Ecology and Biogeography* 16:381–393.

- Stromberg, J.C., S.J. Lite, T.J. Rychener, L.R. Levick, M.D. Dixon, and J.M. Watts. 2006. Status of the riparian ecosystem in the Upper San Pedro River, Arizona: Application of an assessment model. *Environmental monitoring and assessment* 115:145–173.
- Stout, C.A., and D.W. Duszynski. 1983. *Coccidia* from kangaroo rats (*Dipodomys* spp.) in the western United States, Baja California, and northern Mexico with descriptions of *Eimeria merriami* sp. n. and *Isospora* sp. *The Journal of Parasitology* 69, 209–214.
- Sublette, J.E., and M.S. Sublette. 1967. The limnology of playa lakes on the Llano Estacado, New Mexico and Texas. *The Southwestern Naturalist* 12:369.
- Sullins, M.R. 2002. Factors affecting pronghorn antelope populations in Trans-Pecos, Texas. Pages 29-36. In: Harveson, L.A., P.M. Harveson, and C. Richardson (Eds.), *Proceedings of the Trans-Pecos Wildlife Conference*. Presented at the Trans-Pecos Wildlife Conference, Alpine, TX.
- Sullivan Graham, E.J., A.C. Jakle, and F.D. Martin. 2015. Reuse of oil and gas produced water in southeastern New Mexico: resource assessment, treatment processes, and policy. *Water International*, 40(5–6):809–823. Available at: <http://www.tandfonline.com/doi/full/10.1080/02508060.2015.1096126>.
- Sumner, M.L. 2006. Managing pronghorn habitat in west Texas. Presented at the Pronghorn Symposium 2006, Alpine, TX, pp. 47–52.
- Sumrall, L.B., B.A. Roundy, J.R. Cox, V.K. Winkel. 1991. Influence of canopy removal by burning or clipping on emergence of *Eragrostis lehmanniana* seedlings. *International Journal of Wildland Fire* 1:35–40.
- Swetnam, T.R., and J.L. Betancourt. 1990. Fire-Southern Oscillation relations in the southwestern United States. *Science* 24: 1017–1020.
- Swetnam, T.W., and J.L. Betancourt. 1998. Mesoscale disturbance and ecological response to decadal climatic variability in the American southwest. *Journal of Climate* 11:3128–3147.
- Swhart, R.K., and N.A. Slade. 1984. Road crossing in *Sigmodon hispidus* and *Microtus ochrogaster*. *Journal of Mammalogy* 65:357–360. doi:10.2307/1381184
- Szynkiewicz, A., B. Talon Newton, S.S. Timmons, and D.M. Borrok. 2012. The sources and budget for dissolved sulfate in a fractured carbonate aquifer, southern Sacramento Mountains, New Mexico, USA. *Applied Geochemistry* 27:1451–1462. Available at: <http://dx.doi.org/10.1016/j.apgeochem.2012.04.011>.
- Szynkiewicz, A., D.M. Borrok, G. Skrzypek, and M.S. Rearick. 2015a. Isotopic studies of the Upper and Middle Rio Grande. Part 1 - Importance of sulfide weathering in the riverine sulfate budget. *Chemical Geology* 411:323–335.
- Szynkiewicz, A., D.M. Borrok, G.K. Ganjegunte, G. Skrzypek, L. Ma, M.S. Rearick, and G.B. Perkins. 2015b. Isotopic studies of the Upper and Middle Rio Grande. Part 2 - Salt loads and human impacts in south New Mexico and west Texas. *Chemical Geology* 411:336–350. Available at: <http://dx.doi.org/10.1016/j.chemgeo.2015.05.023>.
- Szynkiewicz, A., R.C. Ewing, C.H. Moore, M. Glamoclija, D. Bustos, and L.M. Pratt. 2010. Origin of terrestrial gypsum dunes-Implications for Martian gypsum-rich dunes of Olympia Undae. *Geomorphology*, 121(1–2):69–83.
- Tamarisk Coalition. 2015. 2007-2015 Distribution of Tamarisk Beetle (*Diorhabda* spp.). Tamarisk Coalition, Grand Junction, CO. Online: [http://www.tamariskcoalition.org/sites/default/files/files/2015_Beetle_Distribution_Map_Final_12_9\(1\).pdf](http://www.tamariskcoalition.org/sites/default/files/files/2015_Beetle_Distribution_Map_Final_12_9(1).pdf).

- Tank, J.L., E.J. Rosi-Marshall, N.A. Griffiths, S.A. Entekin, and M.L. Stephen. 2010. A review of allochthonous organic matter dynamics and metabolism in streams. *Journal of the North American Benthological Society* 29:118–146.
- Tausch, R.J., and P.T. Tueller. 1977. Plant succession following chaining of pinyon-juniper woodlands in eastern Nevada. *Journal of Range Management* 30:44–49.
- Tausch, R.J., N.E. West, A.A. Nabi. 1981. Tree age and dominance patterns in Great Basin pinyon-juniper woodlands. *Journal of Range Management* 34:259–264.
- Taylor, D.W. 1987. Fresh-Water Molluscs from New Mexico and Vicinity. New Mexico Bureau of Mines and Mineral Resources, Bulletin 116, Socorro, New Mexico.
- Taylor, R.B. 2006. The effects of off-road vehicles on ecosystems. Texas Parks and Wildlife Department, Uvalde, TX.
- Texas Demographic Center. 2015. Total Population by County, 2014. http://demographics.texas.gov/Resources/TPEPP/Estimates/2014/2014_txpopest_county.csv. Accessed September 27, 2016.
- Texas General Land Office. 2015. George P. Bush's Energy Map of Texas. <http://commissionerbushmaps.com/> Accessed September 26, 2016.
- Texas Land Trends. 2015. Texas A&M Institute of Renewable Natural Resources, College Station, Texas. <http://txlandtrends.org>. Accessed September 26 2016.
- Texas Parks and Wildlife Department (TPWD). 2012. Texas Conservation Action Plan 2012 – 2016. Texas Parks and Wildlife Department, Austin, TX.
- Texas Parks and Wildlife Department (TPWD). 2010. Wildlife management activities and practices: comprehensive wildlife management planning guidelines for the Trans-Pecos ecological region. Texas Parks and Wildlife Department, Austin, TX.
- Texas Water Development Board (TWDB). 2016. 2016 Far West Texas Water Plan. Texas Water Development Board, Austin, TX. <https://www.twdb.texas.gov/waterplanning/rwp/plans/2016/index.asp>.
- Texas Water Development Board (TWDB). 2016. Texas Water Development Board 2016 Adopted Regional Water Plans. <https://www.twdb.texas.gov/waterplanning/rwp/plans/2016/index.asp>. Accessed January 4, 2017.
- Thibault, K.M., S.K.M. Ernest, E.P. White, J.H. Brown, and J.R. Goheen. 2010. Long-term insights into the influence of precipitation on community dynamics in desert rodents. *Journal of Mammalogy* 91:787–797. doi:10.1644/09-MAMM-S-142.1
- The Nature Conservancy (TNC). 2016. Places we protect, Chihuahuan Desert [WWW Document]. The Nature Conservancy. URL <http://www.nature.org/ourinitiatives/regions/northamerica/mexico/placesweprotect/chihuahuan-desert.xml> (accessed 1.9.16).
- Theobald, D.M. 2013. A general model to quantify ecological integrity for landscape assessments and US application. *Landscape Ecology*, 28(10):1859–1874.
- Theobald, D.M., D.M. Merritt, and J.B. Norman III. 2010. Assessment of threats to riparian ecosystems in the western U.S. Western Environmental Threats Assessment Center, Prineville, OR.
- Theobald, D.M., W.R. Travis, M.A. Drummond, and E.S. Gordon. 2013. The Changing Southwest. Pages 37–55. In: Garfin, G., A. Jardine, R. Merideth, M. Black, and S. LeRoy (Eds.), *Assessment of Climate Change in the Southwest United States: A Report Prepared for the National Climate Assessment*. A report by

the Southwest Climate Alliance. Washington, DC: Island Press.

- Thomas, P.A. 1991. Response of succulents to fire: a review. *International Journal of Wildland Fire* 1:11–22.
- Thoms, M.C. 2006. Variability in riverine ecosystems. *River Research and Applications* 22:115–121.
- Throop, H.L., L.G. Reichmann, O.E. Sala, and S.R. Archer. 2012. Response of dominant grass and shrub species to water manipulation: an ecophysiological basis for shrub invasion in a Chihuahuan Desert Grassland. *Oecologia* 169:373–383. doi:10.1007/s00442-011-2217-4
- Tillman, F.D., J.T. Cordova, S.A. Leake, B.E. Thomas, and J.B. Callegary. 2011. *Water Availability and Use Pilot: Methods Development for a Regional Assessment of Groundwater Availability, Southwest Alluvial Basins, Arizona*. U.S. Geological Survey, Scientific Investigations Report 2011–5071, Reston, VA.
- Tluczek, M. 2012. *Diet, Nutrients, and Free Water Requirements of Pronghorn Antelope on Perry Mesa, Arizona (Masters Thesis)*. Arizona State University, Tempe, AZ.
- Tockner, K., and J.A. Stanford. 2002. Riverine flood plains: present state and future trends. *Environmental Conservation* 29:308–330.
- Tockner, K., F. Malard, and J.V. Ward. 2000. An extension of the flood pulse concept. *Hydrological Processes* 14:2861–2883.
- Tockner, K., M. Pusch, D. Borchardt, and M.S. Lorang. 2010. Multiple stressors in coupled river-floodplain ecosystems. *Freshwater Biology* 55:135–151.
- Tollefson, T.N., L.A. Shipley, W.L. Myers, and N. Dasgupta, N. 2011. Forage Quality's Influence on Mule Deer Fawns. *Journal of Wildlife Management* 75:919–928.
- Travis, W.R. 2007. *New Geographies of the American West: Land Use and the Changing Patterns of Place*. Washington, DC: Island Press.
- Tress, J.A., Jr., Klopatek, J.M. 1987. Successional changes in community structure of pinyon-juniper woodlands on north-central Arizona. Pages 80-85. In: *Proceedings of the Pinyon-Juniper Conference*. GTR-INT-215. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, UT, USA.
- Trush, W.J., S.M. McBain, and L.B. Leopold. 2000. Attributes of an alluvial river and their relation to water policy and management. *Proceedings of the National Academy of Sciences of the United States of America* 97:11858–11863.
- Tull, J.C., and P.R. Krausman. 2007. Habitat use of a fragmented landscape by females in a small population of desert mule deer. *The Southwestern Naturalist* 52:104–109. doi:10.1894/0038-4909(2007)52[104:HUAFL]2.0.CO;2
- Turner, D.S., and M.D. List. 2007. Habitat mapping and conservation analysis to identify critical streams for Arizona's native fish. *Aquatic Conservation: Marine and Freshwater Ecosystems* 17:737–748.
- Turner, T.F., and M.S. Edwards. 2012. Aquatic foodweb structure of the Rio Grande assessed with stable isotopes. *Freshwater Science* 31:825–834.
- Turner, T.F., T.J. Krabbenhoft, and A.S. Burdett. 2010. Reproductive Phenology and Fish Community Structure in an Arid-Land River System. Pages 427-446. In: Gido, K.B. and D.A. Jackson (Eds.), *Community Ecology of Stream Fishes: Concepts, Approaches, and Techniques*. Bethesda, MD: American Fisheries Society Symposium 73.
- Tyus, H.M., and W.L. Minckley. 1988. Migrating mormon crickets, *Anabrus simplex* (Orthoptera: Tettigoniidae), as food for stream fishes. *Great Basin Naturalist* 48:25–30.
- U.S. Air Force (USAF). 1999. *Integrated Natural Resources Management Plan*. Holloman AFB, NM.

- U.S. Army Corps of Engineers (USACE). 2011. Final Detailed Project Report With Integrated Environmental Assessment: Section 1135 Las Cruces Dam Environmental Restoration Project. USACE Albuquerque District, South Pacific Division, Albuquerque, NM.
- U.S. Bureau of Reclamation (USBR). 2011. Literature Synthesis on Climate Change Implications for Water and Environmental Resources, 2nd Edition. U.S. Department of the Interior, Bureau of Reclamation, Research and Development Office, Denver, CO.
- U.S. Bureau of Reclamation. (USBR) 2011. Rio Grande Project. http://www.usbr.gov/projects/Project.jsp?proj_Name=Rio%20Grande%20Project&pageType=ProjectPage. Accessed September 28, 2016.
- U.S. Bureau of Reclamation (USBR). 2012. Draft Environmental Assessment and Biological Assessment, Pecos River Restoration at the Overflow Wetlands, Area of Critical Environmental Concern, Chaves County, New Mexico. U.S. Department of the Interior, Bureau of Reclamation, Albuquerque Area Office, Albuquerque, New Mexico.
- U.S. Bureau of Reclamation (USBR). 2013. West-Wide Climate Risk Assessment: Upper Rio Grande Impact Assessment. U.S. Department of the Interior, Bureau of Reclamation, Upper Colorado Region, Albuquerque Area Office, Albuquerque, NM.
- U.S. Department of Agriculture, Forest Service, Missoula Fire Sciences Laboratory. 2012. Information from LANDFIRE on fire regimes of southwestern desert grasslands. In: Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Missoula Fire Sciences Laboratory (Producer). Available: https://www.fs.fed.us/database/feis/fire_regimes/SW_desert_grass/all.html [2017, July 9] USDA, NRCS, 2006. Ecological site descriptions. US Department of Agriculture, Natural Resources Conservation Service [WWW Document]. URL <https://esis.sc.egov.usda.gov> (accessed 1.15.14).
- U.S. Department of Agriculture, Natural Resources Conservation Service (USDA NRCS). 2014. The PLANTS Database. National Plant Data Team, Greensboro, NC, USA [WWW Document]. URL <http://plants.usda.gov> (accessed 7.30.14).
- U.S. Department of Agriculture, Natural Resources Conservation Service (USDA NRCS). 2016. USDA-NRCS Ecological Site Descriptions U.S. Department of Agriculture, Natural Resources Conservation Service, Las Cruces, NM. Online: <https://esis.sc.egov.usda.gov>.
- U.S. Department of Agriculture, Natural Resources Conservation Service (USDA NRCS). 2016. The PLANTS Database. National Plant Data Team, Greensboro, NC, USA [WWW Document]. URL <http://plants.usda.gov> (accessed 2.5.16).
- U.S. Department of Agriculture, Natural Resources Conservation Service (USDA-NRCS). 2006. Land Resource Regions and Major Land Resource Areas of the United States, the Caribbean, and the Pacific Basin. U.S. Department of Agriculture, Natural Resources Conservation Service, Handbook 296, Washington, D.C.
- U.S. Energy Information Administration (USEIA). 2015. New Mexico State Profile and Energy Estimates. U.S. Energy Information Administration, State Profiles and Energy Estimates. Available at: <http://www.eia.gov/state/?sid=NM> [Accessed July 14, 2016].
- U.S. Energy Information Administration (USEIA). July 19, 2014. Six formations are responsible for surge in Permian Basin crude oil production. <https://www.eia.gov/todayinenergy/detail.php?id=17031>. Accessed September 29, 2016.
- U.S. Energy Information Administration (USEIA). September 2016. Permian Region: Drilling Productivity Report. <http://www.eia.gov/petroleum/drilling/pdf/permian.pdf>. Accessed September 29, 2016.
- U.S. Environmental Protection Agency (USEPA). 2005. National Management Measures to Control Nonpoint

Source Pollution from Urban Areas. United States Environmental Protection Agency, Office of Water, EPA-841-B-05-004, Washington, D.C.

- U.S. Environmental Protection Agency (USEPA). 2013. Level III Ecoregions of the Continental United States. U.S. EPA National Health and Environmental Effects Research Laboratory, Corvallis, Oregon. Online: <https://www.epa.gov/eco-research/level-iii-and-iv-ecoregions-continental-united-states>.
- U.S. Fish and Wildlife Service (USFWS). 2009. Chiricahua leopard frog (*Lithobates [Rana] chiricahuensis*): Considerations for making effects determinations and recommendations for reducing and avoiding adverse effects. Original 2. U.S. Fish and Wildlife Service, New Mexico Ecological Services Field Office, Southwest Endangered Species Act Team, Albuquerque, NM.
- U.S. Fish and Wildlife Service (USFWS). 2010. Pecos Bluntnose Shiner (*Notropis simus pecosensis*) 5-Year Review: Summary and Evaluation. U.S. Fish and Wildlife Service, New Mexico Ecological Services Field Office, Albuquerque, New Mexico. URL: http://ecos.fws.gov/docs/five_year_review/doc3233.pdf.
- U.S. Fish and Wildlife Service (USFWS). 2011. Designation of Critical Habitat for Roswell Springsnail, Koster's Springsnail, Noel's Amphipod, and Pecos Assiminea. Federal Register 76:33036–33064.
- U.S. Fish and Wildlife Service (USFWS). 2012. Determination of Endangered Status for the Chupadera Springsnail and Designation of Critical Habitat. Federal Register 77:41088–41106.
- U.S. Fish and Wildlife Service (USFWS). 2013. Designation of Critical Habitat for Six West Texas Aquatic Invertebrates. Federal Register 78:40970–40996.
- U.S. Forest Service (USFS) 2014. Field guide for managing Lehmann and weeping lovegrasses in the Southwest, TP-R3-16-21. US Department of Agriculture, Forest Service, Southwestern Region, Albuquerque, NM, USA.
- Urbanczyk, K., D. Rohr, and J.C. White. 2001. Geologic History of West Texas. Pages 17-25. In: Mace, R.E., W.F. Mullican III, and E.S. Angle (Eds.), *Aquifers of West Texas*, Austin, TX: Texas Water Development Board Report 356. Available at: https://www.twdb.texas.gov/publications/reports/numbered_reports/doc/R356/356_AquifersofWestTexas.pdf.
- Urness, P.J. 1981. Desert chaparral habitats Part 1. Food habits and nutrition. Pages 347-365. In: Wallmo, O.C. (Ed.), *Mule and Black-Tailed Deer of North America*. University of Nebraska Press, Lincoln, NE.
- Valone, T.J., J.H. Brown, C.L. Jacobi. 1995. Catastrophic decline of a desert rodent, *Dipodomys spectabilis*: Insights from a Long-Term Study. *Journal of Mammalogy* 76:428–436. doi:10.2307/1382353
- Valone, T.J., S.E. Nordell, S.K.M. Ernest. 2002. Effects of fire and grazing on an arid grassland ecosystem. *The Southwestern Naturalist* 47:557. doi:10.2307/3672659
- Van Auken, O.W. 2000. Shrub invasions of North American semiarid grasslands. *Annual Review of Ecology and Systematics* 31:197-215.
- Vander Wall, S.B., and R.P. Balda. 1977. Coadaptations of the Clark's nutcracker and the pinon pine for efficient seed harvest and dispersal. *Ecological Monographs* 47:89–111.
- Van Devender, T.R. 1990. Late Quaternary vegetation and climate of the Chihuahuan Desert, United States and Mexico. Pages 104-133. In: Betancourt, J.L., T.R. Van Devender, and P.S. Martin, P.S. (Eds.), *Packrat Middens: The Last 40,000 Years of Biotic Change*. University of Arizona Press, Tucson, AZ, USA.
- Van Devender, T.R., A.L. Reina-Guerrero, and J.J. Sanchez-Escalante. 2013. Flora of Chihuahuan Desertscrub on limestone in northeastern Sonora, Mexico. Pages 229-235. In: Gottfried, G.J., P.F. Ffolliott, B.S. Gebow, L.G. Eskew, and L.C. Collins (Eds.), *Merging Science and Management in a Rapidly Changing World: Biodiversity and Management of the Madrean Archipelago III*. Proceedings. RMRS-P-67. US

Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO, USA.

- Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* 37:130–137.
- Varyu, D., and L. Fotherby. 2015. Vegetation calibration in a sediment transport model of the middle Rio Grande, New Mexico. Pages 693–704. In: *Proceedings of the 3rd Joint Federal Interagency Conference on Sedimentation and Hydrologic Modeling*, April 19-23, 2015, Reno, Nevada, USA. Advisory Committee on Water Information, Washington, D.C.
- Vickery, P.D. 1996. Grasshopper Sparrow (*Ammodramus savannarum*). *The Birds of North America Online*. doi:10.2173/bna.239
- Vorhies, C.T. 1922. Life history of the kangaroo rat: *Dipodomys spectabilis spectabilis* Merriam. *US Department of Agriculture* 1091:1–40.
- Vorhies, C.T., and W.P. Taylor. 1993. Animal feedbacks in desertification: An overview. *Revista Chilena de Historia Natural* 66:243–251.
- Wagner, J.A., and R.T. Gage. 2012. Wildlife and vegetation response to Spike 20P in the Trans-Pecos region of Texas. In: *Proceedings of the Trans-Pecos Wildlife Conference-2012*. Presented at the August 2-3, 2012 Trans-Pecos Wildlife Conference, Alpine, TX.
- Wainwright, J. 2006. Climate and climatological variations in the Jornada Basin. In: Kris M. Havstad, L.F. Huenneke, and W.H. Schlesinger (Eds.), *Structure and Function of a Chihuahuan Desert Ecosystem: the Jornada Basin Long-Term Ecological Research Site*. New York: Oxford University Press. <http://jornada.nmsu.edu/files/bibliography/06-054.pdf>. Accessed September 30, 2016.
- Walker, H.A. 2006. Southwestern avian community organization in exotic Tamarix: Current patterns and future needs. Pages 274–286. In: Aguirre-Bravo, C., P.J. Pellicane, D.P. Burns, and S. Draggan (Eds.) *Monitoring Science and Technology Symposium: Unifying Knowledge for Sustainability in the Western Hemisphere*, September 20-24, 2004, Denver, CO. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, *Proceedings RMRS-P-42CD*, Fort Collins, CO.
- Wallace, J.B., and N.H. Anderson. 2008. Habitat, Life History, and Behavioral Adaptations of Aquatic Insects. Pages 41–73. In: R.W. Merritt and K.W. Cummins (Eds.), *An Introduction to the Aquatic Insects of North America*. Third. Kendall Hunt Publishing Company, Dubuque, IA.
- Wallmo, O.C. 1981. Mule and Black-Tailed Deer Distribution and Habitats. Pages 1-26. In: Wallmo, O.C. (Ed.), *Mule and Black-Tailed Deer of North America*. University of Nebraska Press, Lincoln, NE.
- Walsh, E.J., M.L. Arroyo, T. Schröder, and R.L. Wallace. 2014. Species richness and species turnover (complementarity) of Rotifera in selected aquatic systems of Big Bend National Park, Texas. Pages 185–204. In Hoyt, C.A. and J. Karges (Eds.), *Proceedings of the Sixth Symposium on the Natural Resources of the Chihuahuan Desert Region*. Chihuahuan Desert Research Institute, Fort Davis, TX.
- Ward, J.V., K. Tockner, D.B. Arscott, and C. Claret. 2002. Riverine landscape diversity. *Freshwater Biology* 47:517–539.
- Ward, K.O. 1977. Two year vegetation response and successional trends for spring burns in pinyon-juniper woodlands (Thesis). University of Nevada, Reno, Reno, NV, USA.
- Waring, G.L., and N.S. Cobb. 1992. The impact of plant quality on herbivore population dynamics: the case of plant stress. In: Bernays, E.A. (Ed.), *Focus on Insect-Plant Interactions*, Vol III. CRC Press, Boca Raton, FL, USA.
- Warnock, B.J. 2006. Restoration of the Duff Springs riparian area and the associated upland grasses to increase pronghorn habitat. Pages 65-68. In: Cearley, K.A., and S. Nelle (Eds.), *Pronghorn Symposium*.

Presented at the Pronghorn Symposium, Alpine, TX.

- Warren, A., Holechek, J., Cardenas, M. 1996. Honey mesquite influences on Chihuahuan desert vegetation. *Journal of Range Management* 49:46–52.
- Waser, P.M., and J.M. Ayers. 2003. Microhabitat use and population decline in banner-tailed kangaroo rats. *Journal of Mammalogy* 84:1031–1043. doi:10.1644/BBa-032
- Waser, P.M., and W.T. Jones. 1991. Survival and reproductive effort in banner-tailed kangaroo rats. *Ecology* 72:771. doi:10.2307/1940579
- Watts, D.A., and G.W. Moore. 2011. Water-use dynamics of an invasive reed, *Arundo donax*, from leaf to stand. *Wetlands* 31:725–734.
- Waterfall, U.T. 1946. Observations on the Desert Gypsum Flora of Southwestern Texas and Adjacent New Mexico. *The American Midland Naturalist*, 36(2):456–466.
- Weaver, J.H., L.A. Harveson, K. Waldrup, S.S. Gray, and B.L. Tarrant. 2012. An investigation into the recent pronghorn decline in the Trans-Pecos region of Texas. Pages 49-50. In: *Proceedings of the Trans-Pecos Wildlife Conference-2012*. Presented at the Trans-Pecos Wildlife Conference - 2012, Alpine, TX.
- Webb, R.H., and J.L. Betancourt. 1992. Climatic effects on flood frequency of the Santa Cruz river, Pima County, Arizona. U.S., Water-Supply Paper 2379. U.S. Geological Survey.
- Webb, R.H., and S.A. Leake. 2006. Ground-water surface-water interactions and long-term change in riverine riparian vegetation in the southwestern United States. *Journal of Hydrology* 320:302–323.
- Weber, D.J., E.D. Bunderson, J.N. Davis, D.L. Nelson, and A. Hreha. 1999. Diseases and environmental factors of the pinyon-juniper communities. Pages 118-120. In: *Monsen, S.B., and R. Stevens (Eds.), Proceedings: Ecology and Management of Pinyon Juniper Communities within the Interior West*. RMRS-P-9. U.S. Department of Agriculture, Forest Service, Ogden, UT, USA.
- Webster, G.L., and C.J. Bahre. (Eds.) 2001. *Changing plant life of La Frontera: Observations on vegetation in the United States/Mexico borderlands*, 1st ed. University of New Mexico Press, Albuquerque, NM.
- Wellard Kelly, H.A., E.J. Rosi-Marshall, T.A. Kennedy, R.O. Hall, W.F. Cross, and C.V. Baxter. 2013. Macroinvertebrate diets reflect tributary inputs and turbidity-driven changes in food availability in the Colorado River downstream of Glen Canyon Dam. *Freshwater Science* 32:397–410.
- Wenzel, R.P. 1994. A new hantavirus infection in North America. *New England Journal of Medicine* 330:1004–1005. doi:10.1056/NEJM199404073301410
- West, N.E. 1999. Distribution, composition, and classification of current juniper-pinyon woodlands and savannas across western North America. Pages 20-23. In: *Monsen, S.B., and R. Stevens (Eds.), Proceedings: Ecology and Management of Pinyon Juniper Communities within the Interior West*. RMRS-P-9. U.S. Department of Agriculture, Forest Service, Ogden, UT, USA.
- West, N.E. 1999. Piñon-Juniper savannas and woodlands of Western North America, in: *Anderson, R.C., Fralish, J.S., Baskin, J.M. (Eds.), Savannas, Barrens, and Rock Outcrop Plant Communities of North America*. Cambridge University Press, New York, NY, USA.
- West, N.E., K.H. Rea, and R.J. Tausch. 1975. Basic synecological relationships in pinyon-juniper woodlands. Pages 41-52. In: *Gifford, G.F., and F.E. Busby (Eds.), The Pinyon-Juniper Ecosystem: A Symposium*. Utah State University, Logan, UT, USA.
- Westerling, A.L., H.G. Hidalgo, D.R. Cayan, T.W. Swetnam. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. *Science* 313:940-943.
- Western Regional Climate Center (WRCC). 2016. Cooperative climatological data summaries. *Jornada*

Experimental Range, NM. <http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?nmjorn> Accessed November 1, 2016.

- Whitaker, J.O. 1996. National Audubon Society field guide to North American mammals. Alfred A. Knopf, Inc., New York, NY, USA.
- White, J.A., C.M. Giggelman, and P.J. Connor. 2006. Recommended Water Quality for Federally Listed Species in Texas, Austin, TX: U.S. Fish and Wildlife Service, Region 2, Environmental Contaminants Program, Technical Report.
- White, J.D., K.J. Gutzwiller, W.C. Barrow, L. Johnson-Randall, L. Zygo, P. Swint. 2011. Understanding Interaction Effects of Climate Change and Fire Management on Bird Distributions through Combined Process and Habitat Models. *Conservation Biology* 25:536–546. doi:10.1111/j.1523-1739.2011.01684.x
- White, W.H., N.P. Hyslop, K. Trzepla, S. Yarkin, R.S. Rarig, T.E. Gill, and L. Jin. 2015. Regional transport of a chemically distinctive dust: Gypsum from White Sands, New Mexico (USA). *Aeolian Research*, 16:1-10.
- White, W.N., H.S. Gale, and S.S. Nye. 1941. Geology and Ground-Water Resources of the Balmorhea Area Western Texas, in: Contributions to the Hydrology of the United States, 1940. U.S. Geological Survey, Water-Supply Paper 849-C, Washington, D.C., pp. 83–146.
- Whitford, W., and K. Bixby. 2006. The last desert grasslands: The biological case for protecting Otero Mesa, 1st ed. The Southwest Environmental Center, Las Cruces, NM.
- Whitford, W.G. 1997. Desertification and animal biodiversity in the desert grasslands of North America. *Journal of Arid Environments* 37:709–720. doi:10.1006/jare.1997.0313
- Whitford, W.G. 2002. Ecology of Desert Systems. Academic Press, San Diego, CA.
- Whitford, W.G. and B.T. Bestelmeyer. 2006. Chihuahuan Desert fauna: Effects of ecosystem properties. In: Havstad K.M., L.F. Huenneke, and W.H. Schlesinger (Eds.), Structure and Function of a Chihuahuan Desert Ecosystem: the Jornada Basin Long-Term Ecological Research Site. New York: Oxford University Press. <http://jornada.nmsu.edu/files/bibliography/06-063.pdf>. Accessed September 30, 2016.
- Whitford, W.G., A.G. De Soyza, J.W. Zee, J.E. van, Herrick, and K.M. Havstad, K.M. 1998. Vegetation, soil, and animal indicators of rangeland health. *Environmental monitoring and assessment*. 51:179–200.
- Whitney, J.E., K.B. Gido, and D.L. Propst. 2014. Factors associated with the success of native and nonnative species in an unfragmented arid-land riverscape. *Canadian Journal of Fisheries and Aquatic Sciences* 71:1134–1145.
- Whitney, J.E., K.B. Gido, T.J. Pilger, D.L. Propst, and T.F. Turner. 2015. Consecutive wildfires affect stream biota in cold- and warmwater dryland river networks. *Freshwater Science* 34:000–000.
- Whitney, J.E., K.B. Gido, T.J. Pilger, D.L. Propst, and T.F. Turner. 2016. Metapopulation analysis indicates native and non-native fishes respond differently to effects of wildfire on desert streams. *Ecology of Freshwater Fish* 25:376–392.
- White, J.D., K.J. Gutzwiller, W.C. Barrow, L. Johnson-Randall, L. Zygo, and P. Swint. 2011. Understanding interaction effects of climate change and fire management on bird distributions through combined process and habitat models: Combined vegetation-bird models. *Conservation Biology* 25:536–546. doi:10.1111/j.1523-1739.2011.01684.x
- Wiken, E., F.J. Nava, and G. Griffith. 2011. North American Terrestrial Ecoregions—Level III. Commission for Environmental Cooperation, Montreal, Canada.
- Wild, C. 2011. Beaver as a Climate Change Adaptation Tool: Concepts and Priority Sites in New Mexico. Seventh Generation Institute, Santa Fe, NM.

- Wilkins, D.E. 1997. Hemiarid basin responses to abrupt climatic change: Paleolakes of the Trans-Pecos closed basin. *Physical Geography*, 18(5):460–477.
- Wilkinson, M.C. 1971. Reconstruction of Historical Fire Regimes along an elevation and vegetation gradient in the Sacramento Mountains, New Mexico (Thesis). University of Arizona Press, Tucson, AZ, USA.
- Williams, G.P., and M.G. Wolman. 1984. Downstream Effects of Dams on Alluvial Rivers. U. S. Geological Survey, Professional Paper 1286, Washington, D.C.
- Wilson, J.L., Tkacz, B.M. 1992. Pinyon ips outbreak in pinyon juniper woodlands in northern Arizona: a case study. Pages 187-190. In: Ffolliott, P.F., G.J. Gottfried, D.A. Bennett, V.M. Hernandez, A. Ortega-Rubio, R.H. Hamre (Eds.), *Ecology and Management of Oak and Associated Woodlands: Perspectives in the Southwestern United States and Northern Mexico*. GTR-RM-218. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO, USA.
- Winemiller, K.O., and A.A. Anderson. 1997. Response of endangered desert fish populations to a constructed refuge. *Restoration Ecology* 5:204–213.
- Wisdom, M.J., Chambers, J.C. 2009. Concepts for ecologically-based management of Great Basin shrublands. *Restoration Ecology* 17:740–749.
- Witte, C.L. 2005. Epidemiologic Analysis of Risk Factors for Local Disappearances of Native Ranid Frogs in Arizona. M.S. Thesis, University of Maryland, College Park, MD.
- Wittler, R.J., D.R. Levis, J.E. Klawon, R.E. Klinger, B.P. Greimann, M.R. Delcau, and J.F. England. 2004. Upper Gila River Fluvial Geomorphology Study Final Report Arizona. U.S. Department of the Interior, Bureau of Reclamation, Technical Service Center, Denver, CO.
- Wohl, E. 2006. Human impacts to mountain streams. *Geomorphology* 79:217–248.
- Wohl, E., B.P. Bledsoe, R.B. Jacobson, N.L. Poff, S.L. Rathburn, D.M. Walters, and A.C. Wilcox. 2015. The natural sediment regime in rivers: Broadening the foundation for ecosystem management. *BioScience* 65:358–371.
- Wolaver, B.D., J.M. Sharp, J.M. Rodriguez, and J.C.I. Flores. 2008. Delineation of regional arid karstic aquifers: an integrative data approach. *Ground Water* 46:396–413.
- Wondzell, S.M., J.M. Cornelius, and G.L. Cunningham. 1990. Vegetation patterns, microtopography, and soils on a Chihuahuan desert playa. *Journal of Vegetation Science* 1:403–410.
- Wood, K.A., M.T. O’Hare, C. McDonald, K.R. Searle, F. Daunt, and R.A. Stillman. 2016. Herbivore regulation of plant abundance in aquatic ecosystems. *Biological Reviews*. doi:10.1111/brv.12272.
- Woodbury, A.M. 1947. Distribution of pigmy conifers in Utah and northeastern Arizona. *Ecology* 28:113–126.
- Woodin, H.E., and A.A. Lindsey. 1954. Juniper-pinyon east of the Continental Divide, as analyzed by the line-strip method. *Ecology* 35:473–489.
- World Wildlife Fund and Sky Island Alliance (WWF-SIA). 2007. Natural Heritage of the Peloncillo Mountain Region, Appendixes. In: Bodner, G., J.A. Montoya, R. Hanson, W. Anderson (Eds.), *Natural Heritage of the Peloncillo Mountain Region*. World Wildlife Fund and Sky Island Alliance, Tucson, AZ.
- Worthington, R.D., and R.D. Corral. 1987. Some effects of fire on shrubs and succulents in a Chihuahuan Desert community in the Franklin Mountains, El Paso County, Texas, No. 3. Pages 1-8. In: Powell, A.M. (Ed.), *Contributed Papers of the Second Symposium on Resources of the Chihuahuan Desert Region United States and Mexico*. Chihuahuan Desert Research Institute, Alpine, Texas, USA.
- Worthington, T.A., S.K. Brewer, N. Farless, T.B. Grabowski, and M.S. Gregory. 2014. Interacting effects of

discharge and channel morphology on transport of semibuoyant fish eggs in large, altered river systems. *PloS one* 9:e96599.

- Wright, H.A. 1971. Why squirreltail is more tolerant to burning than needle-and-thread. *Journal of Range Management* 24:277–284.
- Wright, H.A. 1972. Shrub response to fire. Pages 204-217. In: McKell, C.M., J.P. Blaisdell, J.R. Goodin (Eds.), *Wildland Shrubs--Their Biology and Utilization*. GTR-INT-1. US Department of Agriculture, Forest Service, Intermountain Experiment and Range Station, Ogden, UT, USA.
- Wright, H.A. 1974. Range Burning. *Journal of Range Management* 27:5. doi:10.2307/3896428
- Wright, H.A., Bailey, A.W. 1980. Fire ecology and prescribed burning in the Great Plains--a research review, General Technical Report 77. US Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT, USA.
- Wright, H.A., and A.W. Bailey. 1982. *Fire Ecology: United States and southern Canada*. John Wiley and Sons, New York, NY, USA.
- Wright, H.A., S.C. Bunting, and L.F. Neuenschwander. 1976. Effect of fire on honey mesquite. *Journal of Range Management* 29:467–471.
- Wright, H.A., L.F. Neuenschwander, and C.M. Britton. 1979. The role and use of fire in sagebrush-grass and pinyon-juniper plant communities: A state-of-the-art review. GTR-INT-58. US Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT, USA.
- Wright, J.W., and C.H. Lowe. 1968. Weeds, polyploids, parthenogenesis, and the geographical and ecological distribution of all-female species of *Cnemidophorus*. *Copeia* 1968:123–138.
- Wright, R.A., and G.M. Van Dyne. 1976. Environmental factors influencing semidesert grassland perennial grass demography. *Southwestern Naturalist* 21:259–274.
- Yanoff, S., and E. Muldavin. 2008. Grassland–shrubland transformation and grazing: A century-scale view of a northern Chihuahuan Desert grassland. *Journal of Arid Environments* 72:1594–1605. doi:10.1016/j.jaridenv.2008.03.012
- Yao, J., D.P.D. Peters, R.P. Gibbens, and K.M. Havstad, K.M. 2002. Response of perennial grasses to precipitation in the northern Chihuahuan Desert: implications for grassland restoration. *Bulletin of the Ecological Society of America* 83:302.
- Yoakum, J.D. 2004a. Distribution and Abundance. Pages 75-105. In: O’Gara, B.W., and J.D. Yoakum (Eds.), *Pronghorn Ecology and Management*. University Press of Colorado, Boulder, CO.
- Yoakum, J.D. 2004b. Relationships with Other Herbivores. Pages 503-538. In: O’Gara, B.W., and J.D. Yoakum (Eds.), *Pronghorn Ecology and Management*. University Press of Colorado, Boulder, CO.
- Yoakum, J.D. 2004c. Habitat Characteristics and Requirements. Pages 409-446. In: O’Gara, B.W., and J.D. Yoakum (Eds.), *Pronghorn Ecology and Management*. University Press of Colorado, Boulder, CO.
- Yoakum, J.D. 2004d. Foraging Ecology, Diet Studies and Nutrient Values. Pages 447-502. In: O’Gara, B.W., and J.D. Yoakum (Eds.), *Pronghorn Ecology and Management*. University Press of Colorado, Boulder, CO.
- Yoakum, J.D. 2004e. Habitat Conservation. Pages 571-630. In: O’Gara, B.W., and J.D. Yoakum, J.D. (Eds.), *Pronghorn Ecology and Management*. University Press of Colorado, Boulder, CO

- York, J.C., and W.A. Dick-Peddie. 1969. Vegetation changes in southern New Mexico during the past hundred years. Pages 157-199. In: McGinnies, W.F., and B.J. Goldman (Eds.), *Arid Lands in Perspective*. University of Arizona Press, Tucson, AZ, USA.
- Young, J.A., and R.A. Evans. 1978. Population dynamics after wildfires in sagebrush grasslands. *Journal of Range Management* 31:283–289.
- Young, V.A. 1956. The effects of the 1949-1954 drought on the ranges of Texas. *Journal of Range Management* 9:139–142.
- Zedler, P.H., C.R. Gautier, and G.S. McMaster. 1983. Vegetation change in response to extreme events: the effect of a short interval between fires in California chaparral and coastal scrub. *Ecology* 64:809–818.
- Zwartjes, P.W., J.-L.E. Cartron, P.L. Stoleson, W.C. Haussamen, and T.E. Crane. 2005. Assessment of native species and ungulate grazing in the Southwest: terrestrial wildlife (Gen. Tech. Rep. No. RMRS-GTR-142). U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO, USA.
- Zymonas, N.D., and D.L. Propst. 2007. Ecology of Blue Sucker and Gray Redhorse in the Lower Pecos River, New Mexico, 2000-2006. New Mexico Department of Game and Fish, Conservation Services Division, Santa Fe, NM.

20 Glossary

Assessment Management Team (AMT): BLM’s team of BLM staff and partners that provides overall guidance to the REA regarding ecoregional goals, resources of concern, conservation elements, CAs, MQs, tools, methodologies, models, and output work products. The team generally consists of BLM State Resources Branch Managers from the ecoregion, a point of contact (POC), and a variety of agency partners depending on the ecoregion.

Attribute: A defined characteristic of a geographic feature or entity.

Change Agent (CA): An environmental phenomenon or human activity that can alter/influence the future status of resource condition. Some CAs (e.g., roads) are the result of direct human actions or influence. Others (e.g., climate change, wildland fire, or invasive species) may involve natural phenomena or be partially or indirectly related to human activities.

Community: Interacting assemblage of species that co-occur with some degree of predictability and consistency.

Conservation Element (CE): A renewable resource object of high conservation interest often called a conservation target by others. For purposes of this TO, conservation elements will likely be types or categories of areas and/or resources including ecological communities or larger ecological assemblages.

Development: A type of change (CA) resulting from urbanization, industrialization, transportation, mineral extraction, water development, or other non-agricultural/silvicultural human activities that occupy or fragment the landscape or that develops renewable or non-renewable resources.

Ecological Integrity: The ability of an ecological system to support and maintain a community of organisms that have the species composition, diversity, and functional organization comparable to those of natural habitats within the ecoregion.

Ecological Status: The condition of an ecological community or system relative to its known, or predicted historical range of variability.

Ecological System: In this REA, ecological systems are defined as groups of plant communities that tend to co-occur within landscapes with similar ecological processes, substrates, and/or environmental gradients; the term is used to refer to ecological systems as classified by Nature Serve (Comer et al. 2003) and mapped by NatureServe (2013)

Ecoregion: An ecological region or ecoregion is defined as an area with relative homogeneity in ecosystems. Ecoregions depict areas within which the mosaic of ecosystem components (biotic and abiotic as well as terrestrial and aquatic) differs from those of adjacent regions (Omernik and Bailey 1997).

Ecosystem: The interactions of communities of native fish, wildlife, and plants with the abiotic or physical environment.

Analysis Extent: Every REA addresses an area slightly larger than its Level-III ecoregion(s), termed the “analysis extent,” that includes all watersheds that overlap the Level-III boundaries. The analysis extent for

the Chihuahuan Desert REA overlaps with the analysis extents for the Madrean Archipelago and Southern Great Plains REAs.

Fire Regime: Description of the patterns of fire occurrences, frequency, size, severity, and sometimes vegetation and fire effects as well, in a given area or ecosystem. A fire regime is a generalization based on fire histories at individual sites. Fire regimes can often be described as cycles because some parts of the histories usually get repeated, and the repetitions can be counted and measured, such as fire return interval (LANDFIRE 2016).

Fragmentation: The separation or division of habitats by intervening infrastructure (e.g., roads or utility corridors) or anthropogenic land uses (development, agriculture); as patches of habitat are increasingly divided into smaller and smaller units or increasingly isolated from other patches of habitat, their utility as habitat may be lost.

Geographic Information System (GIS): A computer system designed to collect, manage, manipulate, analyze, and display spatially referenced data and associated attributes.

Grid Cell, Grid Unit: When used in reference to raster data, a grid cell is equivalent to a pixel (also see *pixel*). When a raster data layer is converted to a vector format, the pixels may instead be referred to as grid cells.

Habitat: A place where an animal or plant normally lives for a substantial part of its life, often characterized by dominant plant forms and/or physical characteristics.

Hydrologic Unit: An identified area of surface drainage within the U.S. system for cataloging drainage areas, which was developed in the mid-1970s under the sponsorship of the Water Resources Council and includes drainage-basin boundaries, codes, and names. The drainage areas are delineated to nest in a multilevel, hierarchical arrangement. The hydrologic unit hierarchical system has four levels and is the theoretical basis for further subdivisions that form the *watershed boundary dataset* containing the 5th and 6th levels. (Seaber et al. 1987).

Invasive Species: Species that are not part of (if exotic non-natives), or are a minor component of (if native), an original community that have the potential to become a dominant or co-dominant species if their future establishment and growth are not actively controlled by management interventions, or that are classified as exotic or noxious under state or federal law. Species that become dominant for only one to several years (e.g., in a short-term response to drought or wildfire) are not invasive (modified from BLM Handbook 1740-2, Integrated Vegetation Handbook. (BLM 2008)

Key Ecological Attribute: Key ecological attributes include defining physical, biological, and ecological characteristics of a Conservation Element, along with its abundance and/or spatial distribution. When one or more key ecological attributes of a CE become stressed in a specific setting, i.e., are altered so that they depart significantly from long-term historic conditions, the entire Conservation Element in that setting is degraded or, in extreme circumstances, will disappear. A well-constructed conceptual model for a Conservation Element necessarily identifies a *limited* set of key ecological attributes to represent the overall condition of the CE. Ecosystem complexity, the limits of scientific knowledge, and the constraints of budgets prevent evaluation of all possible characteristics and processes of any single resource. The key ecological attributes identified in the conceptual ecological models for the fourteen Conservation Elements for the

Chihuahuan Desert REA served as crucial guides for identifying datasets for analysis during the Assessment phase of the REA.

Management Questions: Questions from decision-makers that usually identify problems and request how to fix or solve those problems.

Metadata: The description and documentation of the content, quality, condition, and other characteristics of geospatial data.

Native Plant and Animal Populations and Communities: Populations and communities of all species of plants and animals naturally occurring, other than as a result of an introduction, either presently or historically in an ecosystem (BLM 2001).

Native Species: Species that naturally occur in a particular geographic area and were not introduced by humans.

Natural Heritage Program: An agency or organization, usually based within a state or provincial natural resource agency, whose mission is to collect, document, and analyze data on the location and condition of biological and other natural features (such as geologic or aquatic features) of the state or province. These programs typically have particular responsibility for documenting at-risk species and threatened ecosystems. (See natureserve.org/ for additional information on these programs.)

Pixel: A pixel is a cell or spatial unit comprising a raster data layer; within a single raster data layer, the pixels are consistently sized; a common pixel size is 30 x 30 meters square. Pixels are usually referenced in relation to spatial data that are in raster format. In this REA, some pixels sizes included 30 x 30 m and 2 x 2 km (also see *Grid Cell, Grid Unit*).

Population: Individuals of the same species that live, interact, and migrate through the same niche and habitat.

Rapid Ecoregional Assessment (REA): The methodology used by the BLM to assemble and synthesize that regional-scale resource information, which provides the fundamental knowledge base for devising regional resource goals, priorities, and focal areas, on a relatively short time frame (within 2 years).

Status: Formally, the Global or State conservation status of a species (e.g., “extinct,” “vulnerable,” “threatened,” etc.). Informally, the presence/absence, abundance, or other measure of the condition of an ecological resource relative to some reference condition.

Stressor: A factor causing negative impacts to the biological health or ecological integrity of a CE. Factors causing such impacts may or may not have anthropogenic origins. In the context of the REAs, these factors are generally anthropogenic in origin.

Watershed: A watershed is the 5th-level, 10-digit unit of the hydrologic unit hierarchy. Watersheds range in size from 40,000 to 250,000 acres. Also used as a generic term representing a drainage basin or combination of hydrologic units of any size (see Hydrologic Unit).

Wildland Fire: Any non-structure fire that occurs in the wildland. Three distinct types of wildland fire have been defined and include wildfire, wildland fire use, and prescribed fire (LANDFIRE 2016).

21 Acronyms

AMT	Assessment Management Team
BLM	Bureau of Land Management
CA	Change Agent
CCSP	Climate Change Science Program
CDIMN	Chihuahuan Desert Inventory and Monitoring Network
CDRI	Chihuahuan Desert Research Institute
CE	Conservation Element
CEC	Commission for Environmental Cooperation
CHD	Chihuahuan Desert
CI	Confidence Interval
COG	Council of Governments
CWCS	Comprehensive Wildlife Conservation Strategy
DLO	Driver-Linkage-Outcome
EPA	Environmental Protection Agency
ERP	Ecosystem Restoration Program
HUC	Hydrologic Unit Code
IBWC	International Boundary Waters Commission
ICLUS	Integrated Climate and Land Use Scenarios
ILAP	Integrated Landscape Assessment Project
IUCN	International Union for Conservation of Nature
MHP	Montane-Headwater Perennial
MQ	Management Question
MVEDA	Mesilla Valley Economic Development Alliance
NAFTA	North American Free Trade Agreement
NASS	National Agriculture Statistics Service
NHNM	National Heritage New Mexico
NMDGF	New Mexico Department of Game and Fish
NMEMNRD	New Mexico Energy, Minerals and Natural Resources Department
NMOGA	New Mexico Oil and Gas Association
NMOSE	New Mexico Office of the State Engineer
NPS	National Park Service
NRCS	Natural Resource Conservation Service
ORV/OHV	Off Road Vehicle/Off Highway Vehicle
PDSI	Palmer Drought Severity Index
PNVT	Potential Natural Vegetation Type
PRISM	Parameter-elevation Regressions on Independent Slopes Model
REA	Rapid Ecoregional Assessment
SENMEDD	Southeast New Mexico Economic Development District
SGCN	Species of Greatest Conservation Need

STM	State-Transition Model
TPWD	Texas Parks and Wildlife Department
TWDB	Texas Water Development Board
USAEC	U.S. Army Corps of Engineers
USAF	U.S. Air Force
USBR	U.S. Bureau of Reclamation
USDA	U.S. Department of Agriculture
USEIA	U.S. Energy Information Administration
USEPA	U.S. Environmental Protection Agency
USFS	U.S. Forest Service
USGS	U.S. Geological Survey
USLE	Universal Soil Loss Equation
USFWS	U.S. Fish and Wildlife Service
VDDT	Vegetation Dynamics Development Tool
WRCC	Western Regional Climate Center
WSNM	White Sands National Monument
WWF-SIA	World Wildlife Fund-Sky Island Alliance