Chapter 7

Maintaining and Restoring Sustainable Ecosystems in Southern Nevada

Jeanne C. Chambers, Burton K. Pendleton, Donald W. Sada, Steven M. Ostoja, and Matthew L. Brooks

Introduction

Managers in southern Nevada are challenged with determining appropriate goals and objectives and developing viable approaches for maintaining and restoring sustainable ecosystems in a time of rapid socio-ecological and environmental change. Sustainable or "healthy" ecosystems supply clean air, water and habitat for a diverse array of plants and animals. As described in Chapter 1, sustainable ecosystems retain characteristic processes like hydrologic flux and storage, geomorphic processes, biogeochemical cycling and storage, biological activity and productivity, and population regeneration and reproduction over the normal cycle of disturbance events (modified from Chapin and others 1996 and Christensen and others 1996). Ecological restoration of stressed or disturbed ecosystems is an integral part of managing for sustainable ecosystems. The Society for Ecological Restoration International (SERI) defines ecological restoration as the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (SERI 2004).

Many of southern Nevada's ecosystems are being subjected to anthropogenic stressors that span global, regional, and local scales (Chapter 2), and are crossing ecological thresholds to new alternative states (Chapter 4 and Chapter 5). These alternative states often represent novel communities with disturbance regimes that differ significantly from historic conditions. Past management and restoration goals often focused on returning ecosystems to pre-disturbance conditions (Harris and others 2006). This approach assumes stable or equilibrium conditions and ignores changes in ecosystem processes due to land uses, increases in CO_2 concentrations, and climate change. A more realistic approach is to base management and restoration goals on the current potential of an ecosystem to support a given set of ecological conditions, and on the likelihood of future change due to a warming climate (Harris and others 2006). This approach requires understanding ecosystem resilience to anthropogenic disturbance and climate change, the alternative states that exist for ecosystems, and the factors that result in threshold crossings (Bestelmeyer and others 2009; Hobbs and Harris 2001; Stringham and others 2003; Whisenant 1999). It also requires the ability to predict how climate is likely to influence ecosystems in the future (Harris and others 2006).

This chapter addresses the restoration aspects of Sub-goal 1.3 in the SNAP Science Research Strategy which is to restore and sustain proper function of southern Nevada's watersheds and landscapes (table 1.3; Turner and others 2009). The effects of global, regional and local stresses on southern Nevada ecosystems are presented in Chapter 2. Here, we discuss appropriate objectives and develop guidelines for maintaining and restoring southern Nevada ecosystems. We then discuss the differences in ecological resilience to stress and disturbance and resistance to invasive species in southern Nevada ecosystems and describe restoration and management approaches for the different ecosystem types. We conclude with knowledge gaps and management implications.

Resistance and Resilience of Southern Nevada Ecosystems

The overarching objective for restoration and management of southern Nevada ecosystems is to maintain and restore sustainable ecosystems that are resilient to stress and disturbance and resistant to invasion. Resilience is defined as the capacity of an ecosystem to regain its fundamental structure, processes, and functioning (or recover) when subjected to stressors or disturbances like drought, livestock grazing, or wildfire (e.g., Allen and others 2005; Holling 1973; Walker and others 1999). In this context, resilience is a function of the underlying ecosystem attributes and processes that determine ecosystem recovery. Resistance is the capacity of an ecosystem to retain its fundamental structure, processes, and functioning (or remain largely unchanged) despite stresses, disturbances or invasive species. Resistance to invasion is a function of the biotic and abiotic factors and ecological processes in an ecosystem that limit the establishment and population growth of an invading species (D'Antonio and Thomsen 2004). The abiotic and biotic attributes and ecosystem processes that determine resilience to stressors and resistance to invasion can be illustrated with a simple conceptual model (fig. 7.1). Environmental characteristics as defined by climate, topography, and soils determine the abiotic and biotic attributes and processes of an ecosystem. In turn, the abiotic and biotic attributes and processes provide feedbacks to one another and determine the inherent potential of an ecosystem to support a given set of ecological conditions and plant species. Over time, climate, disturbance and stressors affect the abiotic and biotic attributes and processes and determine the current ecological conditions of the system. The current ecological conditions, as influenced by the legacy of past disturbances and stressors, determine resilience to disturbance and resistance to invaders at any point in time.

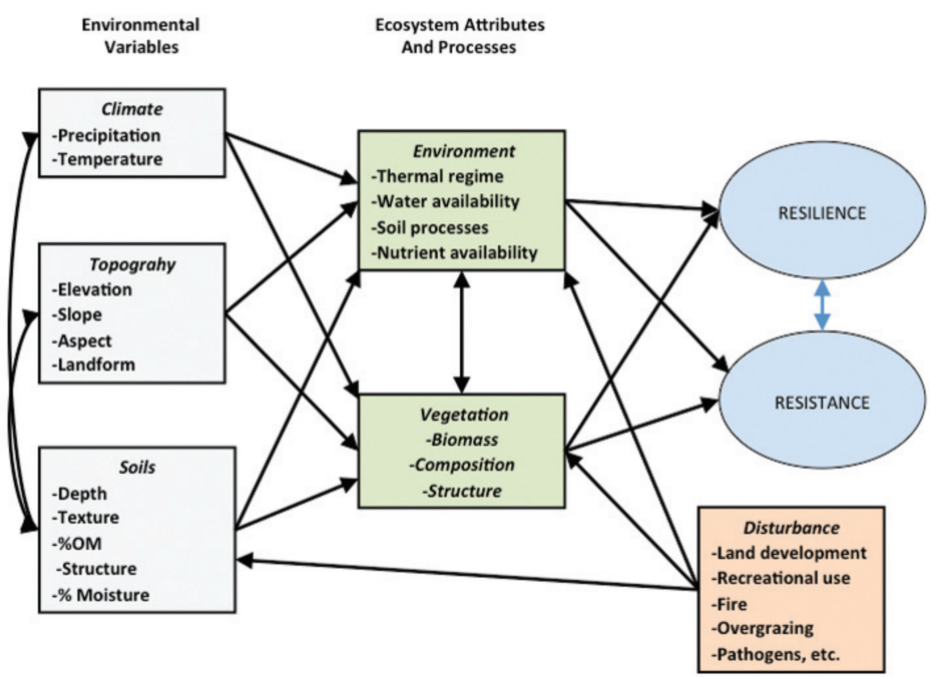

Figure 7.1—The environmental variables and site conditions that influenced resilience to disturbance and resistance to invasion. Disturbances can decrease ecological site conditions and negatively affect resilience and resistance.

Southern Nevada ecosystems differ in ecological resistance and resilience because of strong elevation/climate gradients and large differences in their environmental characteristics (Chapter 1). The Clark County Multiple Species Habitat Conservation Plan (MSHCP) categorizes 11 ecosystems based on elevation and soil moisture (fig. 7.2). In general, temperature regimes and effective precipitation are the primary drivers of ecological processes and determine overall resource availability and ecosystem productivity. The resilience of southern Nevada ecosystems to stresses typically increases along these environmental/productivity gradients (Brooks and Chambers 2011). These gradients also determine the likelihood that climate conditions are suitable for establishment of non-native grasses and other invaders (e.g., Chambers and others 2007; Condon and others 2011). Ecosystems influenced by elevated water tables and high levels of soil moisture are in a separate category, as environmental conditions can vary considerably among these ecosystems. Factors like soil and water chemistry are important drivers of ecosystem processes for these ecosystems.

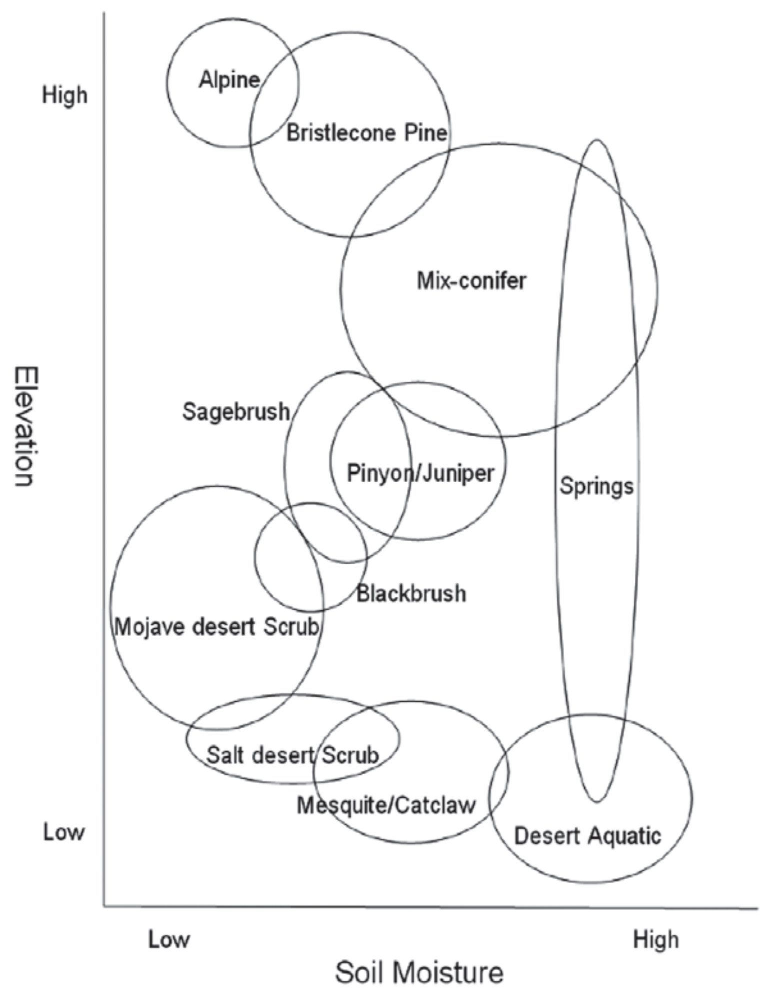

Figure 7.2—A conceptual model that categorizes 11 ecosystems of the Clark County MSHCP along two environmental gradients: elevation and soil moisture. This model is based on general knowledge of environmental gradients of ecosystems. The shape, size, and relative position of the ellipses and circles are hypothetical (from Desert Research Institute 2008).

Restoration Considerations for Southern Nevada Ecosystems

Restoration and management priorities and activities differ significantly among southern Nevada ecosystems because of large variation in their resilience and resistance. Overarching strategies are protection, prevention, and restoration (table 7.1). Passive restoration to eliminate or minimize stress is a component of protection and prevention; active restoration is a component of prevention and restoration. Guidelines for the restoration and management of the diverse ecosystems in southern Nevada can be developed based on an understanding of their relative resilience and resistance, the dominant stressors that affect them, and the actions most appropriate to maintain and restore them (table 7.2).

Table 7.1—An approach for categorizing management activities in southern Nevada ecosystems based on protection, prevention and restoration (modified from D'Antonio and Chambers 2006; Brooks and Chambers 2011).

Protection	
Focus	Ecosystems with low resilience and/or resistance, ecosystems of high conservation concern, and ecosystems at risk of crossing ecological thresholds to new alternative states.
Objectives	Eliminate or minimize current and future stressors.
Activities	Closure or active control of recreational use and burro and cattle grazing to allow natural regeneration; fire suppression in Mojave Desert scrub, blackbrush and lower elevation sagebrush and piñon and juniper ecosystems to prevent an invasive annual grass-fire cycle; control of placement and development of road and utility corridors, urban expansion, and solar energy projects to minimize fragmentation and surface disturbance.
Prevention	
Focus	Ecosystems with inherently higher resilience and/or resistance that are in moderate to high ecological condition.
Objectives	Maintain or increase resilience and resistance of areas with declining ecological conditions.
Activities	Vegetation management to decrease risk of high severity fires, maintain understory composition, and prevent invasion; mechanical vegetation management treatments to decrease decadent or over-dense shrubs and increase perennial herbaceous vegetation.
Restoration	
Focus	Ecosystems known to respond favorably to restoration treatments and ecosystems of conservation concern.
Objectives	Increase resilience and resistance of ecosystems by revegetating or rehabilitating areas disturbed by fire, recreational activities, road and utility corridors, urban expansion, solar energy projects, and other surface disturbances. Provide assisted migration for species being displaced by climate change.
Activities	Soil surface stabilization to curtail dust; seedbed preparation to mitigate soil physical and chemical disturbance and provide favorable conditions for plant establishment; transplanting or seeding native species adapted to the local environment and climate warming.

Table 7.2—Resilience and resistance characteristics of the major ecosystem types in southern Nevada and guidelines for appropriate management actions.

Ecosystem	Resilience and resistance	Guidelines for appropriate management actions
Alpine and Bristlecone pine	*Resilience* – Very low to low. Extreme temperatures, short growing seasons, slow growth, and low establishment rates. Low capacity to adapt/migrate with climate warming. *Resistance* – Moderate to high. Few annual species are adapted to extreme environment; resistance may decrease as climate warms.	*Protection* – Primary emphasis. Minimize stress from recreational activities, including firewood gathering. Monitor changes in temperature and precipitation and in species distributions and community composition. *Prevention* – Rarely warranted except to suppress fires with potential to spread. *Restoration* – Rarely warranted except for assisted migration of trees or revegetation of areas with die-off. Information on species environmental and establishment requirements is required.
Mixed conifer	*Resilience* – Moderate to high. Relatively high precipitation, long growing seasons, and moderate growth and establishment rates. Potential to migrate upslope with climate warming. *Resistance* – Moderate to low. Multiple non-native invaders adapted to environmental conditions; competition with invaders from established native plants can be high.	*Protection* – Control inappropriate recreational activities and overgrazing; detect and eradicate invasive species. *Prevention* – Warranted to decrease fuel loads, restore understory composition, and decrease invasion. Potential for Wildland Fire Use and prescribed fire where risk of large or high severity fire is low and fire spread can be controlled, and for tree thinning followed by surface fire or pile burning in WUI and areas with higher fuel loads. However, more information is needed on the responses of southern Nevada ecosystems. *Restoration* – Warranted following surface disturbance or in areas with insufficient fire tolerant understory species for site recovery after fire. Seed burial (drilling) or transplanting natives adapted to local site conditions and climate warming preferred.
Piñon and Juniper	*Resilience* – Moderate. Moderate precipitation, long growing seasons, moderate to slow growth and establishment. Potential for die-off at lower elevations with climate warming. *Resistance* – Low. Many non-native invaders adapted to environmental conditions; competition from established shrubs and herbaceous species dependent on site productivity and ecological condition.	*Protection*- Control inappropriate recreational activities and overgrazing; detect and eradicate invasive species; suppress fires at lower elevations and that threaten ecosystem integrity. *Prevention*-Warranted to decrease fuel loads, restore understory composition, and decrease invasion. Focus is on mesic sites in early to intermediate stages of tree expansion, and in moderate to high ecological condition. Potential for Wildland Fire Use and prescribed fire on productive sites at high elevation; mechanical treatments more appropriate on sites with low productivity. *Restoration* – Warranted following surface disturbance and in areas with insufficient fire tolerant understory species for site recovery after fire. Seed burial (drilling) or transplanting natives adapted to local site conditions and climate warming preferred.
Sagebrush	*Resilience* – Moderate to low. Types at higher elevations and with deeper soils have moderate resilience; types at lower elevations and on shallow soils have low resilience. *Resistance* – Moderate to low. Types at higher elevations are more resistant to annuals invaders than those at lower elevations. Resistance generally decreases as site productivity or herbaceous perennial species and ecological condition decreases.	*Protection*–Control inappropriate recreational activities and overgrazing, detect and eradicate invasive species, suppress fires at lower elevations and that threaten ecosystem integrity. *Prevention*– Warranted to restore or maintain sagebrush types, and increase understory species and resistance to invaders. Focus is on resilient and resistant sites. Potential for Wildland Fire Use and prescribed fire to control tree expansion, and shrub mowing and selective herbicides to decrease competition from overstory sagebrush. Information on ecosystem response is needed. *Restoration* – Warranted following surface disturbance and in areas with insufficient fire tolerant understory species for site recovery after fire. Seed burial (drilling) or transplanting natives adapted to local site conditions and climate warming preferred. Livestock closures required post-restoration to facilitate recovery.

(continued)

Table 7.2—(Continued).

Ecosystem	Resilience and resistance	Guidelines for appropriate management actions
Blackbrush	*Resilience* – Low to very low. Low precipitation, moderately high temperatures, episodic recruitment. Potential to migrate upslope with climate warming. *Resistance* – Low to very low. Environmental conditions conducive to establishment of invasive annual bromes. Low competition from native species due to low productivity.	*Protection* – Primary emphasis. Suppress fires, actively control cattle and burro grazing and inappropriate recreational activities, detect and eradicate new invaders. *Prevention* – Not warranted under most circumstances. *Restoration* – Warranted following surface disturbance and in areas with insufficient fire tolerant understory species for site recovery after fire. Seed burial (drilling) or transplanting natives adapted to local site conditions and climate warming preferred. Livestock closures required after fire and restoration activities to facilitate recovery of native perennial plants.
Mojave Desert scrub	*Resilience* – Very low. Extreme environmental conditions, episodic recruitment, slow ecosystem recovery. *Resistance*– Low. Environmental conditions of more mesic systems conducive to establishment of annual grasses; few species adapted to most extreme conditions. Low competition from natives due to low productivity.	*Protection* – Primary emphasis. Suppress fires, actively control inappropriate recreational activities and overgrazing by cattle, horses and burros, detect and eradicate new invaders. *Prevention* – Not warranted under most circumstances. *Restoration* – Warranted following surface disturbance and in areas with insufficient fire tolerant understory species for site recovery after fire. Seed burial (drilling) or transplanting natives adapted to local site conditions and climate warming preferred. Livestock closures required after fire and restoration to facilitate recovery of native perennial plants.
Riparian and Spring	*Resilience* –Low to moderately high. High water availability but high water temperatures, harsh water chemistry, and scouring floods. Water availability likely to decrease with climate warming. *Resistance* – Low. Many invasive species in a variety of taxa adapted to high availability of water.	*Protection*– Maintain or increase current water allocations and in stream flows, actively control inappropriate land uses, recreational activities and overgrazing, detect and eradicate new invaders. *Prevention*–Warranted to reduce non-native tamarisk and Russian olive and to manage fuels. Biocontrol, prescribed fire, mechanical treatments, or herbicides can be used, but restoration of native species must follow. *Restoration*– Warranted to maintain river and stream channels by manipulating flow regimes, and to restore or create habitat for native species of concern. Methods include manipulating water depths, velocities and temperatures to meet requirements for species establishment and persistence, and revegetating with native species adapted to the site conditions.

Consideration of the predicted effects of climate change on the different ecosystems and the implications for management will be needed to maintain and restore southern Nevada ecosystems. Climate change models predict high rates of temperature increase for desert ecosystems like the Mojave (Loarie and others 2009). By 2100, climate change is likely to result in the disappearance of some existing climate conditions, the appearance of some novel climate conditions, and the formation of new communities with no past or present analogs (Williams and Jackson 2007). Bioclimatic envelope models predict shifts in the distributions of keystone species like creosote bush (*Larrea tridentata*) (Rehfeldt and others 2006) and Joshua tree (*Yucca brevifolia*) (Cole and others 2011), and of invasive species like cheatgrass (Bradley and others 2009). Due to the rapid rate of change, many species may require assisted migration, and "transformative" restoration may be needed in areas that no longer have the climate conditions necessary to support the current set of species (Harris and others 2006; Bradley and others 2009).

Alpine Ecosystem

Alpine ecosystems occur on the Spring Mountains above 3,500 m. They are comprised of alpine fell-fields on exposed rocky, dry soils, and alpine meadows that occur in swales where soil moisture is higher and sand and silt soils accumulate (Clokey 1951). They have generally low resilience to disturbance and low productivity due to short growing seasons, temperature extremes, and slow growth rates (Chambers 1995). Few invasive species are adapted to the extreme environmental conditions. Because alpine species have slow growth and low establishment rates, the potential for adapting to a warming climate is low; because of their locations at the top of mountain ranges, the capacity to migrate also is low.

Modeling of species-environment relationships and potential changes in plant species distributions with a 3 °C increase in temperature for the White Mountains, California, indicated that species distributions would shift upwards and decrease in extent but that specific outcomes would depend on species affinities for various soil types (Van de Ven and others 2007). For a similar temperature increase in Great Basin alpine ecosystems, predicted extinction rates were 44% for mammals, 23% for butterflies, and 17% for plants (Murphy and Weiss 1992).

Restoration and management goals for alpine ecosystems in southern Nevada necessarily focus on protection due to their low resilience (table 7.2). Human activities are generally low, but stress from recreational activities should be minimized. A key aspect of managing these ecosystems is monitoring the rate and magnitude of change in the abiotic environment (temperature and precipitation) and in species distributions and community composition (Desert Research Institute 2008). North-facing slopes and certain soil types may serve as refugia for many native species (Van de Ven and others 2007), but these relationships are not well understood. Species loss in these fragile ecosystems from rapid warming may require assisted migration in the form of revegetation with species from lower elevation zones. Methods for restoring high elevation ecosystems are well-researched and include revegetation with seeds and transplants (Chambers 1997; Urbanska and Chambers 2002). Specific approaches for assisted migration, such as methods for species selection and matching species to newly available sites, have yet to be investigated.

Bristlecone Pine Ecosystem

The bristlecone pine ecosystem occurs in the Spring and Sheep Mountains at elevations of 2,700 m to 3,500 m on exposed, dry, rocky slopes and ridges in the subalpine zone (Pase and Brown 1982). This ecosystem is comprised of evergreen conifer forest dominated by widely spaced Great Basin bristlecone pine (*Pinus longaeva*) that frequently forms pure stands from the tree line down to its contact with limber pine (*P. flexilis*). Associated shrub species include dwarf juniper (*Juniperus communis*), Clokey mountain sage (*Salvia dorrii* var. *clokeyi*), and sagebrush (*Artemisia* spp.), but dense bristlecone pine forests often have low understory species richness and productivity (RECON 2000).

Similar to alpine ecosystems, cold temperatures, intense sunlight, low soil nutrients, a short growing season, and lengthy periods of snow cover result in low productivity and low ecosystem resilience (Holtmeier and Broll 2007). Recruitment is episodic and depends on local topography, soil types, short-term weather patterns and longer term climate, and, for bristlecone pines, seed predation/caching by rodents and birds. Recent research shows tree-ring growth in bristlecone pine within the region over the last century that is unmatched in millennia (3,700 yrs) indicating environmental changes that are probably linked to increases in temperature (Salzera and others 2009).

An advance of treeline has been documented for mountain ranges that have already exhibited temperature increases, and is predicted for other ranges as the climate warms (Grace and others 2002). Although seedling establishment of bristlecone pine may increase with warmer temperatures, the species may become more susceptible to pathogens and disease. White pine blister rust (*Cronartium ribicola*) was recently reported in Rocky Mountain bristlecone pine (*P. aristata*) (Blodgett and Sullivan 2004), and Great Basin bristlecone pine is a potential host for white pine blister rust (Kliejunas and Adams 2003). Mountain pine beetle (*Dendroctonus ponderosae*) may reproduce more rapidly due to climate warming and cause greater damage to bristlecone pines. Currently, fires are caused by lightning and are small and infrequent. Higher fuel loads due to warmer weather and increased tree growth plus deadwood caused by bark beetles may increase fire frequency (Desert Research Institute 2008).

As for alpine ecosystems, restoration and management goals for bristlecone pine ecosystems in southern Nevada focus on protection (table 7.2). Human activities primarily involve recreational use and firewood gathering for campfires, which can damage trees and initiate fires. These types of uses should be discouraged. Management should include monitoring the rate and magnitude of change in the abiotic environment (temperature and precipitation) and in species distributions and community composition (Desert Research Institute 2008). Understanding the population dynamics of bristlecone pine (recruitment and mortality) is essential for determining if assisted migration of the trees or revegetation of areas with high mortality is required. Understanding the environmental and establishment requirements of both the trees and associated species is necessary to determine appropriate revegetation methods.

Mixed Conifer Ecosystems

The mixed conifer ecosystem is comprised of three tree and shrub dominated communities that occur at 1,200 m and 3,200 m in elevation (RECON 2000): the white fir, ponderosa pine, and ponderosa pine/mountain shrub communities. The white fir community occurs in the Spring and Sheep Mountains on north and east-facing slopes at elevations between 2,200 and 3,200 m and is dominated by white fir *(Abies concolor*). Other tree species include bristlecone pine (*P. longaeva*) and limber pine (*P. flexilis*) at higher elevations, and ponderosa pine (*P. ponderosa*) at lower elevations. The ponderosa pine community covers the largest area of any conifer forest in southern Nevada, ranges from 1,200 m to 2,700 m in elevation, and is dominated by ponderosa pine. Associated tree species include white fir, bristlecone pine, limber pine, singleleaf piñon (*P. monophylla*), Utah juniper (*Juniperus osteosperma*), and mountain mahogany (*Cercocarpus* spp.). The ponderosa pine/mountain shrub community has less ponderosa pine and is co-dominated by mountain shrubs, like oak (*Quercus gambelii)*, mountain mahogany, snowberry (*Symphoricarpos albus*), and manzanita (*Arctostaphylos* spp.). Relatively high precipitation, mild temperatures, and long growing seasons result in moderately high ecological resilience in the mixed conifer ecosystem. Multiple non-native invaders are adapted to these communities decreasing ecological resistance.

Interactions among climate, fire, and pine bark beetles strongly affect the structure and composition of mixed conifer ecosystems. Overall growth rates in southern Nevada mixed conifer ecosystems are low, reflect those in other southwest ecosystems, and are strongly influenced by drought (Biondi and others 2011). In general, trees respond positively to winter-spring precipitation and negatively to spring-summer temperature (Biondi and others 2011). Recruitment is associated with moist and fire-free periods, while die-off is related to droughts lasting more than 10 years (Brown and Wu 2005). The likelihood of pine bark beetle outbreaks is increased by higher temperatures and forest homogeneity, and trees are most susceptible to beetles when drought stressed or after fire (Raffa and others 2008). As the climate warms, increases in pine bark beetle outbreaks and die-offs are likely for trees growing at the margin of their ecological tolerances. Upslope movement of tree populations will require favorable conditions for tree recruitment and lack of widespread fire.

Increases in tree densities and fuel loads and reductions in plant species diversity have occurred in many southwest mixed conifer forests due to factors like favorable climate for tree recruitment, overgrazing, and fire suppression (Allen and others 2002). Fuel loads and fire risk are increasing in some mixed conifer communities in southern Nevada (Abella and others 2011). Fire regimes and fire return intervals depend on the aridity and topographic characteristics of the site. Fire regimes vary from high frequency, low severity to mixed severity (Biondi and others 2011; Jamieson 2008; Kilpatrick 2009; Kitchen 2010). On an arid site (Mt. Irish), the fire return interval for fires that scarred at least 10% of recorder trees was 66 years (Biondi and others 2011). On a mesic site (Clover Mountains), comparable fire return intervals ranged from 17 to 34 years (Kilpatrick 2009). Many stands in the ponderosa pine community have limited extent and are characterized by old age trees, especially on drier mountain ranges (fig. 7.3). Understory species composition affects resilience to both fire and fuel treatments due to effects on tree regeneration, fire behavior and soil erosion (Allen and others 2002). Many of the understory shrubs in the ponderosa pine/mountain shrub community are fire tolerant and promote recovery after fire (fig. 7.3). Perennial herbaceous species increase resistance to annual grasses in all community types (Chambers and others 2007).

Restoration and management goals for mixed conifer ecosystems include protection and restoration, but emphasize prevention (table7.2). Most campgrounds and recreational activities occur in this type. Allen and others (2002) list 16 broad principles for restoring southwestern ponderosa pine ecosystems including preventing or minimizing crown fire, restoring or maintaining understory composition, and preventing invasion by non-native species. The focus is on resilient and resistant stands with climatic conditions and understory species that will ensure recovery. Considerable information exists on specific fire and fuels treatments for managing ponderosa pine and mixed confer ecosystems (Brown and others 2000). In a ponderosa pine community in Grand Canyon National Park, wildland fire use decreased fuel loads, reduced duff layers, and increased species richness of annual and biennial forbs (Laughlin and others 2004). Tree thinning coupled with understory burning decreased the risk of crown fire and increased stand resilience, but tree kill and understory response depended on climate and pre-treatment ecological conditions in other southwest ponderosa pine ecosystems (Fule and others 2005; Moore and others 2006). Because of the aridity and small extent of many of southern Nevada's mixed conifer ecosystems, and the number of species of conservation concern, the emphasis should be on use of thinning, and mechanical treatments until additional information is available on responses to fire treatments. Information also is needed on restoring landscape heterogeneity and the effects of climate change.

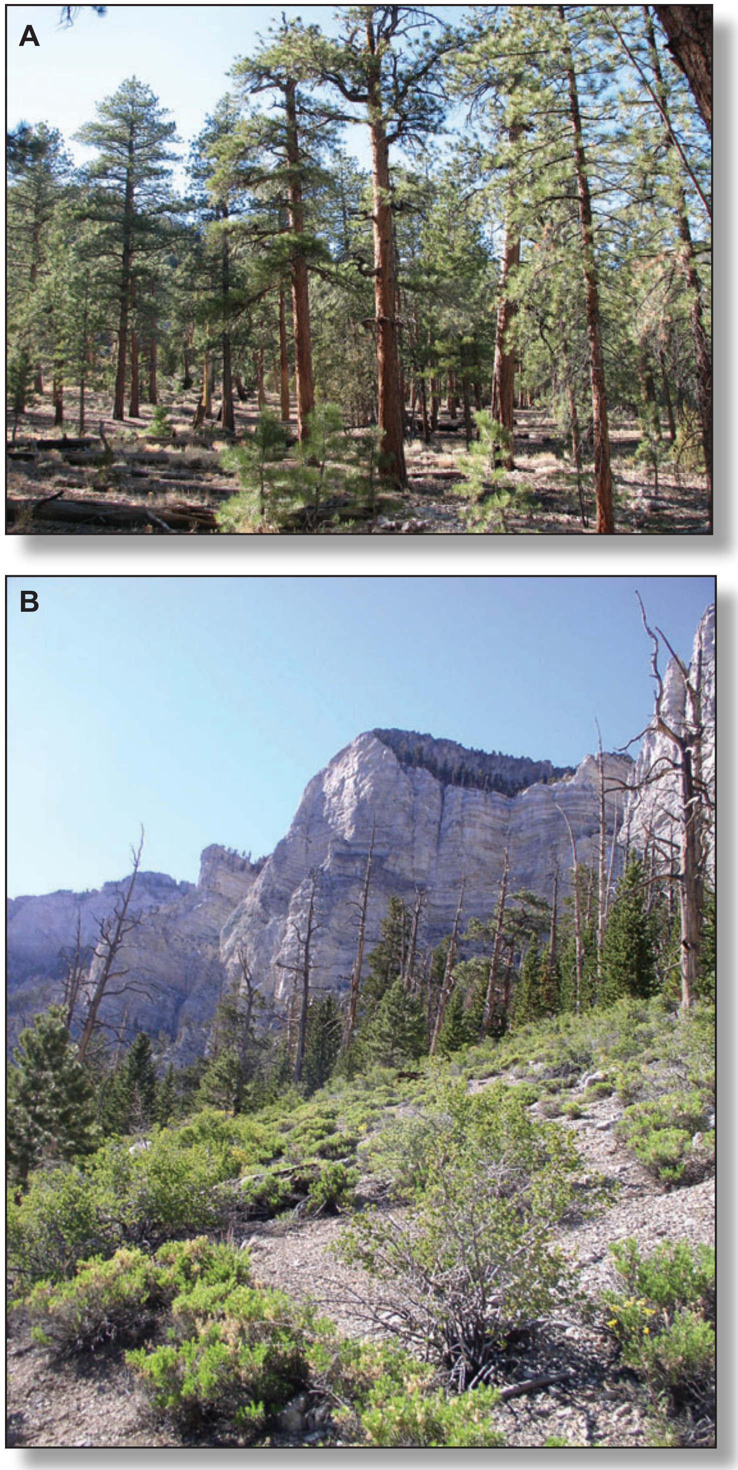

Figure 7.3—(A) An arid ponderosa pine community on the Sheep Range with a sparse understory of sagebrush. Many of these communities are of limited extent and are characterized by old aged trees. Historic fire return intervals were long and preventative management to control fuels needs to be exercised with caution. (B) A ponderosa pine/mountain shrub community on the Spring Mountains after a wildfire. This community is characterized by shrub species that resprout after fire and promote site recovery. Preventative management should focus on preventing crown fires and maintaining the understory (photos by Patti Novak-Echenique).

Piñon and Juniper Ecosystem

The piñon and juniper ecosystem occurs from 1,500 m to 2,500 m in the Spring, Sheep, Virgin, and McCullough Mountains (RECON 2000). Sagebrush co-exists with piñon and juniper at all elevations. At higher elevations, singleleaf piñon dominates, but co-occurs with other coniferous trees and shrubs like Gambel's oak (*Quercus gambelii*) and mountain mahogany (*Cercocarpus* spp.). At lower elevations, Utah juniper (*J. osteosperma*) dominates and Rocky Mountain juniper (*J. scopulorum*), western juniper (*J. occidentalis*), rabbitbrush (*Chrysothamnus* spp.) and blackbrush (*Coleogyne ramosissima*) are minor components. Singleleaf piñon and Utah juniper co-dominate at middle elevations. Relatively high precipitation, mild temperatures, and long growing seasons result in moderate resilience, but local conditions influence disturbance and treatment outcomes. Many non-native invaders are adapted to these communities and ecological resistance is low.

Piñon and juniper ecosystem in the southwestern United States respond to the high variation in precipitation with major pulses of woody plant establishment and mortality (Swetnam and Betancourt 1998). Water stress can trigger rapid and extensive dieback (Breshears and others 2005). Broadscale tree mortality can shift ecotones between vegetation types, alter regional distributions of overstory and understory vegetation, and change disturbance processes such as fire and erosion (Allen and others 2010). Drought-associated water stress also can increase susceptibility of trees to insects and other pathogens (Breshears and others 2005). The piñon ips beetle (*Ips confusus*) occurs in southern Nevada and is a species that can undergo broadscale outbreaks following high temperatures and drought (Breshears and others 2005).

Piñon and juniper expansion and infilling are occurring in southern Nevada shrublands due to a variety of factors including climate change, increased CO_2 concentrations, overgrazing, and fire suppression (fig. 7.4; Abella and others 2011; Miller and others 2008). The increase in trees has the potential to significantly increase fuel loads. Fire frequency, size, and severity in piñon and juniper ecosystems are strongly influenced by fuel loads and climate. Recent fire history studies in central Nevada and Mesa Verdi National Park, Colorado, indicate that historical fire regimes were characterized by small, infrequent and high severity fires, and that stand replacing fires occurred about every 200 to 400+ years (Bauer and Weisberg 2009; Floyd and others 2004; Romme and others 2009). Fires occurred with higher frequency during droughts and on more mesic sites (Bauer and Weisberg 2009).

Restoration and management goals in piñon and juniper ecosystems include protection and restoration, but emphasize prevention (table 7.2). Restoration goals include preventing or minimizing crown fire, restoring or maintaining shrubs and perennial herbaceous species in the understory, and preventing invasion by non-native species. The focus is on resilient and resistant ecosystems with sufficient perennial herbaceous species and shrubs for recovery. Resilient piñon and juniper ecosystems are typically on more mesic sites, in the early to intermediate stage of tree expansion (i.e., phase I to phase II woodlands; *sensu* Miller and others 2005), and in moderate to high ecological condition (fig. 7.4). Both prescribed fire and mechanical treatments, like shredding and cutting and leaving the trees, are used to reduce fuel loads and increase resilience in these ecosystems (Miller and others 2005; Monsen and others 2004; Pyke 2011). Mechanical treatments are typically of lower severity than prescribed fire and are used on sites with more severe environmental conditions and with a high risk of fire-tolerant invaders like cheatgrass. Because of the aridity of many of southern Nevada's piñon and juniper ecosystems, and the number of species of conservation concern, the emphasis should be on thinning and mechanical treatments until additional information is available on responses to fire treatments. Information also is needed on restoring landscape heterogeneity and the effects of climate change.

A

B

C

Figure 7.4— (A) Piñon expansion into a mountain big sagebrush ecological site type on the Desert National Wildlife Refuge. Preventative management to maintain the understory sagebrush community using mechanical tree removal, Wildland Fire Use or prescribed fire can be considered on sites with favorable climatic conditions and sufficient perennial herbaceous species and shrubs for recovery. (B) Old age piñon on a black sagebrush ecological site on the Desert National Wildlife Refuge. Protective management of old age stands located on harsh ecological sites should be considered. (C) A dense piñon stand on the Spring Mountains. Preventative management using mechanical treatments to decrease fire risk should be considered in WUI areas. Due to tree competition and understory depletion, restoration will be required following fire (photos by Patti Novak-Echenique).

Sagebrush Ecosystem

The sagebrush ecosystem typically ranges in elevation from 1,500 m to 2,800 m in the Spring, Sheep, and Virgin Mountains (RECON 2000). In southern Nevada, sagebrush species include big sagebrush (*A. tridentata*), low sagebrush (*A. arbuscula*), Bigelow sagebrush (*A. bigelovii*), silver sagebrush (*A. cana*), and black sagebrush (*A. nova*). The dominant sagebrush communities differ in response to local topography, soil type, and water availability. Big sagebrush community types can occur as relatively, large, open, and discontinuous stands, but often occur with trees species (piñon pine, junipers, ponderosa pine, mountain mahogany) and other shrubs (bitterbrush, rabbitbrush, snakeweed [*Gutierrezia sarothrae*], blackbrush, shadscale [*Atriplex confertifolia*], and spiny hopsage [*Grayia spinosa*]) (fig. 7.4) (Clokey 1951). Ecological resilience is a function of local site conditions and the community type. Higher elevation types with greater precipitation and productivity like *A. tridentata* ssp. *vaseyana* have moderately high resilience while lower elevation and less productive types like *A. tridentata* ssp. *wyomingensis* and *A. nova* have lower resilience (Brooks and Chambers 2011). Resistance decreases as elevation and productivity decrease (Chambers and others 2007).

Risks to sagebrush ecosystems include overgrazing, land use change, piñon and juniper expansion, invasion of non-native plants, and altered fire regimes (Wisdom and others 2005). Climate change poses a substantial additional risk. Sagebrush species are likely to respond to climate warming by moving northward or upslope in response to shifts in frost lines (Neilson and others 2005). Risk analyses of sagebrush types in southern Nevada that assessed the interactive effects of land use conversion, land use (roads, agriculture, etc.), and cheatgrass invasion indicated that sagebrush communities in southern Nevada were at greater risk of losing suitable habitat due to climate change than due to disturbance (Bradley 2010).

In southwestern and arid ecoregions, precipitation in seasons prior to the fire season is more highly associated with burn area than warmer temperatures or drought the year of fire due to the importance of fine fuel production (e.g., invasive annual grasses) (Littell and others 2009). However, increasing aridity may result in a decrease in area burned due to a reduction in fine fuels.

Restoration and management goals and methods vary for these diverse ecosystems due to differences in resilience and resistance (table 7.2). In general, the focus of restoration is on maintaining and restoring a desirable proportion of sagebrush types, increasing the abundance of perennial understory species that promote resilience, and increasing resistance to invasive species. Strategies for sagebrush types with inherently low resilience and resistance focus on protection and include eliminating stressors like inappropriate recreation, overgrazing, and fire. Prevention can be an effective strategy for sagebrush types with higher resilience and resistance. Wildland Fire Use and prescribed fire have been used in higher elevation types exhibiting tree expansion (fig. 7.4A), but additional information on fire effects are needed for southern Nevada ecosystems. Mowing and selective herbicides have been used to increase perennial herbaceous species by reducing competition from decadent or over-dense sagebrush. Restoring sagebrush ecosystems that have crossed ecological thresholds to invasive grass dominance is expensive, difficult, and of lower priority. Although methods exist for protection, prevention, and restoration of mesic sagebrush ecosystems (e.g., D'Antonio and others 2009; Monsen and others 2004; Pyke 2011), we still lack the necessary tools to manage and restore more arid sagebrush ecosystems. Sagebrush ecosystems are highly susceptible to climate change and we know little about assisted migration or transformative restoration.

Blackbrush Ecosystem

The blackbrush ecosystem occurs at elevations between 1,250 and 1,800 m in the transition zone between the Mojave Desert and Great Basin (RECON 2000). At upper elevations it integrates into the Utah juniper community, while at lower elevations it transitions into the creosote-bursage community. It is the dominant shrub in the understory of most Joshua tree communities. Shrubs associated with blackbrush include spiny hopsage, mormon tea, shadscale, desert thorn, and snakeweed. The blackbrush ecosystem has very little resilience to disturbance due to low effective precipitation, moderately high temperatures and episodic recruitment of blackbrush (Bowns 1973). Resistance to invasion by annual grasses, especially red brome (*Bromus madritensis* ssp. *rubens*), is extremely low because the environmental conditions that characterize this ecosystem are ideally suited to its establishment and reproduction.

Risks to the blackbrush ecosystem include overgrazing by cattle and burros, off-highway vehicle (OHV) and recreational activities, annual grass invasion, and altered fire regimes. Historically, the blackbrush ecosystem experienced small localized fires and recovery occurred within a few decades (Brooks and Matchett 2006). However, due to invasion by exotic annual grasses and an increase in fine fuel loads and fuel continuity, extensive areas of the blackbrush ecosystem have burned within the last decade (Chapter 4 and Chapter 5). Blackbrush is not fire-adapted and is incapable of resprouting following fire. Consequently, parts of this ecosystem are being converted to annual grass dominance and are susceptible to repeated burns (Brooks and others 2007).

In the blackbrush ecosystem, the focus of restoration and management is on protection coupled with restoration (table7.2). Protection of this ecosystem includes suppressing large fires and actively controlling overgrazing by wild horses, burros, and cattle. The characteristics of this ecosystem vary considerably over environmental/productivity gradients (fig. 7.5). Lower elevation blackbrush communities are characterized by thermic soils and often contain a minor component of creosote bush. Revegetation is difficult due to low spring and early summer moisture, increasing CO_2, and higher nighttime winter temperatures (Jones 2011; Zitzer 2009). The distribution of the creosote-bursage community is limited by nighttime winter temperatures (Webb and Bowers 1993) and is expected to move up-slope with climate change. Thus, at lower elevations within the blackbrush ecosystem, early seral shrubs from the creosote and bursage community may be better candidates for restoration than species from the blackbrush community. Higher elevation and more mesic blackbrush communities are characterized by mesic soils, and species from the blackbrush community are the best choice for seed mixes.

Considerable information exists on the establishment requirements of blackbrush including seed dispersal and longevity in the soil (Auger 2005; Zitzer 2009), seed dormancy, germination, and survival (Meyer and Pendleton 2005; Pendleton 2008; Pendleton and Meyer 2004), and the importance of mycorrhizae and biological soil crusts (Pendleton and others 1999). Blackbrush typically establishes under nurse plants, and establishment success of blackbrush may be increased by including other species with high seedling establishment in the seed mix (Abella and Newton 2009). Transplanting seedlings of blackbrush and the other dominant shrubs in the community into shrub islands that can serve as future seed sources also should be considered but several years of supplemental water may be required (Pendleton 2008; Winkel and others 1995). Blackbrush seldom produces large crops of seeds, but the seeds that are produced exhibit long-term viability (Pendleton and others, in press). This indicates that seeds can be harvested during mast years and stored until needed. Additional information is needed on relationships among

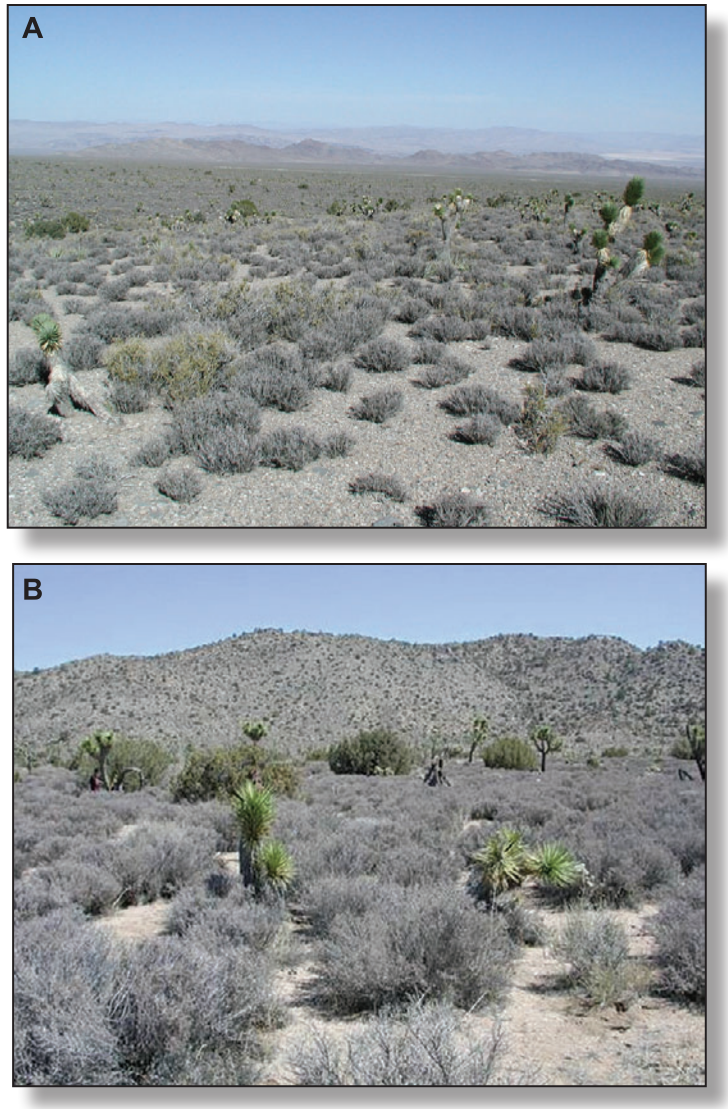

Figure 7.5—(A) A more arid blackbrush ecological site on thermic soils that exhibits low productivity and has a minor component of creosotebush (photo by Patti Novak-Echenique). (B) A more mesic blackbrush ecological site on mesic soils with slightly higher productivity. Note the presence of juniper (photo by Matt Brooks). Protective management should be emphasized on both site types. Due to ongoing climate change, restoration of the thermic site should include early seral species from the Mojave Desert scrub type.

climate, soil characteristics, and establishment of the species in both creosote/bursage and blackbrush communities if assisted migration and transformative restoration is to succeed.

Mojave Desert Scrub Ecosystem

Mojave Desert scrub occurs at elevations below 1,220 m and is the most widespread ecosystem in southern Nevada (~73% of Clark County) (RECON 2000). This type is characterized by three community types: Mojave Desert mixed scrub, mesquite/catclaw, and salt desert shrub.

Mojave Desert mixed scrub—Mojave Desert mixed scrub is unique because it includes a wide variety of distinctive soil types and plant communities often intermix. Bajadas, the most common landform, are dominated by creosote bush and white bursage (*Ambrosia dumosa*); subdominants include desert thorn (*Lycium andersonii*), shadscale, spiny hopsage, ratany (*Krameria erecta*), bladder sage (*Salazaria mexicana*), indigo bush (*Psorothamnus fremontii*), blackbrush, brittlebush (*Encelia farinosa*), and burro bush (*Hymenoclea salsola*) (RECON 2000). Sand dunes, gypsum soils, cliff/rock outcrops, and steep slopes occur as isolated patches that support Joshua tree, prickly pear cactus (*Opuntia spp.*), yucca (*Yucca* spp.), cholla (*Cylindropuntia* spp.), and hedgehog cactus (*Echinocereus* spp.).

Mojave Desert mixed scrub communities have low ecological resilience and have exhibited little resistance to invasive species (Brooks and Chambers 2011). Most long-lived Mojave Desert scrub species are poorly adapted to disturbances that remove aboveground biomass or kill plants, although most short-lived early seral species increase in abundance following disturbance. Reproduction of long-lived species depends on appropriate environmental conditions for either clonal regeneration and/or seedling establishment and is episodic (Webb and others 2009). The life history strategies of these species coupled with the harsh environment significantly increases the complexity of restoration.

Mojave Desert mixed scrub is subject to most of the stressors listed in Chapter 2, including urbanization, energy development, invasive species and the resulting increase in fire frequency, unregulated recreation, and OHV activity. Climate change further strains this ecosystem. The distribution of many Mojave Desert scrub species is limited by cold temperatures (Pockman and Sperry 1997; Webb and Bowers 1993), and climate models predict that Mojave Desert scrub species will move northward and upslope as temperatures continue to warm. One model predicts that northern Nevada and southern Idaho may have climates suitable for creosotebush by 2060 (Rehfelt and others 2006).

Invasion of non-native annual grasses (*Bromus madritensis* ssp. *rubens* and *Schismus barbatus*) and some forbs (e.g., *Brassica* spp.) has significantly increased fine fuel loads in Mojave Desert scrub vegetation and is resulting in unprecedented fires (Chapters 4 and 5). Years with high precipitation are associated with high fuel loads and large-scale fires (Brooks and Minnich 2006). Many woody desert species are fire intolerant and do not readily recover after fire; this results in progressive conversion to annual grass dominance, and an annual-grass fire cycle characterized by repeated fires (Brooks 2011).

Due to inherently low resilience and resistance of Mojave Desert scrub communities, restoration and management must focus on protection and restoration (table 7.2). In these ecosystems, protection includes eliminating or reducing stressors such as fire, dispersed recreational activities, OHV use, and overgrazing by wild horses, burros, and cattle. New utility-scale, alternative energy sites should be placed to minimize fragmentation and species loss.

The state-of-knowledge for restoring Mojave Desert scrub was recently summarized by Abella and Newton (2009), Webb and others (2009), and Weigand and Rogers (2009). Restoration activities should address the presence of invasive species, a potential increase in fire frequency, and current and projected climate conditions. Creosotebush, the dominant shrub, exhibits limited resprouting after disturbances that do not kill the plant; other shrubs like white bursage reestablish over time. Also, there has been some success with seeding and transplanting Mojave Desert scrub species when appropriate methods are used (Bainbridge 2007). Full recovery is slow and depends on an absence of repeated fire (Engel and Abella 2011). Similar to the blackbrush ecosystem, Mojave

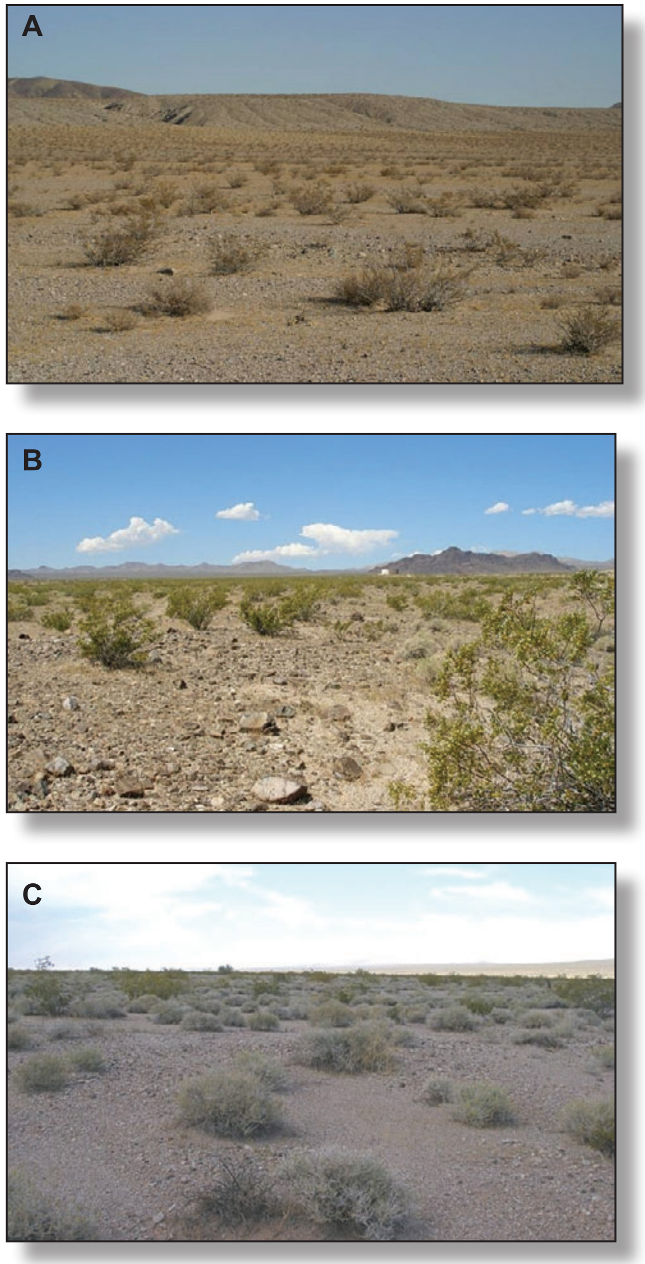

Figure 7.6—(A) An arid creosotebush ecological site with relatively low productivity (photo by Matt Brooks). (B) A slightly less arid creosotebush ecological site. (C) A severely disturbed site on Camp Ibis military base after 50 years of recovery (photos B and C by Patti Novak-Echenique). Protective management should be emphasized on both site types. Due to ongoing climate change, restoration of the less arid site should include species from the more arid type.

Desert mixed scrub communities are wide-ranging and occur over broad environmental gradients that influence resilience and restoration approaches (fig. 7.6). Some species in hotter and drier sites within Mojave Desert mixed scrub communities may be at or past thresholds of persistence due to the novel mix of stressors that include invasive species, warmer winters, and hot and potentially dryer summers. Recent increases in nighttime temperatures due to climate change have relaxed the biogeographical constraints on many desert species, and they can now establish at locations that were previously too cold (Kelly and Goulden 2008; Loarie and others 2009; Tylianakis and others 2008). Thus, restoration strategies for degraded sites at higher elevations should consider including species that are projected to migrate to these areas. Information is needed on the environmental tolerances of these species and their establishment requirements over the range of predicted climates.

Salt desert scrub community—The salt desert scrub ecosystem forms a mosaic within the Mojave Desert mixed scrub and blackbrush ecosystems. It is found between 800 and 1,800 m, and is associated with playas, basins, and poorly drained depressions with silty loam soils. The primary shrubs of the salt desert scrub community are shadscale, desert holly (*Atriplex hymenelytra*), Bailey's greasewood (*Sarcobatus baileyi*), desert thorn, Torrey saltbush, (*Atriplex torreyi*), winterfat (*Krascheninnikovia lanata*), bursage, fourwing saltbush (*Atriplex canescens*), mormon tea, horsebrush (*Tetradymia canescens*), and snakeweed (RECON 2000). Other shrubs include greasewood (*Sarcobatus vermiculatus*), blackbrush, iodine bush (*Allenrolfea occidentalis*), and creosotebush. This lower-elevation and very dry ecosystem has very low ecological resilience, but may have slightly higher resistance to invasive species than Mojave Desert mixed scrub due to low moisture availability. Salt desert scrub in the Great Basin is more resilient than in southern Nevada due to higher rainfall and cooler temperatures in the Great Basin.

Risks to the salt desert shrub ecosystem include utility-scale renewable energy development (particularly in playas), urbanization, burro and cattle grazing, recreational and OHV activity and accompanying dust production, invasive annual grasses, and associated increases in fire frequency. Urbanization is a primary concern as this ecosystem has the highest percentage loss to urban land development of all major ecosystems in the SNAP area (Chapter 2). Large areas of salt desert scrub have been lost to urban development in Green Valley/Henderson and in northwest Las Vegas. Climate change will cause additional stress.

Restoration and management priorities for this ecosystem are similar to those for Mojave Desert mixed scrub: control of recreational activities, including OHV use; careful selection of sites for alternative energy facilities; and maintenance of appropriate populations of burros. Limited information is available on restoration of salt desert shrub species, although seeding of *Atriplex* has had some success (Abella and Newton 2009). Maintaining this ecosystem in southern Nevada will require identifying those locations where it still occurs and prioritizing its protection in land use plans. Specific elements should include preventing fragmentation by maintaining minimum patch sizes.

Mesquite/catclaw community—The mesquite/catclaw community is a small component (~1%) within Mojave Desert scrub. It is comprised of screwbean mesquite (*Prosopis pubescens*), honey mesquite (*P. glandulosa*), and catclaw acacia (*Acacia greggii*) and occurs in patches (1 ha to >1,000 ha) where perennial groundwater is not more than 10 m from the surface. The greatest extent of the mesquite/catclaw community in southern Nevada was found previously in the Las Vegas Valley. Most of this community type now has been lost due to urbanization, OHV use, climate change and exotic species (Desert Research Institute 2008). Maintaining this community in southern Nevada will require identifying the locations where it still occurs and prioritizing its protection in land use plans to ensure that minimum patch sizes are maintained and fragmentation is prevented.

Riparian/Aquatic Ecosystems

In southern Nevada, the desert riparian/aquatic ecosystem occurs primarily along the Colorado River, Las Vegas Wash, and Virgin and Muddy Rivers at elevations below 600 m (RECON 2000). Water is perennial in the Colorado River, Las Vegas Wash, and Muddy River, but is intermittent in the Virgin River. In perennial reaches, the aquatic community includes fish and aquatic macroinvertebrates. The riparian community is characterized by woody, deciduous, and emergent obligatory and facultative wetland vegetation. Native woody vegetation includes Fremont cottonwood (*Populus fremontii*), black cottonwood (*Populus trichocarpa*), sandbar willow (*Salix exigua*), Goodding willow (*S. gooddingii*), velvet ash (*Fraxinus velutina*), desert willow (*Chilopsis linearis*), and honey mesquite. Woody vegetation along intermittent reaches is relatively sparse and consists mostly of desert willow and acacia (*Acacia* spp.). Due to high productivity and water availability, this ecosystem provides essential cover, water, food, and breeding sites for several endangered species and many other species of conservation concern. However, seasonally high water temperatures, harsh water chemistry, high turbidity, scouring floods, and sandy substrates result in moderate to low resilience. These systems exhibit low resistance to woody invaders like tamarisk (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*).

Riparian/aquatic ecosystems are some of the most degraded in southern Nevada (Desert Research Institute 2008). Stressors include altered flow regimes resulting from dams and diversions, impoundments, and channelization; invasive species and altered fire regimes; woodcutting; burro, wild horse, and cattle grazing; agricultural clearing; and urban and ex-urban development. The rapid rate of temperature increase and higher drought severity index predicted to occur with climate change (Loarie and others 2009) will likely result in decreased water availability for riparian/aquatic ecosystems.

Protection, prevention, and restoration are all important activities in these ecosystems (table 7.2). Surface water, groundwater, and sediment dynamics coupled with water and soil chemistry determine the composition and abundance of both aquatic organisms and plant species in these arid riparian/aquatic ecosystems (Anderson and others 2004; Briggs 1996). Specific activities are determined by the characteristics of the system and are usually conducted to maintain or improve channel and flow characteristics and to restore or create habitat for one or more species of concern. Protection is a cross-cutting activity that involves minimizing disturbance and sediment and chemical inputs from the diversity of stressors that affect these ecosystems. Prevention aimed at controlling invasive species also is a cross-cutting activity. Removal of woody invaders, including tamarisk and Russian olive, using biocontrol agents, like the saltcedar leaf beetle (*Diorhabda elongata*), herbicides, or cutting is widespread, but follow-up restoration activities and habitat trade-offs require careful consideration (Shafroth and others 2005; Shanahan and others 2011).

In the Lower Colorado River Basin, the Multi-Species Conservation Program focuses on four ecosystem types: cottonwood/willow, honey mesquite, marsh, or backwater. Active restoration/habitat creation involves assessing the physical, chemical, and biological conditions necessary for establishment and maintenance of species of concern (amphibians and native fish populations) and manipulating flow regimes to obtain the desired characteristics (Scoppettone and others 2005). This can include improving spring and stream connectivity, and restoring pools, riffles, and the natural substrate. The site is then revegetated with native plants adapted to the new conditions. In actively eroding systems like Las Vegas Wash, which has municipal and industrial wastewater inputs, erosion control, environmental monitoring, and wetlands restoration and enhancement are implemented to reduce erosion and address water-quality concerns (RECON 2000). Because of the importance of upstream activities and disturbances on downstream flows, sediment regimes and water quality, increased emphasis is being placed on large-scale watershed planning and regional collaboration (e.g., U.S. Army Corp of Engineers 2008).

Springs Ecosystems

Springs occur from approximately 250 m to 3,300 m elevation in all landscape settings (e.g., mountains, gullies, valley floors, hillside, etc.) in southern Nevada. The MSHCP reported 506 springs in Clark County. However, an inventory of the basic environmental and biological characteristics of most large springs (Sada and Nachlinger 1996, 1998) indicates that Clark County springs dry frequently and that less than 200 persistent springs occur in the county. The geology, climate, topographic position, aquifer size, and flow path of the water determine the hydrologic characteristics of springs. In turn, the hydrologic characteristics of springs structure spring environments and the biotic communities that they support (Sada and Herbst 1999, 2006; Sada and others 2001, 2000). In southern Nevada, springs are generally supported by mountain block, local, or regional aquifers with different environmental and biological characteristics (Knochenmus and others 2007; Mifflin 1968).

Springs in mountainous recharge areas are supported by mountain block aquifers. These springs are generally small and are perched, that is, they discharge water that flows along an impermeable layer near the soil surface. They are typically cold (<10 °C) with low chemical concentrations (specific conductivity <500 µmhos) and neutral pH (6.0 to 8.0). Harsh conditions in these springs are mostly attributed to natural factors such as periodic drying (seasonal or during droughts), scouring floods, and fire. Thus, they are characterized by moderately low resilience. Disturbances are mostly due to trampling and overgrazing by wild horses, burros, elk, and cattle, and recreational use. These springs are minimally affected by groundwater removal, but may be susceptible to drier conditions with climate warming due to low flows.

Local aquifers support springs that are often around the margins of valleys at lower elevations. These aquifers are generally larger than mountain block aquifers, and springs fed by local aquifers are more persistent, less affected by drought, and dry infrequently. Most local springs are cool (10 to <25 °C), their chemical concentrations are low (specific conductance <1000 µmhos), and their pH is generally neutral (6.0 to 8.0). These springs are characterized by moderately high resilience. However, most of these springs have been altered by livestock trampling, annual grass fires, diversions, and/or recreation. These systems also may be impacted by groundwater withdrawal, and decreased groundwater recharge due to climate change.

Springs that are supported by regional aquifers are generally large compared to mountain block aquifer and local aquifer springs. Regional aquifers extend through several topographic basins and encompass thousands of square kilometers. Most importantly, they are persistent over long periods of time (tens of thousands of years) and are minimally affected by drought conditions. Regional springs are warm (25 to 40 °C), their chemical concentrations are relatively benign (specific conductance of generally 500 to 1,000 µmhos, but may be as high as 1,500 µmhos), and their pH is generally neutral (near 7.4). These springs are minimally affected by natural events because they are large and located on valley floors where scouring floods are uncommon; thus, these springs also have moderately high natural resilience. However, most regional springs have been affected by past agricultural practices (pesticides, removal of vegetation, grazing by burros and cattle, ground water pumping, and surface diversions) and continue to be affected by altered flows, recreation, and grazing by horses, burros, and cattle.

As for riparian/aquatic ecosystems, restoration and management strategies for spring ecosystems must be based on their topographic setting, hydrologic characteristics, and macroinvertebrate and vegetation communities (Sada and Herbst 1999, 2006; Sada and others 2000, 2001). In addition, benthic macroinvertebrate and riparian communities of Mojave Desert and Great Basin springs generally differentiate along a physiochemical stress gradient where highly stressed springs support depleted communities composed of animals and plants that are more tolerant of harsh physicochemical environments than springs with less stressful environments (Fleishman and others 2006; Sada and others 2005). Protection of these systems requires maintaining sufficient water availability to support the existing aquatic organisms and riparian vegetation by reducing or minimizing ground water withdrawal and surface diversions. Protection also includes minimizing or eliminating stress due to grazing and trampling by cattle and burros and recreation. Prevention can be used to control invasive woody and herbaceous species. Restoration can be used to restore spring hydrology by eliminating diversions and vegetation communities by revegetation with the appropriate native species. For spring systems supported by regional aquifers with existing or potential stream channels, modification of flows can be used to create stream channels with the necessary water depths, velocities, and temperatures to support native species of concern, like the Amargosa pupfish (Scoppettone and others 2005).

Knowledge Gaps

Cross-cutting information needs include a better understanding of the factors that determine resilience and resistance in southern Nevada's diverse ecosystems and of the interacting effects of the region's stressors. Knowledge of the environmental conditions required for establishment and persistence of native plant species and methods for their restoration is also needed. Information needs specific to the region's stressors include climate change, land use, invasive species, and fire.

Climate Change. More accurate predictions of changes in both temperature and precipitation; ecosystem specific information on the effects of climate change on species distributions, disturbance regimes and recovery processes.

Land Use. Knowledge of the distribution and extent of current and future land uses and their effects on current and future ecosystem resilience; information on the minimum patch sizes and degree of fragmentation that ecosystems can tolerate; information on the amount and effect of N deposition on native ecosystems and annual invaders; land use planning tools to ensure land use is consistent with maintaining and restoring ecosystems.

Invasive Species. Increased knowledge of feedbacks to invasion from regional stressors like increased CO_2, altered fire regimes, and overgrazing; knowledge of the environmental conditions required for establishment and persistence of invasive plants and of their capacities to adapt and migrate in a warming environment; methods for controlling invaders and restoring natives that are consistent with ecosystem restoration and maintenance.

Fire. Increased knowledge of fire effects on annual species invasion and ecosystem recovery for the different ecological sites that characterize Mojave Desert scrub and blackbrush ecosystems; increased knowledge of the interacting effects of effective precipitation, ecosystem productivity, and understory species composition on fire return intervals for southern Nevada mixed conifer and piñon and juniper ecosystems; fire and fire surrogate tools for mixed conifer, piñon and juniper and sagebrush ecosystems.

Management Implications

Protection is a critical component of restoring and maintaining southern Nevada ecosystems due to the arid environment and numerous stressors. Preventative management is a viable option only in more mesic or higher elevation ecosystems that do not comprise much of the total land area. Restoration is challenging in all ecosystems. Maintaining sustainable ecosystems will require a greater focus on assessing ecological condition, prioritizing restoration and management activities, and selecting the most appropriate treatments. Monitoring and adaptive management will be essential.

Assessment and Prioritization

An integrated and consistent assessment of southern Nevada ecosystems and their relative resilience and resistance can be used to categorize and prioritize management and restoration activities. Addressing the widespread stressors affecting these ecosystems and providing habitat for species of concern requires a broad scale approach that crosses administrative boundaries. Most management plans now encompass landscapes with multiple project areas and are developed in consultation with partner agencies. Several tools already exist for developing landscape-scale and cross-boundary assessments. Soil surveys exist for most of southern Nevada including spring systems and lands managed by the BLM, most of Desert National Wildlife Refuge, and Lake Mead (USDA NRCS 2012). Soils characteristics, along with climate and topography determine the potential to support a given ecological site type (fig. 7.1). Draft ecological site descriptions (ESDs) exist for most of the region that has soils surveys (contact NRCS Nevada State Office, http://www.nv.nrcs.usda.gov/contact/). Soil types and ESDs can be used in a GIS environment as the basis for evaluating the relative resilience and resistance of the ecosystems in the region, and the degree to which current ecological conditions deviate from potential conditions. Recent research has developed geospatial tools for identifying critical habitat for species of concern in the Great Basin that could be used in southern Nevada (Meinke and others 2009). Methods also have been developed to examine linkages among adjacent ESDs and the interacting effects of landforms and disturbances (Bestelmeyer and others 2011).

The utility of this approach can be illustrated for the blackbrush ecosystem. ESDs are part of a land classification system that describes the potential of a set of climate, topographic, and soil characteristics and natural disturbances to support a dynamic set of plant communities (Bestelmeyer and others 2009; Stringham and others 2003). ESDs use state (a relatively stable set of plant communities that are resilient to disturbance) and transition (the drivers of change among alternative states) to describe the range in variation of plant communities (Stringham and others 2003). The reference state often includes several plant communities that differ in dominant plant species relative to time since disturbance. Alternative states describe new sets of communities that are separated by largely irreversible transitions (thresholds) and that may persist over time. A generalized state and transition model is presented for the blackbrush ecosystem (fig. 7.7). Different ecological sites occur within the blackbrush ecosystems that are differentiated by relative aridity (thermic vs. mesic soils). Alternative states and transitions differ for the two ecological site types and this has important implications in a warming environment.

Restoration and Management Approaches

Once an area has been prioritized for active restoration and/or management, a series of logical steps can be used to develop the restoration plan. These include identifying

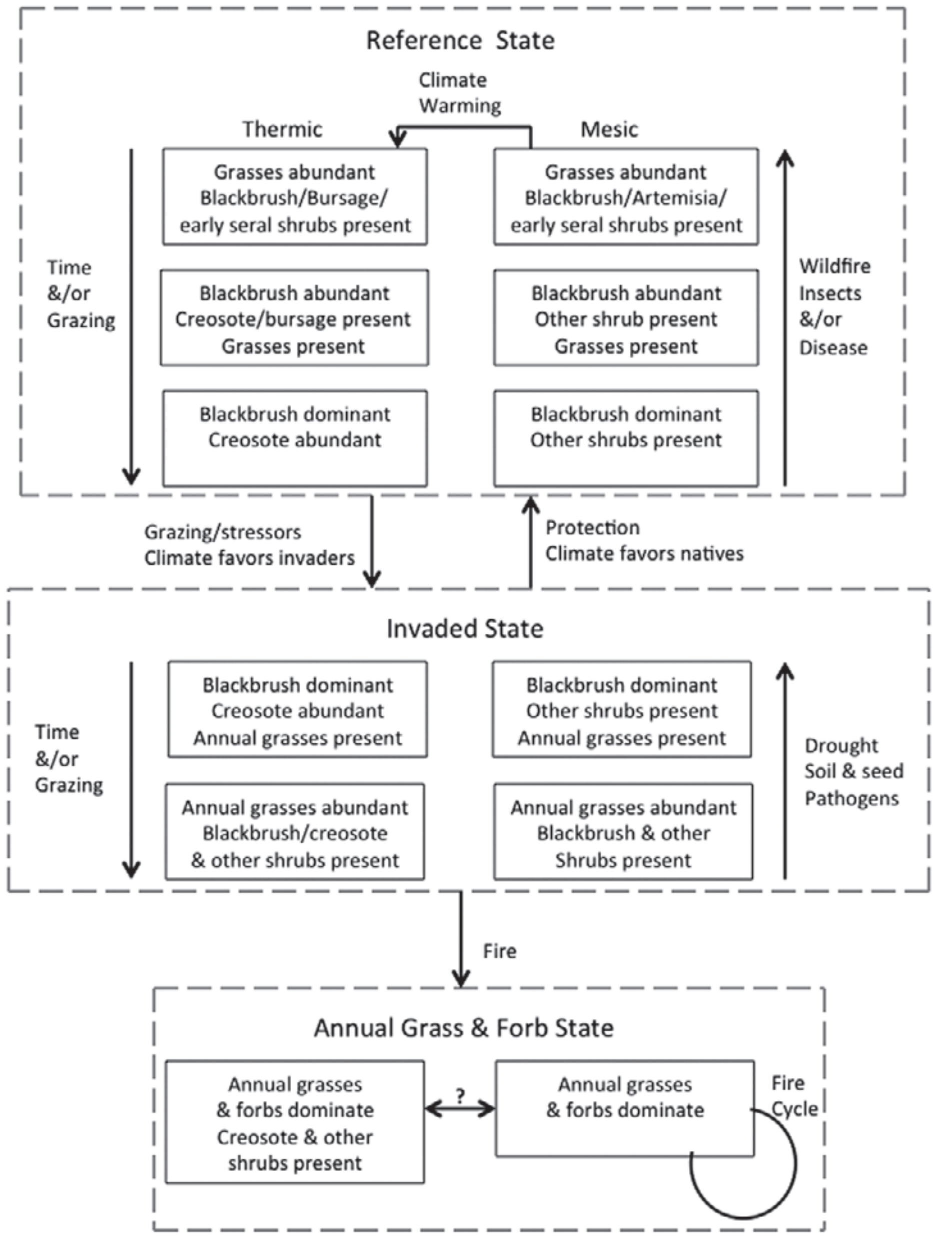

Figure 7.7—Hypothetical state and transition model for the thermic and mesic blackbrush ecological site types that considers potential transitions with a warming climate. The thermic state is more arid and has a minor component of creosotebush and bursage. For both reference states, disturbances that reduce blackbrush increases grass abundance; time and/or grazing increases shrub abundance. Grazing, stressors and climate conditions that favor invaders can result in transition to an invaded state. Time and or grazing results in greater abundance of annual grasses; drought and soil and seed pathogens that target annual grasses result in higher shrub abundance. Return to the reference state requires protection and climate conditions that favor natives. Fire can convert the invaded state to an annual grass and forb state and result in repeated fire. A warming climate may disfavor annual invaders and favor a creosote dominated state.

Table 7.3—General guidelines for conducting a restoration project in southern Nevada (modified from Miller and others 2007; Pyke 2011; Tausch and others 2009).

Steps in the process	Questions to be addressed
I. Identify landscape priorities and ecological sites	1. Where are priority sites for protection, prevention and restoration? Consider the landscape context. 2. What are the topographic characteristics and soils of the site? Verify soils mapped to the location and collect information on soil texture, depth and basic chemistry (pH, calcium carbonate, etc.) 3. How will topographic characteristics and soils affect vegetation establishment and erosion? Evaluate erosion risk based on topography and soil characteristics. 4. What is the potential native plant community for the site? Match soil components to their correlated ESD. This provides a list of potential species for the site.
II. Determine current state of the site	5. Is the site still within the reference state of the state and transition model for this ecological site?
III. Select appropriate action	6. How far does the site deviate from the reference state? 7. Do sufficient perennial shrubs and herbaceous species exist to facilitate recovery? No action is needed. 8. Are invasive species a minor component? Protection or preventative management may prevent conversion. 9. Do invasive species dominate the site while native life forms are missing or severely under represented? Active restoration is required to restore habitat. 10. Are species from drier or warmer ecological sites present? Restoration with species from the drier or warmer site should be considered. 11. Have soils or other aspects of the physical environment been altered? The site may have crossed a threshold and represent a new ecological site type requiring new site-specific restoration approaches.
IV. Determine post-treatment	12. How long should the site be protected before management land uses begin? In general, sites with lower resilience and resistance should be protected for longer periods. 13. How will monitoring be performed? Restoration effectiveness monitoring includes a complete set of measurements, analyses, and a report. 14. Are adjustments to the restoration approach needed? Adaptive management is applied to future projects by compiling information based on consistent findings from multiple locations.

landscape priorities and ecological sites, determining the current state of the site, selecting the appropriate action(s), and determining post-treatment management. A general approach that asks questions to identify the information required in each step is provided in table 7.3. These questions can be modified to include the specific information needed for restoration of different ecosystem types.

Monitoring Activities

Monitoring programs designed to track ecosystem changes in response to both stressors and management actions can be used to increase understanding of ecosystem resilience and resistance and to realign restoration and management approaches and implement adaptive management (Chapter 1). Information is increasing on likely changes in southern Nevada ecosystems with additional stress and climate warming, but a large degree of uncertainty still exits. Strategic placement of monitoring sites and repeated measurements of key abiotic (precipitation, temperature, evaporation) and biotic (dominant native and exotic species) variables and ecological conditions can be used to decrease uncertainty and increase the effectiveness of management decisions. Monitoring also can be used to track changes in regional and local stressors over time like the level of nitrogen deposition and the intensity of wild horse and burro grazing. Monitoring sites should span the environmental/productivity gradients and ecosystem types that occur in southern Nevada. In addition, the following areas of high priority should be monitored:

1. ecosystem types of small extent under development pressure like mesquite/catclaw and salt desert shrub;
2. ecosystems that support numerous species of conservation concern like springs and riparian areas;
3. ecotones between ecosystem types where changes in response to climate are expected to be largest (Loehle 2000; Stohlgren and others 2000);
4. ecological sites with different climatic conditions and soils that are exhibiting invasion and repeated fires; and
5. ecological sites with different climatic conditions and soils that are exhibiting tree expansion and increased fire risk.

Monitoring the response of ecosystems to management actions and active restoration also is of high priority as it provides information on treatment effectiveness that can be used to adjust methodologies.

Monitoring activities are most beneficial when consistent approaches are used among and within agencies to collect, analyze, and report monitoring data. Common databases can be used by agency partners to record and share monitoring data, like the Land Treatment Digital Library (USGS 2010), to facilitate this process. A restoration geodatabase was recently developed by southern Nevada agencies to facilitate the collection and sharing of data from each agency. This geodatabase includes an inventory and assessment of upland ecological disturbances, recommended restoration methods, documentation of restoration treatments, and monitoring of restored sites. Once this database is implemented, analysis of the data will help to identify trends and large-scale problems that can be addressed by the interagency restoration team, as well as to evaluate effectiveness of restoration treatments, determine the best techniques, and make adjustments on the ground as needed.

Protocols have been developed to guide field staff in data collection and decision making for the restoration of road disturbances (DeFalco and Scoles-Sciulla 2011). According to these protocols, implementation monitoring should be conducted for up to 2 years following restoration to evaluate effectiveness of restoration treatments, and ecological monitoring should occur approximately every 5 years to determine how ecosystem functions are recovering. Monitoring includes photo documentation and measuring plants (including non-natives), erosion features, soils (compaction, stability), and biological soil crusts.

References

Abella, S.R.; Engel, E.C.; Lund, C.L.; Spencer, J.E. 2009. Early post-fire plant establishment on a Mojave Desert burn. Madrono. 56:137-148.

Abella, S.R.; Hurja, J.C. ; Merkler, D.J. ; Denton, C.W. ; Brewer, D.G. 2011. Overstory-understory relationships along forest type and environmental gradients in the Spring Mountains of southern Nevada, USA. Folia Geobotanica. doi. a0.1007/sl2224-011-9108-7.

Abella, S.R; Newton, A.C. 2009. A systematic review of species performance and treatment effectiveness for revegetation in the Mojave Desert, USA. In: Fernandez-Bernal, A.; De La Rosa, M. A. Arid environments and wind erosion. Hauppauge, NY: Nova Science Publishers, Inc: 45-74.

Allen, C.D.; Savage, M.; Falk, D.A.; Suckling, K.F.; Swetnam, T.W.; Shulke, T.; Stacey, P.B.; Morgan, P.; Hoffman, M.; Klinger, J.T. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecological Applications. 12:1418-1433.

Allen, C.D.; Macalady, A.; Chenchouni, H.; Bachelet, D. and others. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management. 259:660–684.

Allen, C.R.; Gunderson, L; Johnson, A.R. 2005. The use of discontinuities and functional groups to assess relative resilience in complex systems. Ecosystems. 8:958-966.

Anderson, Michelle D. 2001. *Coleogyne ramosissima*. In: Fire Effects Information System. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Online: http://www.fs.fed.us/database/feis/ [2011, July 27].

Anderson, B.W.; Russell, P.E.; Ohmart, R.D. 2004. Riparian revegetation: an account of two decades of experience in the arid southwest. Blythe, CA: Avvar Books. 268 p.

Auger, J. 2005. Effects of resource availability and food preferences on population dynamics and behavior of Ord's kangaroo rats (*Dipodomys ordii*). Reno, NV: University of Nevada Press. 150 p. Dissertation.

Bainbridge, D.A. 2007. A guide for desert and dryland restoration. Washington, DC: Island Press. 391 p.

Bauer, J.M.; Weisberg, P.M. 2009. Fire history in a central Nevada pinyon-juniper woodland. Canadian Journal of Forest Research. 39:1589-1599.

Bestelmeyer, B.T; Tugel, A.J.; Peacock, G.L., Jr.; Robinett, D.G.; Shaver, P.L.; Brown, J.L.; Herrick, J.E.; Sanchez, H.; Havstad, K.M. 2009. State-and-transition models for heterogeneous landscapes: a strategy for development and application. Range Ecology and Management. 62:1-15.

Bestelmeyer, B.T.; Goolsby, D.P.; Archer, S.R. 2011. Spatial perspectives in state-and-transition models: a missing link to land management? Journal of Applied Ecology. 48: 746-757.

Biondi, F.; Jamieson, L.P.; Strachan, S.; Sibold, J. 2011. Dendrochronology testing of the pyroclimatic hypothesis in the central Great Basin, Nevada, USA. Ecosphere. 2:1-20.

Blodgett, J.T.; Sullivan, K.F. 2004. First report of white pine blister rust on Rocky Mountain bristlecone pine. Plant Disease. 88: 311.

Bowns, J.E. 1973. An autecological study of blackbrush (*Coleogyne ramosissima* Torr.) in southwestern Utah. Logan, UT: Utah State University. 115 p. Dissertation.

Bradley, B.A. 2010. Assessing ecosystem threats from global and regional change: hierarchical modeling of risk to sagebrush ecosystems from climate change, land use and invasive species in Nevada, USA. Ecography. 33:198-208.

Bradley, B.A.; Oppenheimer, M.; Wilcove, D.S. 2009. Climate change and plant invasions: restoration opportunities ahead? Global Change Biology. 15:1511-1521.

Breshears, D.D.; Cobb, N.S.; Rich, P.M.; Price, K.P.; Allen, C.D.; Balice, R.G.; Romme, W.H.; Kastens, J.H.; Floyd, M.L.; Belnap, J.; Anderson, J.J.; Myers, O.B.; Meyer. C.W. 2005. Regional vegetation die-off in response to global-change-type drought. Proceedings of the National Academy of Science. 102:15144-15148.

Briggs, M. K. 1996. Riparian ecosystem restoration in arid lands: strategies and references. Tucson, AZ: University of Arizona Press. 159 p.

Brooks. M.L. 2011. Effects of high fire frequency in creosotebush scrub vegetation of the Mojave Desert. International Journal of Wildland Fire. Online: http://dx.doi.org/10.1071/WF10140.

Brooks, M.L.; Chambers, J.C. 2011. Resistance to invasion and resilience to fire in desert shrublands of North America. Rangeland Ecology and Management. 64:431-438.

Brooks, M.L.; Esque, T.C.; Duck, T. 2007. Chapter 6: Creosotebush, blackbrush, and interior chaparral shrublands. In: Hood, S.M.; Miller, M. (eds.). Fire ecology and management of the major ecosystems of southern Utah. Gen. Tech. Rep. RMRS-GTR-202. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 97-110.

Brooks, M.L.; Matchett, J.R. 2006. Spatial and temporal patterns of wildfire in the Mojave Desert, 1980-2004. Journal of Arid Environments. 67:148-164.

Brooks, M.L.; Minnich, R.A. 2006. Fire in the southeastern Desert Bioregion. In: Sugihara, M.G.; van Wagendonk, J.W.; Fites-Kaufman, J.; Shaffer, K.E.; Thode, A.E. (eds.). Fire in California ecosystems. Berkeley, CA: University of California Press: 391-414.

Brown, James K.; Smith, Jane Kapler. 2000. Wildland fire in ecosystems: effects of fire on flora. Gen. Tech. Rep. RMRS-GTR-42 Vol. 2. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 257 p.

Brown, P.; Wu, R. 2005. Climate and disturbance forcing of episodic tree recruitment in a southwestern ponderosa pine landscape. Ecology. 86:3030-3038.

Chambers, J.C. 1995. Disturbance, life history strategies, and seed fates in alpine herbfield communities. American Journal of Botany. 82:421-433.

Chambers, J.C. 1997. Restoration of alpine ecosystems in the United States. In: Urbanska, K.; Webb, N. (eds.). Restoration ecology and sustainable development. Cambridge, UK: Cambridge University Press: 161-187.

Chambers, J.C.; Roundy, B.A.; Blank, R.R.; Meyer, S.E; Whittaker, A. 2007. What makes Great Basin sagebrush ecosystems invasible by *Bromus tectorum*? Ecological Monographs. 77:117-145.

Chapin, F.S., III.; Torn, S.E.; Tateno, M. 1996. Principles of ecosystem sustainability. The American Naturalist. 148:1016-1037.

Christensen, N.L.; Bartuska, A.M.; Brown, J.H.; Carpenter, S.; D'Antonio, C.; Francis, R.; Franklin, J.F.; MacMahon, J.A.; Noss, R.F.; Parsons, D.J.; Peterson, C.H.; Turner, M.G.; Woodmansee, R.G. 1996. The report of the Ecological Society of America Committee on the scientific basis for ecosystem management. Ecological Applications. 6:665-691.

Clokey, I.W. 1951. Flora of the Charleston Mountains, Clark County, Nevada. Berkeley, CA: University of California Press. 274 p.

Cole, K.L.; Ironside, K.; Eischeid, J.; Garfin, G.; Duffy, P.B.; Toney, C. 2011. Past and ongoing shifts in Joshua tree distribution support future modeled range contraction. Ecological Applications. 21:137–149.

Condon, L; Weisberg, P.L.; Chambers, J.C. 2011. Abiotic and biotic influences on *Bromus tectorum* invasion and *Artemisia tridentata* recovery after fire. International Journal of Wildland Fire. 20:1-8.

D'Antonio, C.; Chambers, J.C. 2006. Using ecological theory to manage or restore ecosystems affected by invasive plant species. In: Falk, D.; Palmer, M.; Zedler, J. (eds.). Foundations of Restoration Ecology. Covelo, CA: Island Press: 260-279.

D'Antonio, C.M.; Chambers, J.C.; Loh, R.; Tunison, J.T. 2009. Applying ecological concepts to the management of widespread grass invasions. In: Inderjit, R.L. (ed.). Ecological invasions and restoration. Netherlands: Springer: 123-149.

D'Antonio, C.M.; Thomsen, M. 2004. Ecological resistance in theory and practice. Weed Technology. 18:1572-1577.

Defalco, L.A.; Scoles-Sciulla, S.J. 2011. Protocol for documenting disturbances, prioritizing restoration, and evaluating restoration effectiveness for vehicle disturbances in Mojave Desert uplands. Unpublished report compiled for Southern Nevada Restoration Team partners, Las Vegas, NV.

Desert Research Institute. 2008. Task 7: First iteration conceptual models for 11 MSHCP ecosystems. Annual Report. Project 2005-DRI-574A-P. Reno, NV, Desert Research Institute.

Engel, C.E.; Abella, S.R. 2011. Vegetation recovery in a desert landscape after wildfires: influence of community type, time since fire and contingency effects. Journal of Applied Ecology. 48:1401-1410.

Fleishmann, E.; Murphy, D.D.; Sada, D.W. 2006. Effects of environmental heterogeneity and disturbance on the native and non-native flora of desert springs. Biological Invasions. 8:1091-1101.

Floyd, M.L.; Hanna, D.D.; Romme, W.H. 2004. Historical and recent fire regimens in piñon-juniper woodlands on Mesa Verde, Colorado, USA. Forest Ecology and Management. 198:269-289.

Fule, P.Z.; Laughlin, D.C.; Covington, W.W. 2005. Pine-oak forest dynamics five years after ecological restoration treatments, Arizona, USA. Forest Ecology and Management. 218:129-145.

Grace, J.; Berninger, F.; Nagy, L. 2002. Impacts of climate change on tree line. Annals of Botany. 90:537-544.

Harris, J.A.; Hobbs, R.J.; Higgs, E.; Aronson. J. 2006. Ecological restoration and global climate change. Restoration Ecology. 14:170-176.

Hobbs, R. J.; Harris, J.A. 2001. Restoration ecology: repairing the earth's damaged ecosystems in the new millennium. Restoration Ecology. 9:239-246.

Holling, C.S. 1973. Resilience and stability in ecological systems. Annual Review of Ecology and Systematics. 4:1-23.

Holtmeier, F.; Broll, G. 2007. Treeline advance: driving processes and adverse factors. Landscape. 1: 1-33. doi: 10.3097/LO.200701.

Jamieson, L.P. 2008. Fire history of a pinyon/juniper-ponderosa pine ecosystem in the Intermountain West. Reno, NV: University of Nevada. 86 p. Thesis.

Jones, L.C. 2011. Ecology and restoration of blackbrush (*Coleogyne ramoissima*) in the Mojave Desert. Abstracts of the 96th annual meeting of the Ecological Society of America; August 7-12, 2011; Austin, TX. Online: http://esameetings.allenpress.com/2011/webprogram/Paper32306.html. [2012, Oct 29]

Kelly, A.E.; Goulden, M.L. 2008. Rapid shifts in plant distribution with recent climate change. Proceedings of the National Academy of Sciences. 105:11823-11826.

Kilpatrick, M. 2009. A tree-ring based fire history of the Clover Mountains, Lincoln County, Nevada. Reno, NV: University of Nevada. 108 p. Thesis.

Kitchen, S.G. 2010. Historic fire regimes of eastern Great Basin (USA) mountains reconstructed from tree rings. Provo, UT: Brigham Young University. 166 p. Dissertation.

Kliejunas, J.; Adams, D. 2003. White pine blister rust in California. Tree Notes No. 27. California Department Of Forestry and Fire Protection. 4 p. Online: http://ceres.ca.gov/foreststeward/pdf/treenote27.pdf. [2012, Oct 29]

Knochenmus, L.; Wilson, J.; Laczniak, R.; Sweetkind, D.; Thomas, J.; Justet, L.; Hershey, R. 2007. Ground-water conditions. In: Welch, A.; Bright, D. (eds.). Water resources of the Basin and Range carbonate aquifer system in White Pine County, Nevada, and adjacent areas in Nevada and Utah. Open-File Report 2007-1156. U.S. Department of the Interior, U.S. Geological Survey: 58-61.

Laughlin, D.C.; Bakker, J.D.; Stoddard, M.T.; Daniels, M.L.; Springer, J.D.; Gildar, C.N.; Green, A.M.; Covington, W.W. 2004. Toward reference conditions: wildfire effects on flora in an old-growth ponderosa pine forest. Forest Ecology and Management. 199:137-152.

Littell, J.S.; McKenzie, D; Peterson, D.L.; Westerling, A.L. 2009. Climate and wildfire area burned in the western U.S. ecoprovinces, 1916-2003. Ecological Applications. 19:1003-1021.

Loarie, S.R.; Duffy, P.B.; Hamilton, H.; Asner, G.P.; Field, C.B.; Ackerly, D.D. 2009. The velocity of climate change. Nature. 462:1052-1055.

Loehle, C. 2000. Forest ecotone response to climate change: sensitivity to temperature response functional forms. Canadian Journal of Forest Research. 30:1362-1645.

Meinke, C.W.; Knick, S.T.; Pyke, D.A. 2009. A spatial model to prioritize sagebrush landscapes in the Intermountain West (U.S.A.) for restoration. Restoration Ecology. 17:652-659.

Meyer, S.E.; Pendleton, B.K. 2005. Factors affecting seed germination and seedling establishment of a long-lived desert shrub (*Coleogyne ramosissima*: Rosaceae). Plant Ecology. 178:171-187.

Mifflin, M.D. 1968. Delineation of ground-water flow systems in Nevada. Technical Report Series H-W: Hydrology and water resources publication No. 4. Reno, NV: Desert Research Institute, University of Nevada System. 103 p.

Miller, R.F.; Bates, J.D.; Svejcar, T.J.; Pierson, F.B.; Eddleman, L.E. 2005. Biology, ecology, and management of western juniper. Tech. Bull. 152. Corvallis, OR: Oregon State University, Agricultural Experiment Station. 77 p.

Miller, R.F.; Bates, J.D.; Svejcar, T.J.; Pierson, F.B.; Eddleman, L.E. 2007. Western juniper field guide: asking the right questions to select appropriate management actions. Circular 1321. Reston, VA: U.S. Department of the Interior, Geological Survey.

Miller, Richard F.; Tausch, Robin J.; McArthur, E. Durant; Johnson, Dustin D.; Sanderson, Stewart C. 2008. Age structure and expansion of piñon-juniper woodlands: a regional perspective in the Intermountain West. Res. Pap. RMRS-RP-69. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 15 p.

Monsen, Stephen B.; Stevens, Richard; Shaw, Nancy L., comps. 2004. Restoring western ranges and wildlands. Gen. Tech. Rep. RMRS-GTR-1136-vol-1, 2, and 3. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 294 + index.

Moore, M.M.; Casey, C.A.; Bakker, J.D.; Springer, J.D.; Fule, P.Z.; Covington, W.W.; Laughlin, D.C. 2006. Herbaceous vegetation responses (1992-2004) to restoration treatments in a ponderosa pine forest. Rangeland Ecology and Management. 59:135-144.

Murphy, D.D.; Weiss, S.B. 1992. The effects of climate change on biological diversity in western North America: species losses and mechanisms. In: Peters, R.; Lovejoy, T. (eds.). Global warming and biodiversity. New Haven, CT: Yale University Press: 355-368.

Nielson, R.P.; Lenihan, J.M.; Bachelet, D.; Drapek, R.J. 2005. Climate change implications for sagebrush ecosystems. Transactions, 70th North American wildlife and natural resources conference; March 16-19, 2005. Arlington, VA: Wildlife Management Institution Publications Department: 145-159.

Pase, C.P.; Brown, D.E. 1982. Rocky Mountain (petran) and madrean montane coniferous forests. In: Brown, D.E., (ed.). Biotic communities of the American Southwest-United States and Mexico. Desert Plants. 4(1/4):43-48.

Pendleton, B.K. 2008. *Coleogyne ramosissima* Torr. In: Bonner, F.T.; Karrfalt, R.P. (eds.). The woody plant seed manual. Agriculture handbook 727. Washington, DC: U.S. Department of Agriculture, Forest Service: 422-425.

Pendleton, B.K.; Meyer, S.E. 2004. Habitat-correlated variation in blackbrush (*Coleogyne ramosissima*: Rosaceae) seed germination response. Journal of Arid Environments. 59:229-243.

Pendleton, R.L.; Pendleton, B.K.; Howard, G.L. 1999. Effects of microbiotic soil crust organisms and mycorrhizal fungi on seedling growth of blackbrush (*Coleogyne ramosissima*). CERL Tech. Rep. 99/108. U.S. Army Corps of Engineers, Engineer Research and Development Center. 26 p.

Pendleton, R.L.; Pendleton, B.K.; Meyer, S.E.; Carlson, S.; Morrison, E. 2012. Viability of blackbrush (*Coleogyne ramosissima* Torr. [Rosaceae]) seed following long-term storage. Native Plant Journal. 13:5-13.

Pockman, W.T.; Sperry, J. S. 1997. Freezing induced xylem cavitation and the northern limit of *Larrea tridentata*. Oecologia. 109:19-27.

Pyke, D.A. 2011. Restoring and rehabilitating sagebrush habitats. In: Knick, S.T.; Connelly, J.W. (eds.). Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology. Berkeley, CA: University of California Press: 531-548.

Raffa, K.F.; Aukema, B.H.; Bentz, B. J.; Carroll, A.L.; Hicke, J.A.; Turner, M.G.; Romme, W.H. 2008. Cross-scale drivers of natural disturbances prone to anthropogenic amplification: the dynamics of bark beetle eruptions. Bioscience. 58:501-517.

RECON (Regional Environmental Consultants, Inc.). 2000. Final Clark County Nevada multiple species habitat conservation plan and environmental impact statement. Prepared for Clark County Administrative Services. 550 p. plus appendices.

Rehfeldt, G.E.; Crookston, N.L.; Warwell, M.V.; Evans, J.S. 2006. Empirical analyses of plant-climate relationships for the western United States. International Journal of Plant Science. 167:1123-1150.

Romme, W.H.; Allen, C.D.; Bailey, J.D.; Baker, W.L.; Bestelmeyer, B.T.; Brown, P.M.; Eisenhardt, K.S.; Floyd, M.L.; Huffman, D.W.; Jacobs, B.F.; Miller, R.F.; Muldavin, E.H.; Swetnam, T.W.; Tausch, R.J.; Weisberg, P.J. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon-juniper vegetation of the western United States. Rangeland Ecology and Management. 62:203-222.

Sada, D.W.; Herbst, D.B. 1999. Habitat use by rare aquatic macroinvertebrates in spring brooks of the upper Muddy River, Nevada. Unpublished report to The Nature Conservancy, Southern Nevada Projects Office, Las Vegas, NV.

Sada, D.W.; Herbst, D.B. 2006. Ecology of aquatic macroinvertebrates in Travertine and Nevares Springs, Death Valley National Park, California, with an examination of diversion effects on their abundance and community structure. Unpublished report to U.S. National Park Service, Death Valley National Park.

Sada, D.W.; Fleishman, E.; Murphy, D.D. 2005. Associations among spring-dependent aquatic assemblages and environmental and land use gradients in a Mojave Desert mountain range. Diversity and Distributions. 11:91-99.

Sada, D.W.; Herbst, D.B.; Smith, F.J. 2000. A survey of springs on Ruby Lake National Wildlife Refuge (Elko and White Pine Counties, Nevada) and the influence of habitat alteration on the structure of aquatic invertebrate communities. Unpublished report to U.S. Fish and Wildlife Service, Ruby Lake National Wildlife Refuge, NV.

Sada, D.W.; Nachlinger, J.L. 1996. Spring Mountains Ecosystem: Vulnerability of spring-fed aquatic and riparian systems to biodiversity loss. Unpublished Report to the U.S. Fish and Wildlife Service, Las Vegas, NV.

Sada, D.W.; Nachlinger, J.L. 1998. Spring Mountains ecosystem: vulnerability of spring-fed aquatic and riparian systems to biodiversity loss: Part II, Springs Surveyed in 1997. Unpublished Report to the U.S. Bureau of Land Management, Las Vegas, NV.

Sada, D.W.; Williams, J.E.; Silvey, J.C.; Halford, A.; Ramakka, J.; Summers, P.; Lewis, L. 2001. Riparian area management: a guide to managing, restoring, and conserving springs in the western United States. Tech. Ref. 1737-17. Denver, CO: Department of the Interior, Bureau of Land Management. BLM/ST/ST-01/001+1737.

Salzera, M.W.; Hughes, M.K.; Bunn, A.G.; Kipfmueller, K.F. 2009. Recent unprecedented tree-ring growth in bristlecone pine at the highest elevations and possible causes. Proceedings of the National Academy of Science. 106:20,348–20,353.

Scoppettone, G.G.; Rissler, P.H.; Gourley, C.; Martinez, C. 2005. Habitat restoration as a means of controlling non-native fish in a Mojave Desert oasis. Restoration Ecology. 13:247-256.

Shafroth, P.B; Cleverly, J.R.; Dudley, T.L.; Taylor, J.P.; Van Riper III, C.; Weeks, E.R.; Stuart, J.N. 2005. Control of *Tamarix* in the Western United States: implications for water Salvage, wildlife use, and riparian restoration. Environmental Management. 35:231-246.

Shanahan, S.A.; Nelson, S.M.; Van Dooremolen, D.M.; Eckberg, J.R. 2011. Restoring habitat for riparian birds in the lower Colorado River watershed: an example from Las Vegas, Wash, NV. Journal of Arid Environments. 75:1182-1190.

Society for Ecological Restoration International Science and Policy Working Group [SERI]. 2004. The SERI Primer on Ecological Restoration. Version 2. Online: www.ser.org and www.ser.org/pdf/primer3.pdf . [2012, April 9].

Stohlgren, T.J.; Owen, A.J.; Lee, M. 2000. Monitoring shifts in plant diversity in response to climate change: a method for landscapes. Biodiversity and Conservation. 9:165-186.

Stringham, T.K.; Krueger, W.C.; Shaver, P.L. 2003. State and transition modeling: an ecological process approach. Journal of Range Management. 56:106–113.

Swetnam, R.W.; Betancourt, J.L. 1998. Mesoscale disturbance and ecological response to decadal climate variability in the American southwest. Journal of Climate. 11:3128-3147.

Tausch, R.J.; Miller, R.R.; Roundy, B.A.; Chambers, J.C. 2009. Piñon and juniper field guide: asking the right questions to select appropriate management actions. U.S. Geological Survey Circular 1335. 94 p. Online: http://pubs.usgs.gov/circ/1335/. [2012, April 9].

Turner, K.; LaVoie, A.M.; Ronning, C.J.; Sharp, R.M;, Palmer, C.J.; Miller, J.M. 2009. SNAP Science and Research Strategy. Southern Nevada Agency Partnership. Online: http://snap.gov/upload/SNAP-S-R-Strategy-2009r.pdf. [2011, April 6].

Tylianakis, J.M.; Didham, R.K.; Bascompte, J.; Wardel, D.A. 2008. Global change and species interactions in terrestrial ecosystems. Ecology Letters. 11:1351-1263.

Urbanska, K.M.; Chambers, J.C. 2002. High-elevation ecosystems. In: Perrow, M.R; Davies, A.J. (eds.). Handbook of restoration ecology. Cambridge, UK: Cambridge University Press. 2: 376-400.

U.S. Army Corp of Engineers. 2008. Virgin River Watershed – Utah, Arizona and Nevada. Comprehensive watershed analysis. Final Report. August, 2008. Online: http://planning.usace.army.mil/toolbox/library/misc/virginriverwatershed.pdf. [2012, Aug 15]

USDA Natural Resources Conservation Service. 2012. Web soil survey. Online: http://websoilsurvey.nrcs.usda.gov/app/WebSoilSurvey.aspx. [2012, April 9].

U.S. Geological Survey (USGS). 2010. Land Treatment Digital Database. Online: http://greatbasin.wr.usgs.gov/ltdl/Default.aspx. [2012, April 9].

Van de Ven, C.M.; Weiss, S.B.; Ernst, W.G. 2007. Plant species distributions under present conditions and forecasted for warmer climates in an arid mountain range. Earth Interactions. 11:1-33.

Walker B.H; Kinzig A.; Langridge J. 1999. Plant attribute diversity, resilience, and ecosystem function: the nature and significance of dominant and minor species. Ecosystems. 2:1–20.

Webb, R.H; Bowers, J.E. 1993. Changes in frost frequency and desert vegetation assemblages in Grand Canyon, Arizona. In: Redmond, K.T.; Tharps, V.L. (eds.). Proceedings of the ninth annual Pacific Climate (PACLIM) workshop; April 21-24; 1992. Tech. Rep. No. 3. California Department of Water Resources, Interagency Ecological Studies Program: 71-82.

Webb, R.H.; Defalco, L.A.; Esque, T.C.; Medica, P.A. 2009. A review of selected long-term ecological studies of the Mojave Desert ecosystem. In: Webb, R.H.; Fenstermaker, L.F.; Heaton, J.S.; Hughson, D.L.; McDonald, E.V.; Miller, D.M. (eds.). The Mojave Desert. Las Vegas, NV: University of Nevada Press: 429-456.

Weigand, J.; Rodgers, J. 2009. Active restoration for the Mojave Desert. In: Webb, R.H.; Fenstermaker, L.F.; Heaton, J.S.; Hughson, D.L.; McDonald, E.V.; Miller, D.M. (eds.). The Mojave Desert. Las Vegas, NV: University of Nevada Press: 378-409.

Whisenant, S.G. 1999. Repairing damaged wildlands – a process-oriented, landscape-scale approach. Cambridge, UK: Cambridge University Press. 312 p.

Williams, J.J.; Jackson, S.T. 2007. Novel climates, no-analog communities, and ecological surprises. Frontiers in Ecology and the Environment. 5:475-482.

Winkel,V.K.; Ostler, W.K.; Gabbert, W.D.; Lyon, G.L. 1995. The effects of seedbed preparation, irrigation and water harvesting on seedling emergence at the Nevada Test Site. In: Roundy, B.A.; MacArthur, E.D.; Haley, J.S.; Mann, D.K., (comps.). 1995. Proceedings, wildland shrub and arid land restoration symposium; October 19-21; Las Vegas, NV. Gen. Tech. Rep. INT-GTR-315. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station: 135-141.

Wisdom, M.J.; Rowland, M.M.; Suring, L.H. (eds.). 2005. Habitat threats in the sagebrush ecosystem: methods of regional assessment and applications in the Great Basin. Lawrence, KS: Alliance Communications Group, Allen Press. 301 p.

Zitzer, S.F.; 2009. Recruitment of native and non-native annual and perennial plant species following wildfire in a blackbrush (*Coleogyne ramosisima*) community in south central Nevada. Poster Session 67-113. Abstracts of the 94th annual meeting of the Ecological Society of America. August 2-7, 2009. Albuquerque, NM. Online: http://eco.confex.com/eco/2009/techprogram/P19214.HTM. [2012, July 22].