Chapter 3.2. Rangeland in Northeastern California

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Introduction

Estimates of rangeland in the United States vary widely depending on the definition used, but, in general about one-third of the area of the United States (511 to 662 million acres [207 to 268 million ha] out of 2 billion acres [800 million ha] in the coterminous United States) is considered rangeland (Reeves and Mitchell 2011; USDA 2016; USDOI 2013). Rangeland should not be confused with grazing land that includes rangeland, pastureland, forestland, or any other land with potential for providing forage for wild or domestic ungulates (Society for Range Management 1998, and see Chapter 3.1, Warren, this synthesis, *Perceptions and History of Rangeland*). The U.S. Department of the Interior manages grazing on about 200 million acres (81 million ha), of which its Bureau of Land Management (BLM) oversees grazing on about 155 million acres (63 million ha) (USDOI 2013), although only about 135 million acres (55 million ha) of the BLM-managed ground are considered rangelands (Reeves and Mitchell 2012). Similarly, although the U.S. Department of Agriculture, Forest Service manages grazing on 95 million acres (38 million ha) (USDA 2016), it only manages about 50 million acres (20 million ha) of rangeland (Reeves and Mitchell 2011).

Forest Service rangeland is managed to sustain its health, diversity, and productivity and to meet society's needs now and in the future (USDA 2015; see USDA 2012 for an overview of the Nation's rangeland status). Rangeland productivity is often associated with forage for livestock. In the 17 Western States, BLM-managed forage accounted for about 1.6 percent of the total livestock receipts (USDOI 2013). Although the number of domestic animals permitted to graze on public lands continues its 100-year decline (USDOI 2013), grazing remains an important employment sector in the Sierra Nevada (Stewart 1996) and plays a role in sustaining viable ranching operations that maintain large tracts of open space (Huntsinger et al. 2010).

Rangeland constitutes a large proportion of the Lassen and Modoc National Forests (hereafter the Lassen and the Modoc). Of the Modoc's 1.6 million acres (647,000 ha), about 1 million (405,000 ha) is considered rangeland, of which 90 percent is suitable for livestock grazing (USDA and USDOI 2007). Of that, about 20 percent (320,000 acres [130,000 ha]) is covered with sagebrush (*Artemisia* spp.) ecosystem (USDA 1991). On the Lassen, about 410,000 acres (166,000 ha) of the forest's 1.2 million acres (486,000 ha) is suitable for grazing. Rangelands include about 27,500 acres (11,130 ha) of sagebrush, 20,000 acres (8100 ha) of antelope bitterbrush (*Purshia tridentata*), and 16,000 acres (6500 ha) grasslands (USDA 1992). Native ungulates on this rangeland include mule deer (*Odocoileus hemionus*) and pronghorn (*Antilocapra americana*). Historically, bison (*Bison bison*) may have also grazed in Northeastern California (Merriam 1926 cited in Norman and Taylor 2005). Both forests have a long tradition of cattle and sheep grazing (see Brown 1945 and Norman and Taylor 2005) that continues through grazing allotments administered through a permit system (fig. 3.2.1). Wild horses also graze these rangelands. Recent, successful breeding packs of gray wolf (*Canis lupus*) in Siskiyou and Lassen Counties suggest these carnivores may become a factor in the management of native ungulates, livestock, and wild horses (see Chapter 4.1, Hanberry and Dumroese, this synthesis, *Biodiversity and Representative Species in Dry Pine Forests*).

Grazing in the West, in California, and on the Lassen and the Modoc, is an important topic because changes in grazing management can have profound effects on plant community composition, fire occurrence and effects, and ranch income and market value (Lewandrowski and Ingram 2002), *The Science Synthesis to Support Socioecological Resilience in the Sierra Nevada and Southern Cascade Range* (hereafter, Sierra Nevada Science Synthesis) published by Long et al. (2014) presented significant discussion about grazing. In particular is Chapter 6.3—*Wet Meadows* (Long and Pope 2014). Despite that seemingly narrow title, the authors note that grazing "involves a complex interplay of social and ecological factors." They also provide a broad overview of grazing strategies applicable to diverse landscapes that includes the socioecological perspective and identify opportunities where disturbance through grazing can potentially help land managers meet desired ecological

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Figure 3.2.1—A long tradition of grazing cattle and sheep on the Lassen and Modoc continues through a permit system that ensures proper management of public land (photo by Ken Sandusky, Forest Service).

conditions. In addition, they discuss grazing management in riparian areas and grazing effects on water quality, soils, nutrients, vegetation, amphibians, and bumblebees. Causes and restoration of channel incision are also presented. Information on monitoring and evaluating management is provided. The authors also mention the deficiencies of published grazing research and conclude with discussion about research needs.

Chapter 9.5 (*Managing Forest Products for Community Benefit*) in the Sierra Nevada Science Synthesis provides a broad and in-depth discussion about balancing ecological goals with local community well-being (Charnley and Long 2014), and Section 5 (Society) in this *Northeastern California Plateaus Bioregion Science Synthesis* (particularly Chapter 5.4, Flores and Stone, this synthesis, *Community Engagement in the Decisionmaking Process for Public Land Management in Northeastern California*) focuses on public input to decision making and in solving conflict associated with resource management. This is critical because management occurs within the social context, no single approach will be universally satisfactory, and often societal needs supersede biophysical factors (see Stanturf et al. 2014). Charnley and Long (2014) also discuss how public land grazing supports conservation of private rangelands by ensuring that a critical baseline of resources is available to the industry.

While most of the grazing concepts and concerns included in the Sierra Nevada Science Synthesis have direct applicability to dry pine forestland and sagebrush rangeland, this synthesis builds upon that effort by focusing more explicitly on the effects of grazing within the Lassen and Modoc, particularly sagebrush rangeland. The goal of this chapter is to review the interactions of climate change, grazing, and carbon storage; the response of native plant communities to grazing; and grazing effects on invasive annual grasses. For additional discussion on grazing, see Chapter 3.1, Warren, this synthesis, *Perceptions and History of Rangeland;* Chapter 3.3, Padgett, this synthesis, *Weeds, Wheels, Fire, and Juniper: Threats to Sagebrush Steppe;* Chapter 4.2, Padgett, this synthesis, *Aquatic Ecosystems, Vernal Pools, and Other Unique Wetlands;* and Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California.*

Interactions of Climate Change, Grazing, and Carbon Storage

During the next century, changes in climate are expected to impact the rangelands of the Lassen and the Modoc. While most models show climate will warm, debate continues about how precipitation may be affected. Some models show drier conditions (Cayan et al. 2008; Polley et al. 2013) whereas others show an increase in precipitation (Allen and Luptowitz 2017; Reeves et al. 2014). As climate changes, weather will likely have more year-to-year variability and more frequent, extreme events, including drought (see Stanturf et al. 2014 and references therein), which will impact Western rangelands (IPCC 2007).

On one hand, a climate shift toward warmer temperatures with less precipitation in Northern California may push perennial systems, especially in concert with grazing, toward an annual-dominated system because they have less resistance (i.e., ability to withstand disturbance) and resilience (i.e., ability to respond to disturbance) than annual systems when faced with drought (Ruppert et al. 2015). On the other hand, a climate shift toward warmer temperatures and more precipitation would increase net primary production (additions of carbon through photosynthesis less carbon loss through respiration). Either change would affect levels of soil organic carbon (SOC), an important issue in the climate change conversation.

In general and on an area basis, rangeland soils store less carbon than other temperate, terrestrial ecosystems (Tanentzap and Coomes 2012), but given their global abundance, the sequestration potential is significant. Grazed lands have potential to sequester about 20 percent of the annual carbon dioxide released (Follett and Reed 2010). Soils comprise the largest pool of terrestrial organic carbon (Jobbágy and Jackson 2000); more carbon is typically stored belowground than in aboveground biomass where it is also less susceptible to loss by disturbance (e.g., fire). In addition, biological soil crusts also contribute to the carbon pool (Elbert et al. 2012; see Chapter 3.4, Warren, this synthesis, *Biological Soil Crusts*). This carbon promotes infiltration of precipitation and a soil's ability to hold water, cycle nutrients, and thereby improve plant growth. On rangelands, these traits provide societal and ecological benefits (e.g., clean water, clean air, erosion control, grazing income, recreation, wildlife, biodiversity, etc.) (Follett and Reed 2010). Thus, research to understand SOC and the factors that influence it, particularly on rangelands, is increasing.

Often, literature devoted to assessing global carbon sequestration potential focuses on "grazing land" that includes, for example, former cropland converted to pasture. On such sites, increases in SOC can be dramatic and skew estimates of carbon sequestration potential, further compounded by differences in other assumptions used in the models (for example, extent of area). On rangelands, vegetation type (grass versus shrub), plant growth characteristics (root mass and dynamics), grazer type (livestock versus wild ungulates), and climate can all influence carbon sequestration (Tanentzap and Coomes 2012). In their meta-analysis, McSherry and Ritch (2013) found that soil texture, precipitation, grass type, grazing intensity, study duration, and soil sampling depth explained most of the variation observed in SOC studies. Overall, they note that increasing grazing intensity on grassland sites dominated by C3 (cool-season) species decreased SOC but concluded that grazer effects on SOC are context-specific, especially because SOC accumulation is influenced by type of grazer; rangelands grazed by wildlife are predicted to annually add three times the SOC compared to those used by livestock (Tanentzap and Coomes 2012).

SOC is a function of net primary production, particularly the carbon allocated to the belowground portions of plants (Jobbágy and Jackson 2000). In their review, Piñeiro et al. (2010) noted that grazed rangelands receiving less than 16 inches (400 mm) of precipitation had larger amounts of roots than nongrazed sites, perhaps a function of changing allocation patterns stimulated by grazing or changes in plant community composition. Moreover, Evans et al. (2012) found that on an arid (10.5 inches [270 mm]) bunchgrass-dominated rangeland, SOC was similar under moderately grazed and nongrazed sites.

Grazing and Native Plant Community Response

While climo-geographic variables (e.g., soil, climate, geography, etc.) and plant adaptedness (e.g., genetics) determine plant communities across the landscape, grazing is implicated as the primary force that alters that plant community composition (Twidwell et al. 2013), and grazing can interact with other factors as well. These changes may be perceived to be negative, neutral, or positive, depending on the context. In addition to species composition, grazing may also affect the ability of a plant community to withstand disturbance (i.e., resistance) induced by other factors, such as drought, and the community's potential to return to the pre-disturbance condition (i.e., resilience).

Broadly at the global scale, when facing climate-induced drought, grazed perennial systems have slightly less resistance and resilience compared to nongrazed systems. Because the effects of drought and grazing are additive, grazed perennial systems frequently exposed to severe drought may be more likely to shift toward annual-dominated systems (Ruppert et al. 2015).

In California, native bunchgrass sites transitioned to communities of nonnative species as the amount of grazing increased (Stein et al. 2016). Once nonnative species invade, grazing of Mediterranean grasslands in California enhanced abundance of exotic forbs already present on the site but did not foster invasion by new exotic forbs, whereas the opposite occurred for exotic grasses. Conversely, the abundance of native forbs remained relatively stable but native forb diversity increased, whereas effects on native grasses were mixed, with this effect becoming more consequential on drier sites (Stahlheber and D'Antonio 2013). For Mediterranean California grasslands, winter and early spring grazing yielded the most consistent increases in native plant cover and diversity (Stahlheber and D'Antonio 2013), which could likely be maintained (especially the herbaceous component; Eldridge et al. 2013) but not enhanced (Stein et al. 2016) with low-intensity grazing.

These examples highlight the ongoing challenge for range managers and scientists. That is, understanding dynamic grazing—plant community changes to inform management decisions to achieve desired states of vegetation on the landscape. For nearly 3 decades, state-and-transition models (Laycock 1991; Westoby et al. 1989) have provided a framework for describing and analyzing a variety of vegetation dynamics. It is a useful tool for identifying stable (or current) vegetation states and subsequent thresholds driven by grazing, invasive species, fire, etc., or their combinations, where that stable vegetation state changes (see the review by Cingolani et al. 2005). The flexibility of this model allows range managers to apply it to a wide variety of field situations, including sagebrush systems (e.g., Allen-Díaz and Bartolome 1998), and be alerted to sustained processes (alone or in combination) that produce undesired states (Bestelmeyer et al. 2004). With this knowledge, focused, site-specific management tools can be developed to guide grazing activities (and foster improved communication among land managers, ranchers, and the general public) in a way as to tailor site-specific prescriptions to reach desired ecological conditions (e.g., Cagney et al. 2010); the desired ecological condition is often considered to be the existing condition before excessive disturbance occurred. When manipulating grazing to obtain a desired plant community composition is proposed, modifying grazing prescriptions may (Porensky et al. 2016) or may not (Stein et al. 2016) result in the desired condition.

Wild Horses

Management of wild horses (*Equus ferus*) and burros (*Equus asinus*) is a contentious and emotionally charged topic (fig. 3.2.2). The 1971 Wild Free-Roaming Horses and Burros Act requires management of these species to maintain their presence on Western rangelands while ensuring that wildlands are sustainably managed. This is a challenge for land managers. First, research has found that wild horse populations can increase up to 20 percent each year and the 1971 Act stipulates removing animals in order to maintain desired ecological conditions (see Garrott and Oli 2013; Natural Research Council 2013). Second, the costs to maintain captured horses increased $55 million between 2000 and 2012, accounting for 60 percent of the entire wild horse and burro management program (Garrott and Oli 2013). Third, divergent social perspectives have made finding consensus about management practices difficult (see Chapter 5.3, Flores and Haire, this synthesis, *Ecosystem Services and Public Land Management*).

On one hand, Beever and Aldridge (2011) note the effects of wild horses on ecosystems, as is the case with all herbivores, will vary depending on a variety of factors, such as elevation, wild horse population, season, and duration of use. On the other hand, the presence of unmanaged or poorly managed wild horses in sagebrush rangelands can lower the cover of grass, shrub, and overall plant cover, which can have serious implications for greater sage-grouse (*Centrocercus urophasianus;* see Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California*); increase the abundance of annual invasive grasses such as cheatgrass (*Bromus tectorum*); and compact the soil to reduce water infiltration and the activity of ants, which provide important ecological services including roles in water infiltration and carbon sequestration (Beever 2003; Beever and Aldridge 2011; Beever and Herrick 2006). Ants also are an important food source for greater sage-grouse, especially chicks (Ersch 2009). Ants play a key role in maintaining rangeland health through, for example,

Figure 3.2.2—Wild horses on the Modoc National Forest are managed through the Devil's Garden Plateau Wild Horse Territory Management Plan (Jeffers 2013) (photo by Ken Sandusky, Forest Service).

their modification of soil chemical and physical properties, seed dispersal activities, and disturbance effects on plant diversity (see Carlisle et al. 2018). Davies et al. (2014) note that impacts from wild horses can be intensive even at low populations because, unlike domestic livestock whose grazing is more intensively managed, wild horse grazing is largely unmanaged and occurs year-round. Thus, wild horse grazing can greatly increase the overall impact from nonnative grazers on the landscape (see also Beever and Brussard 2000). Moreover, this effect may be exacerbated during drought (Beever and Aldridge 2011).

In response to the contentious nature of wild horse and burro management in the West, the National Research Council (2013) released a nearly 400-page review of the science related to sustainable management. The nine-chapter report describes the issues; discusses population processes, size, growth rates, and fertility management; examines genetic diversity and population models; provides framework for establishing and adjusting appropriate management levels; and discusses necessary social considerations for managing horses and burros. Currently, wild horses on the Modoc are managed under the Garden Plateau Wild Horse Territory Management Plan (Jeffers 2013), which outlines management for the next 15 to 20 years on about 225,000 acres (91,000 ha) of the Modoc. The plan, in cooperation with the BLM describes desired future conditions, population control, improvement projects, and habitat monitoring. Still, management remains contentious.

Grazing Effects on Invasive Annual Grasses

The concept of using livestock to reduce fuel loads and subsequent fire frequency and severity, particularly on sites with invasive annual grasses, has been discussed for decades, and was the focus of a recent comprehensive review (Strand et al. 2014; see textbox 3.2.1). On one hand, the authors note that invasion by annual exotic grasses can occur with or without grazing, but high-intensity grazing encourages invasion by reducing competition from desired native vegetation. On the other hand, properly timed grazing has been shown to suppress the invaders, but results vary and are often contradictory because of site-specific characteristics and the timing and intensity of grazing (Chambers et al. 2014).

Textbox 3.2.1—Key findings of Strand et al. (2014).

- High-severity grazing (i.e., greater than 50 percent utilization), especially in the spring during initiation of bolting of perennial grasses, can suppress competition from native herbaceous plants and cause soil disturbance that can favor annual invasive grasses, including cheatgrass.
- Livestock grazing at low/moderate severity (i.e., less than 50 percent utilization) generally has little influence on the cover of perennial grasses and forbs.
- A window of opportunity may exist for targeted grazing to reduce annual grasses before perennial grasses initiate bolting or during dormancy of perennial grasses.
- Livestock grazing can reduce the standing crop of perennial and annual grasses to levels that can reduce fuel loads, fire ignition potential, and spread.
- Grazing after perennial grasses produce seed and enter a dormant state can reduce the residual biomass left on the site, thereby decreasing the fire hazard the following spring and summer.
- Economic analyses reveal that fuel treatments in sagebrush ecosystems have the highest benefit/cost ratio when the perennial grasses comprise the dominant vegetation, i.e. prior to annual grass invasion and shrub dominance.

In Northern Nevada, cattle grazing in May targeting a cheatgrass-dominated rangeland (50 to 60 percent cover) effectively reduced cheatgrass biomass (80 to 90 percent removal achieved in a 2-day period) and subsequent intensity of prescribed fire the following fall. When grazed again the next spring, cheatgrass was reduced to a level that prevented prescribed fire from carrying (Diamond et al. 2009).

In their review of Western rangelands, Strand et al. (2014) concluded that grazing cheatgrass in early spring prior to active growth of native perennial grasses was the best opportunity to reduce cheatgrass. Smith et al. (2012) point out that to be successful, initiation of grazing must be based on the growth stage of the annual grass, not simply a calendar date. Because early season grazing targets removal of green seed heads, Mosely and Roselle (2006) further suggest pulsed grazing events because grazed cheatgrass can regrow seed heads. Moreover, such winter and early spring grazing of interior California grasslands yielded increases in native plant cover and diversity (Stahlheber and D'Antonio 2013). Similarly, in Central California, short-term, high-intensity grazing by sheep conducted just prior to inflorescences reduced medusahead (*Taeniatherum caput-medusae,* syn: *Elymus caput-medusae*) cover substantially and increased forb cover and diversity (DiTomaso et al. 2008).

The presence of greater sage-grouse, however, complicates implementation of spring grazing strategies, especially those with multiple re-entries, as the best time to apply grazing to reduce annual grasses coincides with brood rearing, when maximum herbaceous cover is desired to reduce nest predation (see Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California*). Promptly removing grazers after a single, targeted, short-duration event before resumption of growth of dormant, desired vegetation may be a way to reduce annual grasses and maintain vigor (Strand et al. 2014) and/or increase cover and diversity (Stahlheber and D'Antonio 2013) of native plants beneficial to greater sage-grouse.

Davison (1996) suggests that intensive grazing be employed to reduce fire danger on sites now dominated by annual invasive grasses with little to no desired perennial vegetation. This may also improve the spring nutritional status of livestock, defer grazing on adjacent perennial rangelands, lower fire suppression costs, and protect adjacent, more pristine rangeland. Frost and Launchbaugh (2003) provide a pragmatic approach to developing and implementing grazing plans with an emphasis on weed management, and the Smith et al. (2012) management guide illustrates annual grass phenological stages to target.

Grazing Effects on Soil Properties

In their review, Drewry et al. (2008) note that grazing animals alter soil properties, which can have direct and indirect effects on subsequent plant growth. Changes in soil properties are a consequence of hoof activity. While greater treading by horses, cows, and sheep associated with increasing levels of stocking may damage plants, disrupt the soil surface, and be readily observed by land managers, changes in soil properties are less visible but may be more important. Of the soil properties investigated, bulk density is perhaps the most important. Grazing animals apply appreciable vertical force onto the soil, especially when walking (Greenwood and McKenzie 2001), and this force compresses the soil and thereby increases bulk density.

Grazing-induced increases in bulk density as a result of treading are consistently reported (Drewry et al. 2008; Evans et al. 2012; Greenwood and McKenzie 2001; Tate et al. 2004). The size of the grazing animal also influences the impact; a single cow imparted a greater change in soil bulk density than did three deer or six sheep (Cournane et al. 2011). Areas with higher incidence of treading, such as where animals congregate (i.e., water sources, shade) or travel repeatedly (i.e., along fence lines) have higher bulk densities compared to other grazed areas (Greenwood and McKenzie 2001; Tate et al. 2004). High stocking rates with long treading intervals on wet soils, and the subsequent increases in bulk density are associated with declines in pasture productivity (Drewry et al. 2008).

Increases in soil bulk density are associated with decreases in the abundance of macropores. Macropores promote soil aeration and water infiltration and decrease resistance to root growth. These factors affect plant growth, microbial and invertebrate communities, and accumulation of SOC. The impacts on soil bulk density by treading of grazing animals is greater when soils are wet (i.e., winter and spring) (fig. 3.2.3) and more pronounced on finer-textured soils (i.e., those with more silt and clay relative to sand) (Cournane et al. 2011; Drewry et al. 2008; Greenwood and McKenzie 2001). In a long-term (30-year) grazing study on a semiarid grassland with a loam soil, grazed plots had higher soil bulk density than nongrazed plots, and spring grazing (wetter soil) at moderate stocking (45 to 50 percent consumption of available forage) increased soil bulk density in the top 6 inches (15 cm) of the soil profile more than fall grazing (drier soil) (Evans et al. 2012).

Research by Tate et al. (2004) on long-term plots on the San Joaquin Experimental Range in the Sierra Nevada found that bulk densities varied by canopy cover type, with soils under an open grassland having greater bulk density than soils residing under landscapes having 30 percent tree (either pine [*Pinus*] or oak [*Quercus*]) or shrub (ceanothus [*Ceanothus*]) canopies. Eldridge et al. (2015) concluded that shrubs help moderate grazing impacts by: (1) restricting access of grazing animals to soil under their canopies, which reduces changes to bulk density; (2) improving water infiltration through lower bulk density and increased litter cover, the latter fostering decomposition and soil aggregation; and (3) protecting the biological soil crust (see Chapter 3.4, Warren, this synthesis, *Biological Soil Crusts*).

Grazing intensity also affects bulk density. Increasing the amount of residual dry matter (native annual grass) on a Sierra Nevada site to more than 980 pounds per acre (1,100 kg per ha) by reducing grazing intensity decreased bulk density about 10 percent compared to sites having less than 400 pounds per acre (450 kg ha) (Tate et al. 2004). On these same sites, characterized by coarse-textured soils, absence of livestock grazing allowed soil bulk density to return to the same level as that found on sites where grazing was excluded for more than 26 years. It is likely that recovery time of rangeland soil bulk densities will depend on soil texture, with finer-textured soils requiring more time.

Meeting Grazing Management Objectives

While much of the literature focuses on the ecological aspects of grazing, more emphasis is finally being placed on better understanding the socioecological underpinnings of grazing and their relevance, given the strong and centuries-long influence of human activity (including politics) on grazing of rangelands (Bennett et al. 2013; Thevenon et al. 2010). This is critical, as Briske et al. (2011) conclude that while purposeful rotation of livestock can achieve diverse management goals, a robust science literature shows it may not necessarily provide specific ecological endpoints. The authors contend that this is because experiments have intentionally excluded the human management component, including manager objectives, experience, and decision making. Given this shortcoming but the need to sustain the rangeland ecosystem, Follett and Reed (2010) conclude that development and implementation of necessary and effective management plans are more likely to succeed if done at more local levels with active engagement from all stakeholders. In Chapter 5.1 (Flores, this synthesis, *An Introduction to Social, Economic, and Ecological Factors in Natural Resource Management of Northeastern California Public Lands*) it is noted that decision making for ecosystems requires balance between the complexity of the ecosystem and the various ecosystem benefits provided to a broad palette of stakeholders. Successful collaborations promote dialog and afford the community opportunity to address management challenges and solutions, strengthen local livelihoods, utilize ecosystem service, and sustain the ecosystem. While Flores approaches the topic of community engagement in the decisionmaking process from a forest management perspective, the discussion and tenets are applicable to any

Figure 3.2.3—Livestock grazing can change soil properties, especially when soils are wet (photo by Ken Sandusky, Forest Service).

ecosystem, including rangelands. As Eldridge et al. (2013) conclude: ***"Ultimately, however, the prevailing land use is likely to depend on social systems and human decisions, and how society reconciles competing valuations of ecosystem services related to soil carbon, grazing and wildlife habitat."***

Regardless, meeting management objectives will involve movement of livestock across the landscape at densities that allow the ecological system to thrive. Although Bailey and Brown (2011) suggest that attentive and timely adjustments to herd size and location at the landscape scale is "more likely to be effective in maintaining or improving rangeland health than fencing..." a system of fencing is typically used to constrain livestock movement to focus grazing and avoid overconsumption of preferred plants or grazing areas at crucial times, such as during sage-grouse nesting (see Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California*) or during the critical late-summer period in riparian zones (see Long and Pope 2014), rather than periodic deferments of these areas throughout the season (Bailey and Brown 2011). Such enclosures could be used to provide short-term, concentrated grazing on invasive annual grasses (see above). While fencing may be useful, it comes with drawbacks as well. Different types of fencing can affect movement of wild ungulates differently (e.g., Gates et al. 2012; Karhu and Anderson 2006; and references therein), and even impact greater sage-grouse mortality (see the grazing impact section in Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California*).

Monitoring grazed rangelands is essential to meeting multiple-use objectives because it documents ecological changes on the rangeland that can be used to adjust management. A recent look at Bureau of Land Management allotments found that just less than two-thirds of them had been monitored. Of those not meeting standards, only a third had a full suite of monitoring data (Veblen et al. 2014). The authors noted that their independent data acquisition, along with conversations with rangeland experts, revealed that monitoring ground cover needed more emphasis as a grazing-related metric (table 3.2.1). Indeed, Fynn et al. (2017) conclude that the previous year's effects on grazing (which would be known with monitoring) can have a profound implication for current-year management, because of the lag effect among

Table 3.2.1—Results of informal conversations with Federal (n = 20) and university (n = 20) rangeland science experts on how best to prioritize monitoring of rangeland condition and livestock impacts. Experts were presented with a hypothetical monitoring scenario (table 3 in Veblen et al. 2014).

Monitoring priority	Federal (%)	University (%)
Cover	55	70
Bare ground	25	15
Gap	5	5
Production	10	10
Frequency	5	0
Density	10	10
Utilization	35	25
Cattle and/or wildlife condition	5	10
Soils	25	10
Reference areas or ecological site	30	40
Photos	30	15
Remote sensing	30	35
Identification of at-risk areas	25	15

years on rangeland recovery. They note this lag of recovery will vary more as climatic variability continues to become more extreme, and thus suggest a more conservative approach to determining stocking rates in order to maintain greater heterogeneity on the landscape. Otherwise, a mismatch between stocking levels and carried-forward forage levels can result in overstocking that overrides all other management initiatives and objectives. Tillman et al. (2006) conclude that maintaining biodiversity (another aspect of greater heterogeneity) promotes ecosystem stability. Thus, maintaining greater diversity and heterogeneity can maintain the productivity and stability of livestock populations (Fynn 2012; Hobbs et al. 2008) as well as the stability of wildlife, especially greater sage-grouse, other obligate sagebrush fauna, and mule deer. Finally, increasing plant species richness (diversity) positively increases abundance of other flora and fauna, including pollinators, while decreasing the abundance and diversity of invading plant species (Scherber et al. 2010).

Restoration

Social Aspects

Undoubtedly, restoration of rangelands involves a social component. A discussion (from a forestry perspective) about the necessity of community engagement in meeting management (including restoration) goals is included in Chapter 5.1 (Flores, this synthesis, *An Introduction to Social, Economic, and Ecological Factors in Natural Resource Management of Northeastern California Public Lands*). That discussion is readily applicable to sagebrush rangelands as well, and critical to building the relationships and trust needed to garner greater support for restoration activities. Gordon et al. (2014) add to this discussion by noting specific public views on various rangeland restoration practices based on surveys. They found that the general public was more accepting of prescribed fire, grazing, felling, and mowing treatments than those that included herbicides and chaining. These practices were also more acceptable to people who expressed greater concerns that inaction was unacceptable. Even so, with the exception of livestock grazing to reduce fine fuels, none of these restoration treatments were embraced by more than half of the respondents. Interestingly, simply providing more information about the merits of these techniques was unlikely to gain more support, rather, land managers building trust with the public is the better avenue for implementing any of these practices more widely. As part of this trust, the process used to make management decisions has a strong impact on how stakeholders view the decisions and subsequent implementation (Shinder et al. 2002), and this process requires that all parties participate in give-and-take discussion toward eventually understanding the rationale for the treatment and have input into the potential tradeoffs and outcomes associated with it.

Landscape to Local

A current emphasis of sagebrush rangeland restoration is improving habitat for greater sage-grouse. Some conservationists consider greater sage-grouse an "umbrella species" for sagebrush ecosystems. The assumption with this management philosophy is that other sagebrush-obligate species of concern (as well as other flora and fauna associated with the sagebrush biome) will simultaneously benefit when the sagebrush ecosystem is managed and/or restored for greater sage-grouse (Rowland et al. 2006). For example, 85 percent of the restoration associated with conifer removal to improve greater sage-grouse habitat across the Western United States also coincided with moderate to high levels of the sagebrush-obligate Brewer's sparrow (*Spizella breweri*) (Donnelly et al. 2017). Similarly, in Wyoming, 50 to 90 percent of the restoration work for greater sage-grouse overlapped migration, stopover, and wintering areas of mule deer (Copeland et al. 2014). Conifer removal to restore greater sage-grouse habitat increased butterfly species richness and abundance at most sites (McIver and Macke 2014). And, these treatments were modeled to increase potential forage 37 percent and ranch income 15 percent (McClain 2012; see Chapter 4.3, Dumroese, this synthesis, *Sagebrush Rangelands and Greater Sage-grouse in Northeastern California*). Thus, considering rangeland restoration through the lens of sage-grouse restoration has merit.

Several recent research efforts have concentrated on prioritizing restoration and quantifying its effectiveness at the landscape level. These efforts have included mapping tree cover across the range of greater sage-grouse (Falkowski et al. 2017), determining frameworks for removing encroaching conifers (Reinhardt et al. 2017), and assessing effectiveness of seeding treatments after wildfire (Arkle et al. 2014).

Beginning restoration with an eye toward landscape function is prudent. Fuhlendorf et al. (2017) contend that restoration and conservation should focus on landscape processes such as fragmentation, conifer encroachment, and habitat conversion. Otherwise, reliance on local management efforts that enable populations to persist will be for naught because such activities do not create suitable habitat at a scale necessary for the species to thrive. Even so, incremental, local management efforts, if done in sufficient quantities across the landscape can achieve desired landscape-level results. For example, some recent work is demonstrating that landscape-level restoration for greater sage-grouse is improving rangeland health. In Southeast Oregon (and just north of the Modoc), removing conifers from about 20 percent of an 84,000 acre (34,000 ha) area during an 8-year period in increments of 42 to 6,200 acres (17 to 2,500 ha) resulted in improvements in annual female and nest survival (6.6 and 18.8 percent, respectively) and an estimated 25 percent increase in overall population growth compared to the nontreated control areas (Severson et al. 2017). Conifer removal is also known to increase songbird (Donnelly et al. 2017; Holmes et al. 2017) and forb (Bates et al. 2017) abundance and soil water availability (Roundy et al. 2014), and yield longer seasonal streamflow (Kormos et al. 2017).

The goal of sagebrush rangeland restoration is enhancing the resistance and resilience of the system to current and future disturbances so that ecosystem services and function continue. During the last decade appreciable research and effort has been made to link prioritization of restoration with frameworks to guide the science-based resistance and resilience. In particular, Chambers et al. (2017) describe five indicators of resistance and resilience for sagebrush rangelands, the objectives associated with those indicators, and strategies for achieving those objectives (table 3.2.2). Specific guides to science-based restoration practices have been developed to support the achievement of resistance and resilience in rangeland restoration (table 3.2.3).

Summary

The Lassen and Modoc manage extensive rangeland and grazing land and support a long tradition of livestock grazing. Changes in climate are predicted to impact these ecosystems, but uncertainty remains a challenge for land managers. While rangelands store less carbon than many other types of ecosystems, rangeland SOC is a significant contributor to global carbon sequestration. Livestock grazing affects rangelands. Notably, grazing may affect the ability of rangeland to withstand disturbance (i.e., resistance) induced by drought and other factors, and the potential of rangeland to return to its pre-disturbance condition (i.e., resilience). Grazed rangelands with a high abundance of perennial species have slightly less resistance and resilience compared to nongrazed systems. Grazing can also influence the occurrence and abundance of plants species, which has implications for managing fire frequency and intensity and providing habitat for obligate sagebrush species such as greater sage-grouse. Managing grazing intensity and the level of residual dry matter that remains on the site can ensure soil physical properties conducive to productivity and water infiltration continue. Wild horses, which are often managed less intensively than livestock, present a particularly vexing challenge to land managers, highlighting the fact that societal values and the human decisionmaking process drive management decisions and community engagement is essential in devising successful management plans. Monitoring is necessary to ensure management objectives are achieved, and to determine when restoration is required. Maintaining the ecosystem services provided by rangelands, i.e., rangeland function, may be more appropriate than attempting to restore to an historic reference condition. Recent efforts have provided science-based, practical approaches, frameworks, and guidelines for restoring sagebrush-dominated rangelands.

Table 3.2.2—Five indicators of resistance and resilience in sagebrush rangelands, restoration objectives associated with those traits, and strategies for achieving desired levels of resilience and resistance. From Chambers et al. (2017).

Indicator of resistance and resilience	Restoration objective	Strategies for achievement
Extent and connectivity of sagebrush ecosystems	Minimize fragmentation to maintain large landscape availability and connectivity for sage-grouse and other sagebrush dependent species.	Secure conservation easements to prevent conversion to tillage agriculture, housing developments, etc., and maintain existing connectivity. Develop appropriate public land use plans and policies to protect sagebrush habitat and prevent fragmentation. Manage conifer expansion to maintain connectivity among populations and facilitate seasonal movement. Suppress fires in targeted areas where altered fire regimes (due to invasive annual grasses, conifer expansion, climate change, or their interactions) are resulting in fire sizes and severities outside of the historical range of variability*, increasing landscape fragmentation, and impeding dispersal, establishment, and persistence of native plants and animals.
Functionally diverse plant communities	Maintain or restore key structural and functional groups including native perennial grasses, forbs, and shrubs and biological crusts to promote biogeochemical cycling and hydrologic and geomorphic processes, promote successional processes, and reduce invasion probabilities.	Manage grazing to maintain soil and hydrologic functioning and capacity of native perennial herbaceous species, especially perennial grasses, to effectively compete with invasive plant species. Reduce conifer expansion to prevent high-severity fires and maintain native perennial herbaceous species that can stabilize geomorphic and hydrologic processes and minimize invasions. Restore disturbed areas with functionally diverse mixtures of native perennial herbaceous species and shrubs with capacity to persist and stabilize ecosystem processes under altered disturbance regimes and in a warming environment.
Introduction and spread of nonnative invasive plant species	Decrease the risk of nonnative invasive plant species introduction, establishment, and spread to reduce competition with native perennial species and prevent transitions to undesirable alternative states.	Limit anthropogenic activities that facilitate invasion processes including surface disturbances, altered nutrient dynamics, and invasion corridors. Use early detection and rapid response for emerging invasive species of concern to prevent invasion and spread. Manage livestock grazing to promote native perennial grasses and forbs that compete effectively with invasive plants. Actively manage invasive plant infestations using integrated management approaches such as chemical treatment of invasives and seeding of native perennials.
Wildfire regimes outside of the historical range of variability	Reduce the risk of wildfires outside of the historical range of variability to prevent large-scale landscape fragmentation and/or rapid ecosystem conversion to undesirable alternative states.	Reduce fuel loads to: (1) decrease fire size and severity and maintain landscape connectivity, (2) decrease competitive suppression of native perennial grasses and forbs by woody species, and thus (3) lower the longer-term risk of dominance by invasive annual grasses and other invaders. Suppress fires in low- to moderate-resistance and resilience sagebrush-dominated areas to prevent conversion to invasive annual grass states and thus maintain ecosystem connectivity, ecological processes, and ecosystem services. Suppress fires adjacent to or within recently restored ecosystems to promote recovery and increase capacity to absorb future change. Use fuel breaks in carefully targeted locations along existing roads where they can aid fire-suppression efforts and have minimal effects on ecosystem processes.

(Continued)

Table 3.2.2—(Continued).

Indicator of resistance and resilience	Restoration objective	Strategies for achievement
Ecosystem recovery toward desired states following disturbance	Restore and maintain ecosystem processes and functional attributes following disturbance that are consistent with current and projected environmental conditions and allow ecosystems to absorb change.	Assess postdisturbance conditions and avoid seeding where sufficient native perennial herbaceous species exist to promote successional processes, stabilize hydrologic and geomorphic processes, and make conditions conducive to recruitment of sagebrush. Consider seeding or transplanting sagebrush species adapted to site conditions following large and severe wildfires that decrease recruitment probabilities to increase the rate of recovery and decrease fragmentation. In areas with depleted native perennials, use species and ecotypes for seeding and outplanting that are adapted to site conditions and to a warmer and drier climate where projections indicate long-term climate change. Avoid seeding introduced forage species that outcompete natives.

*Historical range of variability and natural range of variability are essentially the same.

Table 3.2.3—Science-based restoration guides to implement local activities toward achieving resistance and resilience on sagebrush rangelands.

Reference	Description
Pyke et al. 2015a	Concepts for understanding and applying restoration
Pyke et al. 2015b	Landscape-level restoration decisionmaking
Pyke et al. 2017	Site level restoration decisionmaking
Miller et al. 2014	Selecting the most appropriate treatments with respect to invasive annual grasses
Miller et al. 2015	Rapidly assessing post-wildfire recovery potential
Maestas and Campbell 2014	Using soil temperature and moisture regimes to predict potential ecosystem resistance and resilience
Chambers 2016	Compilation of 14 fact sheets with “how-to” descriptions for a variety of restoration activities

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