Chapter 13
Chaparral Restoration

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Abstract Chaparral, among the most stable and resilient vegetation types in California, has shown signs of degradation by altered fire frequency, drought, non-native species, recreation, urban development, and possibly anthropogenic nitrogen deposition in southern California. Restoration has been practiced less frequently in chaparral than other, more extensively disturbed vegetation types, but recent degradation suggests that restoration may be important for the maintenance of ecosystem services such as slope stabilization, carbon sequestration, biodiversity conservation, and scenic beauty. Most chaparral restoration has primarily been "passive restoration", the removal of disturbance stressors or management of fire frequency to promote natural successional processes for plant, animal, and soil recovery. However, areas that have suffered severe disturbance, such as topsoil removal and extensive plant invasions, seldom recover passively or at best may be colonized by early successional shrubs. Active restoration, which can include weeding, planting, seeding, treatments to break seed dormancy, and/or stabilizing soil treatments, may be needed in many cases. We review current knowledge of chaparral stressors and dynamics that relate to restoration as well as restoration methods. The limited information on restoration projects to date indicates that early successional, deciduous shrub species, which are common to sage scrub and have low seed dormancy, are most successful in establishment. These may accomplish some restoration objectives, such as soil stabilization and ability to recover from fire, but fall short in biodiversity goals. Application of techniques to establish evergreen chaparral species, such as large-scale dormancy breaking treatments or facilitation by early successional shrubs, is needed. We also discuss plant traits that might be used to guide restoration toward persistent communities under frequent fire. Our aim is to describe knowledge gaps about chaparral restoration and inspire restoration research, planning, and practice.

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E. C. Underwood et al. (eds.), *Valuing Chaparral*, Springer Series on Environmental Management, https://doi.org/10.1007/978-3-319-68303-4_13

Keywords Active restoration · Chaparral recovery · Chaparral restoration · Passive restoration · Vegetation type-conversion

13.1 Introduction: Why Does Chaparral Need to be Restored?

Ecological restoration is defined as assisting the recovery of a system that has been degraded, damaged or destroyed (SER 2004; Day et al. 2006). For sites with intact soils and seedbanks this can be accomplished passively by removing stressors such as livestock grazing or improving air quality. Sites that are severely disturbed, or where natural rates of succession are too slow for management objectives, may be candidates for active restoration (Allen et al. 2001; Greipsson 2011, see Sect. 13.5). In California, restoration has been practiced extensively in coastal sage scrub, grassland, and wetland ecosystem types that are severely impacted by biotic invasions and anthropogenic disturbance (Mooney and Zavaleta 2016). There are, however, few cases of chaparral restoration because it has been subject to proportionally less anthropogenic disturbance and, until recently, has been considered a resilient vegetation type. Indeed, although there are hundreds of published studies on chaparral species and community responses to fire, much of the information on chaparral restoration is anecdotal (VinZant in prep-b). We highlight situations where chaparral is undergoing disturbances that are reducing its integrity and resilience, and where active restoration may be increasingly needed. We then review approaches to restoration.

13.1.1 Disturbance History and Resilience

Chaparral is one of the most widespread vegetation types in California, and it tends to occur on steep or rocky slopes that are difficult to develop. As a result, proportionally less area has been converted to agricultural or other land uses than other vegetation types. Hence chaparral has not received high priority for conservation and restoration. Instead, it has been considered as one of the most stable ecosystems in terms of constancy of land area occupied, and one of the most resilient ecosystems in its ability to recover from fire (Minnich and Bahre 1995; Keeley et al. 2005b).

Despite its reputation for resilience, chaparral is subject to a range of anthropogenic disturbances that degrade or eliminate it, including purposeful vegetation type-conversion, short fire-return intervals, fire suppression, invasion by non-native species, drought, and fragmentation by urban development, roads, corridors and fuelbreaks. Deliberate type-conversion to grass dominated vegetation has been done to increase forage for livestock (Biswell 1954), enhance water yield (Corbett and

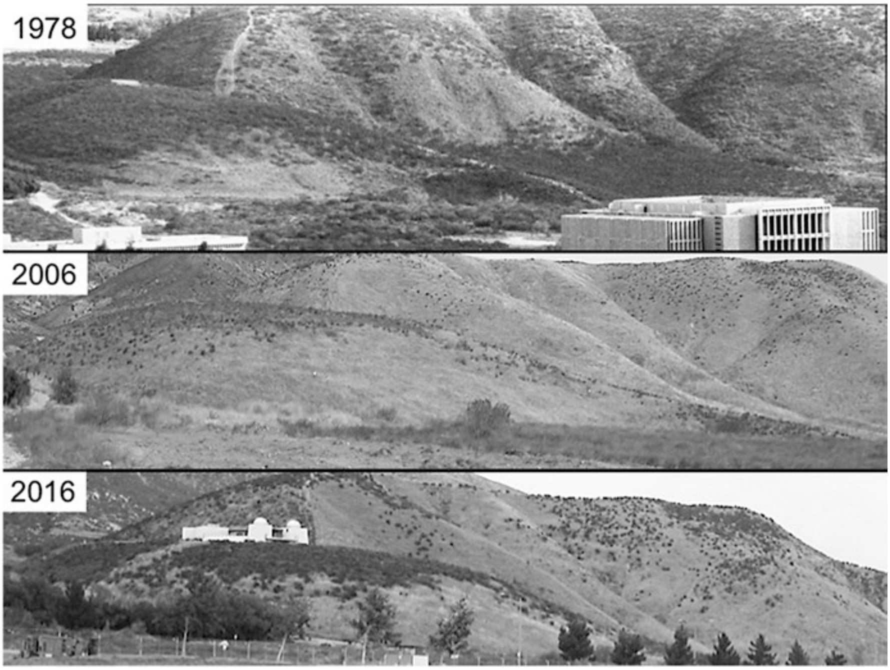

Fig. 13.1 Badger Hill, at California State University San Bernardino, burned in 1980, 1995, and 2003, and has experienced chaparral loss on south-facing slopes. The degree to which various stressors (fire frequency, changing climate, N deposition, and presence of non-native annuals) contribute to poor chaparral recovery at sites such as this may influence decisions about whether to attempt restoration. Photos by the Arthur E. Nelson Archives (1978), held at Pfau Library Special Collections & Archives, California State University San Bernardino, and K. Williams (2006 and 2016)

Rice 1966), or increase wildlife habitat diversity (Rosario and Lathrop 1974). The perception of stability was promulgated in part by the extreme efforts needed to suppress shrub regeneration, which includes combinations of burning, herbicide, and seeding with grasses (Schultz and Biswell 1952; Bentley 1967; Murphy and Leonard 1974). However, chaparral has also been unintentionally degraded by non-native annual grass and forb invasion and reduced shrub density under shortened fire-return intervals (Keeley and Brennan 2012, Fig. 13.1, see Chap. 12) and alternatively by tree encroachment at woodland-chaparral ecotones under fire suppression (Sparling 1994; Van Dyke et al. 2001). Historical repeated burning by Indians may have converted limited areas of chaparral to forb and grassland, especially in southern California (see Chap. 4). Multi-year drought causes mortality of some shrub species with or without fire (Pratt et al. 2014, see Chap. 14), which in turn, increases susceptibility to non-native grasses. Total or partial conversion to non-native grassland can contribute to a positive grass-fire cycle (D'Antonio and Vitousek 1992).

Severely disturbed chaparral, whether by frequent fire or mechanical disturbance, is usually slow, or unable, to recover naturally. Some highly disturbed stands have persisted for decades as non-native forbs and grasses, sometimes with scattered deciduous shrubs that were not constituents of the original chaparral vegetation (Stylinski and Allen 1999). The loss of an extensive area of chaparral has historically been inconceivable given its perceived stability and resilience. To date, there has not been a landscape-scale assessment of the extent of chaparral type-conversion to challenge this perception (but see Chap. 12). Because we know that chaparral degradation is occurring and is likely to accelerate, it is important to develop successful techniques for restoration of chaparral structure and function.

13.1.2 Ecosystem Services of Restored Chaparral

Restoration of chaparral may be justified by the many ecosystem services mature chaparral provides, as described in the preceding chapters in this book. Degraded chaparral results in declines in species diversity (see Chaps. 2 and 3), decreased slope stability from loss of deep-rooted shrubs (see Chap. 7), decreased water infiltration (see Chap. 8), decreased carbon storage caused by a shift from deep-rooted shrubs to shallow-rooted non-native herbs (see Chap. 6), and the loss of recreational opportunities and aesthetic value in damaged lands (see Chap. 10). The extent to which restoration is achieved will be determined by the degree of disturbance and the particular ecosystem services that are prioritized for any site.

13.1.3 Objectives—Steps to Achieve Restoration

Our objectives are to review the steps involved in achieving chaparral restoration. The first consideration is to prioritize sites needing restoration based upon loss of ecosystem services, location in the landscape, and potential for further damage (see Sect. 13.2). The next step is to set restoration goals for provisioning of ecosystem services such as slope stability, ability to recover from fire, reduced fire incidence, enhanced carbon sequestration, or habitat for sensitive species (see Sect. 13.3). Before these goals can be realized, the stressors that caused ecosystem damage in the first place ought to be evaluated, as well as the extent to which pre-disturbance conditions or alternate goals are appropriate to achieve ecosystem services (see Sect. 13.4). Delineation of goals and stressors can assist managers to decide on restoration techniques ranging from passive to active restoration (see Sect. 13.5). Finally, success of restoration activities is assessed in light of the restoration goals, further treatments, adaptive management, and research applied as needed to promote success (see Sect. 13.6).

13.2 Where to Restore

In general, the decision of whether or not to restore chaparral on a specific site will be influenced by: (1) the importance of the benefits of chaparral ecosystem services on that site, (2) the extent of damage, (3) the likelihood of restoration attempts succeeding on that site under current and future conditions (e.g., climate, disturbance), (4) the presence of rare or endangered species, (5) the potential to minimize or mitigate conflicts between chaparral and adjacent landscape elements (e.g., urban areas) at that site, and (6) the cost of restoration. Depending on ownership and available financial resources of affected lands, these decisions are affected by combinations of regulatory constraints and societal input. Spatial information on the value of ecosystem services of chaparral (erosion control, watershed protection, carbon storage, and biodiversity), combined with spatial information on stressors (see Sect. 13.4) that would reduce likelihood of restoration success (e.g., high nitrogen deposition rates and frequent fire), has been used to prioritize areas for postfire restoration on national forest land (see Chap. 15). Explicit consideration of land use around a potential restoration site helps identify both potential benefits of restoration and conflicts that may cause the restoration effort to fail.

Most current restoration projects occur on public lands where agency regulations or permit conditions require revegetation of bare soil after disturbance. Such policies are designed to prevent soil erosion, for example, restoration on road cuts (Allen and Heindl 1993; CalTrans 2008), after pipeline installation (D'Antonio and Howald 1990), or after surface mining (Roy 2009; Wilkin 2009). The US Forest Service in southern California includes permit conditions specifying that restoration is to be carried out after land disturbance by special-use permittees, such as for powerline construction (VinZant in prep-b). Only recently has revegetation of heavily disturbed areas involved pre-disturbance planning and post-disturbance actions to restore chaparral plant communities to a similar pre-disturbance state.

Restoration may also be beneficial where chaparral degradation and type-conversion to annual vegetation has increased threats to adjacent ecosystems or to human infrastructure, such as roads, reservoirs, or urban development. For example, a mudflow through the community of Highland at the base of the San Bernardino Mountains in 2010 may have resulted in part from slope failure on the frequently-burned adjacent foothills during an extreme rain event. Eliminating flashy fuels, typically non-native annual plants and subshrubs of disturbed chaparral, may be most important near ignition sources like urban areas, roads, or areas with moderate housing densities (see Chap. 12). In such areas, heavy human use can lead to fire starts. Even in protected areas such as the Santa Monica Mountains of southern California, chaparral disturbance from trampling, trail creation, off-road vehicle use, purposeful vegetation clearance, and refuse dumping has been extensive and associated with proximity to roads (Sauvajot et al. 1998). The need to replace non-native annual plants with low volume, slow burning shrubs along roads, fuelbreaks, trails, and powerlines through chaparral has been recognized for some time (Nord and Green 1977). The degree to which chaparral restoration, in some form, can contribute to this goal along roadways requires further investigation.

Although benefits of chaparral restoration may be most important at the wildland-urban interface, conflicts between chaparral and other landscape elements are also strongest there, creating challenges to both public acceptance of chaparral restoration and the success of implemented projects. Where mature chaparral abuts urban areas, fuel reduction may be carried out by mastication and other types of mechanical treatments (Brennan and Keeley 2015, see Sect. 13.5.2.1 and Chap. 15). Furthermore, impacts of humans and domesticated animals may threaten the success of restoration projects at the wildland-urban interface (see Sect. 13.4.1.5), and protecting human habitation from wildfire could place restrictions on the design and composition of restored chaparral near human habitation. Therefore, sites that have been type-converted or degraded near urban areas may need thoughtful discussions about the trade-off between reducing ignitions and long-term restoration success (see Chap. 15).

13.3 What to Restore and Restoration Goals

Restoration goals should be specific enough to provide a ready path to measureable criteria to assess success. While many goals in chaparral restoration may focus on ecosystem services, the selection of species to use for the restoration of the site is key for achieving these goals. The choice of species can also be guided by the need to ensure resilience and persistence of the restored community, the ease with which different species establish and spread, and the availability of seed and/or plants. Thus specific goals may have to be adjusted as availability of materials and site constraints are understood. Some chaparral functions may be deemed more critical than others on specific sites and such valuation may influence priorities for re-establishing different types of species. For example, slope stabilization above vulnerable habitats and protection of downslope communities could be enhanced through the establishment of deep-rooted shrubs (see Sect. 13.3.1). Changing climate and landscape conditions (e.g., urban development, fire frequency), as well as practical constraints on seed materials, mean that it may not be possible to restore all the species historically present in all locations. Thus, a useful approach is to focus chaparral restoration projects on recovering ecosystem functions.

13.3.1 Plant Traits to Stabilize Soil and Water Functions

Slope stabilization, water infiltration, and carbon sequestration are typical ecosystem function goals relevant to degraded chaparral. All of these would benefit from the establishment of plants with perennial, deep-root systems that remain intact through the summer drought and are living when the first fall rains arrive. In particular, deep-rooted species may be most effective over long time-scales for both carbon sequestration and protection against slope failure. Deep-rooted shrubs are also, commonly, resprouters (Hellmers et al. 1955; Davis et al. 1999), rendering them relatively resilient to occasional shoot loss from grass fires, grazing, and other disturbances that may be part of the early restoration environment. Once deep-rooted shrubs are well-established they are likely to maintain a well-developed canopy which will deflect the impact of rain on the soil surface. However, they may be slow growing. Thus, to avoid initial runoff and erosion, shorter-lived, fast-growing species such as sage scrub dominants (e.g., California buckwheat [*Eriogonum fasciculatum*], California sagebrush [*Artemisia californica*]) may be employed in combination with slower growing shrubs (Sect. 13.5.2.4).

13.3.2 Habitat Restoration to Maintain Animal Diversity

For the purposes of enhancing animal diversity, vegetation structure and plant species composition are important. Interactions between animal diversity and vegetation structure and composition have been extensively reviewed (Keeley and Swift 1995). Shrub cover attracts small mammals, habitat edges between shrub cover and more open vegetation attract a variety of mammals, and insect pollinators, especially bee species, are abundant in chaparral (Moldenke 1976; Keeley and Swift 1995). The abundance of pollinators can be influenced by plant species diversity and this is particularly important in chaparral because of the large number of specialist pollinator/shrub relationships. Pollinator abundance has also been attributed to vegetation structure and the sheltering effect of the chaparral canopy on ground-nesting bees (Moldenke 1976; Force 1990; Keeley and Swift 1995). If restoration of animal communities is a goal of the restoration project then it may be important to design a project that maximizes vegetation diversity and structure.

13.3.3 Restoring Uncommon and Rare Species

Re-establishing rare plants or habitat for rare animals helps protect global biodiversity. Restoration projects may have permit requirements for development activities that adversely affect rare species. Some of the rare or unusual plant species of chaparral are soil specialists or island endemics (see Chap. 2), and will, therefore, be appropriately re-introduced only at specific sites. In general, restoration of rare plant species can be challenging because material for restoration (seeds or other propagules) is usually limited, little is known about germination and growth requirements for many rare chaparral species, and many rare species may be more vulnerable to stressors than common species. Although restoring chaparral for many purposes, including habitat restoration, may focus on shrub species, approximately 70% of the plant taxa in chaparral that are considered rare, threatened, or endangered (i.e., California Native Plant Society (CNPS) ranks 1A and 1B) are either annual or perennial herbs (CNPS 2016).

Herbaceous species that require specific conditions to germinate and/or bloom may remain unapparent for years, making assessment of restoration success difficult.

Among the shrub species of chaparral, those that are slow-maturing, fire-dependent, obligate seeders are increasingly threatened by short fire-return intervals (Zedler 1995) and are, therefore, of greater conservation concern than most resprouters. For example, the rare morro manzanita (*Arctostaphylos moroensis*) requires especially long fire-free periods for successful recruitment (Odion and Tyler 2002) and therefore would not be a good restoration candidate in areas with short fire-return intervals. Wet-season burns reduced recruitment of several rare species (Parker 1987). Species that are locally or globally rare because of their high vulnerability to stressors (e.g., altered fire regime, competition from non-native annuals, and predation by domestic animals) may be poor candidates for restoration unless (or until) those stressors can be eliminated. For instance, the endangered shrub Nevin's Barberry (*Berberis nevinii*) is challenging to establish in wildlands because of human and grazing/burrowing animal impacts, but is available commercially and thrives in protected gardens (CNPS n.d.). Rare species may be particularly sensitive to changes in fire regime and efforts should be taken to consider whether recurrent unnatural disturbances are likely to impede restoration success.

13.3.4 Plant Traits for a Resilient Chaparral Community

Successful decadal-scale restoration requires persistence and resilience of the restored chaparral to future perturbations. Thus resilience to future fire or drought should be an important goal in chaparral restoration. Resilience of chaparral is due in large part to species traits and life-forms that enable post-disturbance regeneration (see Chaps. 1 and 2). Species may be selected for restoration based on these life-forms and traits to provide stability and ecosystem services to sites with novel disturbances. Resprouting species have the ability to recover from shoot loss, rendering them relatively resilient to above-ground disturbances produced by fire, grazing, and manual cutting of stems (Mooney and Hobbs 1986). Combining traditionally recognized responses to fire with considerations of life-form, seedbank dynamics, and time-to-maturation has proven useful in describing the rate at which different species spread and persist on burned sites (Keeley et al. 2006). Applying such concepts to chaparral restoration suggests that species that mature quickly, and produce seeds that do not require fire to germinate, may spread most rapidly on a restoration site. These traits will also contribute to the persistence of their populations after disturbance (Mooney and Hobbs 1986). Many of these species, however, are sage scrub species, suffrutescent species or subshrubs, rather than dominant species of mature chaparral (Keeley et al. 2006). Therefore, their role in ecosystem function and chaparral development on a restoration site should be considered as part of goal setting, in addition to their contribution to resilience (see Sect. 13.5.2.4).

13.3.5 Restoration under Changing Climate and Landscape Conditions

Although restoration goals are generally linked to a reference condition that depicts a less disturbed system, changing climate or landscape conditions may limit the degree to which full restoration of a chaparral community is feasible or even desirable on a site (Harris et al. 2006; Hobbs et al. 2009, 2014). Landscape conditions include fragmentation, urbanization, frequent fire, invasion, or vegetation type-conversion, or other changes in the adjacent landscape. Certain species may no longer persist in their previous location due to changing climate, air pollution, or altered disturbance regimes (see Sect. 13.4). Some species may not be welcome near human habitation (e.g., large predators and venomous animals), and some characteristics of the vegetation may be considered too hazardous to create at the wildland-urban interface (e.g., dangerous fuel structure or flammability). Where climate has changed to such an extent that chaparral restoration is likely to fail (see Sect. 13.4.1.2 and Chap. 14), options may be limited to deciding not to restore or accepting a different community, such as coastal sage scrub, as an acceptable "restoration" endpoint. Where the human landscape has rendered some characteristics of chaparral undesirable, novel ecosystems that include some historical elements of chaparral and some new elements that are compatible with current and future conditions may add to the success of restoration projects and help practitioners attain restoration goals (Wiens and Hobbs 2015; Miller and Bestelmeyer 2016 and references therein).

Restoration within the wildland-urban interface may often involve use of novel vegetation combinations because fire behavior from typical chaparral (characterized by high fuel loads) can threaten human communities. Early searches for low-volume shrubs that could replace non-native annual grasses on fuelbreaks (Nord and Green 1977) or stabilize slopes near human habitation (McKell et al. 1966; Westman 1976) included many non-native species. In general, species that have relatively thick leaves, a low tendency to hold dead tissue high in the canopy, and low production of volatile and resinous compounds tend to ignite less readily and burn more slowly than other species (Montgomery and Cheo 1971; Green 1981; Schwilk 2003). Chaparral shrubs with these characteristics, mostly resprouters of mixed chaparral, appear on lists of plants approved for fuel modification zones by various organizations and municipalities in California (e.g., County of Riverside [n.d.]; Santa Monica Mountains Fire Safe Alliance 2010). Although these species may provide a number of ecosystem services attributed to chaparral, if the species did not originally exist on the site, their establishment may not be viewed as traditional ecological restoration.

13.4 Limitations that Affect Project Design and Restoration Success

Assessing site conditions and alleviating stressors is an important early step in any restoration plan (Whisenant 1999; Galatowitsch 2012). Stressors may be abiotic (frequent fire or fire suppression, drought, soil disturbance, erosion, air pollution) or biotic (competition from non-native species, animal impacts, seedbank loss, lack of germination cues for refractory seeds). Limitations may determine the extent to which restoration goals can be achieved and shape planning and implementation. Understanding the degree to which these stressors and limitations can (or cannot) be overcome may help inform decisions on whether successful restoration is feasible on a site.

13.4.1 Stressors

13.4.1.1 Drought and Summer Moisture Availability

The summer dry season experienced by chaparral has prompted studies that collectively provide an understanding of drought stress physiology and adaptations of mature plants (Parker et al. 2016, see Chap. 1). Adaptations to drought include deep-rooting, resistance to xylem cavitation, and sclerophyllous leaves. Drought tolerance in woody plants is generally high but varies among shrub species. Recent prolonged drought has caused shrub mortality in areas that have had stable chaparral stands for many decades (Davis et al. 1999; Paddock et al. 2013; Pratt et al. 2014), and some species of resprouters suffered high postfire mortality during a single drought year (Pratt et al. 2008). Mesic coastal populations of chaparral shrubs are more sensitive to drought than inland populations (Vasey et al. 2012). General strategies of drought tolerant (reseeding) and drought avoiding (resprouting) chaparral species as well as adapted microsites (e.g., north- and south-facing slopes) are recognized (see Chaps. 2 and 15). However, drought tolerance changes with shrub age, and less is known about moisture requirements and the seedling establishment niche, yet these are critical for successful restoration. During postfire succession, even in average precipitation years, 90% or more of seedlings succumb during the dry summer (Kummerow et al. 1985; Moreno and Oechel 1992). Oechel (1988) identified minimum water potentials for postfire seedling establishment, a concept that can be applied to restoration practice (e.g., providing irrigation). Obligate resprouters have higher recruitment from seed in shade than in full sunlight, related to their higher moisture requirement (Pratt et al. 2008). An understanding of moisture requirements for establishing seedlings is key to active restoration of severely disturbed areas where there is little cover and potentially poor water infiltration. Seeding or planting may need to be done in multiple years if plants suffer severe drought mortality in the first years of a project.

13.4.1.2 Climate Change

There is evidence that climate change is shifting chaparral ecotones and causing increased mortality of some species under drought (see Chap. 14). For instance, several chaparral shrub species moved upslope an average of 65 m (213 feet) in response to climate warming over 30 years in desert chaparral (Kelly and Goulden 2008), and three of seven resprouting chaparral shrubs experienced the highest ever recorded mortality during a severe drought following a fire (Pratt et al. 2014). One climate change modeling effort showed that many chaparral species would be reduced to an average of 50% of their current area in conservation reserve lands with 2.2 °C warming in the next 50 years (Principe et al. 2013). In addition, a warming climate has been cited as one cause of increased fire frequency (see Chap. 1). Given the potential for unsuitable conditions and permanent vegetation shifts, it suggests that consideration should be given to planting species adapted to projected climate conditions. For instance, as the climate warms, species common to sage scrub may be better adapted to future conditions than chaparral shrubs. Alternatively, sage scrub shrubs may be used as nurse plants to facilitate establishment of chaparral shrubs (see Sect. 13.5.2.4) A recommended approach for shifting climate envelopes is to plant species adapted to the new conditions of the area, which in southern California would include selecting more drought-adapted species (Harris et al. 2006).

13.4.1.3 Nutrient Availability, Nitrogen Deposition, and Invasion

Mature chaparral soils tend to be relatively low in mineral nitrogen (N) and phosphorus, as these are immobilized both in living plant tissue and in slowly decomposing, high C:N litter (Parker et al. 2016, see Chap. 1). Following fire, nutrients are mineralized and initially taken up by fast-growing native herbs (Hanan et al. 2016). However, where seeds of non-native grasses and forbs have dispersed into chaparral, they have a postfire advantage of earlier germination phenology, higher densities, and higher growth rates under high soil N (Chiariello 1989; Padgett and Allen 1999; Cox and Allen 2008), so they are better able to take up the initial flush of mineral nutrients postfire than native herbs and shrubs. These non-native grasses and forbs would be expected to have a high reproductive output of high quality seeds in such settings, thus setting the stage for suppression of desired native species.

Anthropogenic N deposition increases shrub growth in unburned sage scrub stands (Vourlitis 2012) while also increasing productivity of non-native annuals (Fenn et al. 2010; Valliere et al. 2017). Invasion is more likely after a fire, as most annuals will not invade the understory of a closed canopy shrubland. Shrub mortality under elevated N or drought that creates canopy openings and promotes invasion has been documented in sage scrub (Vourlitis 2017; Valliere et al. 2017) but not, to date, in chaparral. As with sage scrub, N deposition may operate in conjunction with frequent fire and drought to cause conversion of chaparral towards a grassy shrub savanna. The relationship between N deposition and type-conversion in sage scrub shows a high probability of conversion above a critical load of 11 kg N ha^{-1} $year^{-1}$ (Cox et al. 2014). If similar analyses are conducted for chaparral ecosystems these could be important inputs for developing restoration site plans. Of the total land area of chaparral, 14.6% receives more than 10 kg N ha^{-1} $year^{-1}$ deposition (~2 kg N ha^{-1} $year^{-1}$ is considered background for clean air), suggesting vulnerability to elevated N (Fenn et al. 2010). Legislative efforts have been enacted to reduce N deposition, and restoration may be most effective in regions that fall below critical N loads (Fenn et al. 2010).

13.4.1.4 Altered Fire Frequency

Fire frequency, whether shorter or longer intervals than natural fire regimes, may alter chaparral plant composition and affect restoration potential. In southern California fire frequency has increased adjacent to developed areas because of human-caused ignitions (Keeley et al. 2005b; Syphard et al. 2007, see Chap. 1). Fire-return intervals of less than 5–10 years will kill obligate seeders before they mature, as well as potentially killing resprouters that have not recovered fully from the previous fire (Zedler et al. 1983; Keeley et al. 2005b; Keeley and Brennan 2012; Lippitt et al. 2013). A single short interval fire may not be enough to cause complete loss of chaparral in most areas (S. Ma unpublished data; Meng et al. 2014), but species composition may be changed if obligate seeders are reduced or eliminated (Zedler et al. 1983; Keeley and Brennan 2012). Chaparral stands are frequently colonized by non-native grasses and forbs 2–5 years after fire if they are located near a seed source (Keeley et al. 2005a). If a fire occurs again during this period, the lower fire intensity allows more non-native seeds to survive, and non-native species cover is higher during the next recovery period (Beyers et al. 1998; Keeley and Brennan 2012). High N deposition may exacerbate this situation (Allen et al. 1998; Padgett and Allen 1999; Fenn et al. 2010, see Sect. 13.4.1.3), increasing the potential for a grass-fire cycle.

Non-native grasses are strong competitors with seedlings of chaparral shrubs, as was demonstrated in early studies by the inability of obligate seeder shrubs to establish in burn sites heavily seeded with non-native annual grass (Schultz et al. 1955). For decades, non-native grasses were seeded postfire in an effort to reduce immediate postfire erosion (Robichaud et al. 2000, see Chap. 7). These seeding projects were seldom successful in reducing erosion, but high grass densities displaced native herbaceous fire-followers and reduced the density of native shrub seedlings (Beyers 2004). Postfire grass seeding is seldom done in southern California today (Wohlgemuth et al. 2009), but a lesson from those studies is that non-native grasses can interfere with successful restoration (Roy 2009; Engel et al. in prep; VinZant in prep-b).

In contrast to non-native grass invasion, tree invasion occurs at the chaparral-woodland ecotone in the mesic ranges of chaparral under long-term fire suppression (Vasey et al. 2012). Coast live oak (*Quercus agrifolia*) invades coastal chaparral unless there has been a fire within the past 70 years (Callaway and Davis 1993; Van Dyke et al. 2001). Encroachment of Douglas fir (*Pseudotsuga menziesii*) into chaparral has been noted in Santa Clara County (Greenlee et al. 1983) and Marin County (Sparling 1994; Dunne and Parker 1999; Horton et al. 1999). Fire suppression will continue where chaparral abuts human development, so if a restoration goal is the maintenance of diverse chaparral, prescribed fire may be needed to reverse tree encroachment. Conversely, invasion by coast live oak (*Q. agrifolia*) may have a negative feedback on fire regimes because oaks are less flammable than shrubs, and this may be a desired outcome near homes.

Fig. 13.2 Left photo: heavily deer-grazed scrub oak (*Quercus berberidifolia*) regenerating from salvaged root material in revegetation area. Right photo: oak sprouts 5 months after fencing (with meter stick in foreground). Green plant to right of meter stick is deerweed (*Acmispon glaber*) that was ungrazed. Photos by C. D'Antonio

13.4.1.5 Human Activities and Domestic Animals

Some activities of humans and their domestic animals can degrade chaparral and pose challenges to restoration. Humans, vehicles, and livestock can directly destroy plantings, disturb soil, and spread seed of non-native annuals into restoration sites. Domestic and feral cats are well known for decimating populations of native animals in many parts of the world (e.g., Marzluff and Ewing 2001; Medina et al. 2011). Studies in southern California suggest that restoration of native birds in shrubland near urban areas may only succeed where restored patches attract top predators like coyotes or otherwise discourage the presence of the meso-predators (domestic cats, raccoons and opossums) that reduce bird populations (Soulé et al. 1988; Crooks and Soulé 1999; Galatowitsch 2012). Human welfare, however, may conflict with attempts to encourage substantial populations of some predators near human habitation.

13.4.1.6 Native Animals

Although native animals are integral parts of chaparral ecosystems, their actions can benefit or slow restoration efforts. Granivory and herbivory can deplete seeds and damage seedlings and resprouting shrubs (Fig. 13.2). The magnitude of these processes varies with site degradation, fire frequency, animal composition, fluctuations in animal populations (van Mantgem et al. 2015), distance to cover for herbivores (site size), and the balance between direct negative impacts of these animals on native plants and indirect positive effects of herbivory on competing, non-native vegetation.

Grazing and browsing by native animals can create bare zones at the edges of shrub stands (Bartholomew 1970) and reduce herbaceous vegetation in the understory (Swank and Oechel 1991; Tyler 1995). Such herbivory may benefit restoration by reducing competition from non-native annual species and by preventing encroachment of perennial non-native species (Lambrinos 2006). Direct damage to chaparral species, however, may hinder restoration. Flowering of chaparral geophytes is reduced by leaf loss (Borchert and Tyler 2009; Williams and Burck unpublished data). Shrub establishment and survival may be reduced by herbivory, but assessments of its importance in chaparral dynamics, such as recovery after fire, yield conflicting results (e.g., Davis 1967; Bullock 1991; Tyler 1995; Potts et al. 2010; Ramirez et al. 2012). The rapidity with which seedlings become resilient to herbivory varies among species. Seedlings of some species (e.g., scrub oaks [*Quercus berberidifolia*]) can resprout following shoot loss their first year. Others can resprout when they are a bit older, and even seedlings of obligate seeders can survive some level of browsing (e.g., 2 or 3 years postfire, Mills 1986; Tyler and D'Antonio 1995). Even in resprouters, root:shoot allocation shifts that depress root growth following herbivory may render seedlings more susceptible to drought-induced mortality (e.g., McPherson 1993). Thus, the use of native browsers and grazers to control non-native annual vegetation is likely to benefit restoration efforts only if native species are protected from herbivory.

Below-ground browsing (root damage) can be more serious than shoot loss. Burrowing animals, such as Botta's pocket gopher (*Thomomys bottae)*, damage roots and bury seed, presenting obstacles to woody plant establishment in forestry, agriculture, and restoration projects in California (e.g., Crouch 1982; Borchert et al. 1989; Witmer and Engeman 2007; Tyler et al. 2008). Adams et al. (1997), studying native oak plantings in California rangelands, noted that "pocket gophers can present a prolonged threat" to restoration efforts, because saplings remained vulnerable until they were fairly large in diameter. Thus, the presence of pocket gophers and the distribution of site characteristics that discourage them (e.g., rocky soil) should be considered when evaluating the likely success of chaparral restoration efforts.

13.4.2 Seed Limitation

Seed limitation is broadly acknowledged to be highly important in influencing community development and critical to passive habitat restoration, however little work has been done on seeding as an active restoration strategy in chaparral. Seed limitation can be extreme when no propagules are available or moderate when safe sites (sensu Harper 1977) outnumber available seed. Here we focus on broad considerations related to the seedbank, seed predation, seed availability, and germination cues.

Obligate seeders have a persistent seedbank composed of refractory (with high dormancy) seeds and are stimulated to germinate based on a range of cues typically related to fire (Parker and Kelly 1989; Keeley 1991, see Chap. 2). Because the adult plants are destroyed by fire, these species are dependent upon a robust seedbank for persistence (Zammit and Zedler 1988; Keeley 1991). Seed longevity in these species is considered to be great (Parker and Kelly 1989), yet finding commercially available seed or collecting it for restoration is difficult because of the generally low seed output and specific germination requirements (Emery 1988). Obligate resprouters often have a transient, or non-refractory, seedbank composed of seeds that are generally killed by fire and depend on animal or wind dispersal during fire-free intervals. Thus seed availability, ease of germination, seeding rate necessary for establishment of adults, and safe sites for establishment may be very different for these broad functional shrub groups, and restoration methods will vary as a result.

Seedbanks of obligate seeders can be depleted through a number of means. Repeated short-interval fires can stimulate germination but not replenish the seedbank if a second fire occurs before plants have reached reproductive maturity (Keeley and Brennan 2012). While individual plants may mature in less than 10 years (Zammit and Zedler 1993), some communities may require multiple decades to regenerate a seedbank large enough to maintain the population (Odion and Tyler 2002). Intense seed predation by some rodents (Keeley and Hayes 1976; Parker and Kelly 1989; Deveny and Fox 2006) can also deplete the seedbank. Non-native annual grasses and forbs can support granivores (Orrock and Witter 2010), and so as non-native annuals invade or dominate degraded chaparral, granivores may reduce the remaining seedbank, further restricting the likelihood of passive recovery. During active restoration, the control of seed predators may be important in the initial stages.

Finally, seed limitation in refractory seeds can arise if seeds are present in the seedbank but are in secondary dormancy and appropriate cues are not experienced (e.g., heat, charate or smoke compounds, period of stratification). Seed limitation for non-refractory species could arise if the degraded area is large and seed sources for re-colonization or seed collection are lacking. Active restoration might then focus on direct seeding with exposure to appropriate germination cues, planting seedlings, or planting nurse plants that might attract animal dispersers of the desired seeds.

Even if seed is available, getting the right germination cues can be challenging. Many chaparral species germinate following a fire cue, generally heat or smoke (Keeley 1991). For this reason, seed treatments that mimic fire are successful in inducing germination in the absence of fire (Wilkin et al. 2013). Heat treatment by placing seeds in boiling water or an oven have successfully increased germination of obligate seeding species of *Ceanothus* (Quick 1935, 1959; Radwan and Crouch 1977) and facultative seeders such as sugar bush (*Rhus ovata*) (Stone and Juhren 1951) and chamise (*Adenostoma fasciculatum*) (Christensen and Muller 1975). Other sources suggest additional dormancy-breaking techniques, such as acid treatment for *A. fasciculatum* (Emery 1988), illustrating the complexity of selecting germination treatments. Perhaps not surprisingly, use on a scale large enough for restoration has been limited. Charate (charred wood) can also induce germination in obligate seeders, such as some *Arctostaphylos* species, and resprouters, such as *Adenostoma fasciculatum* (Keeley 1984, 1991), and could be artificially applied, or charred wood of cleared shrubs could be raked into the soil of small sites to aid germination. Smoke can also induce germination of obligate seeding shrub and herbaceous species (Keeley and Bond 1997; Keeley and Fotheringham 1998). Scarification is another useful technique to induce seed germination. Hadley (1961) found scarifying bigpod ceanothus (*C. megacarpus*, obligate seeder) seeds increased germination almost fourfold compared to a heat treatment. Stone and Juhren (1951) found *R. ovata* (resprouter) had the highest germination rate when the seed coat was removed. Germination rates can further be improved upon by combining treatments, for example scarification and heat. However, best practices seem to be species-specific (Keeley 1991) and can even vary within a species (Stone and Juhren 1951). The application of these laboratory studies to field restoration, particularly on a large scale, has rarely been done (but see Sect. 13.5.2.2).

13.5 How to Restore Chaparral

There are a limited number of available studies on chaparral that document both passive and active forms of restoration. Passive and active restoration fall along a gradient of treatments depending on the extent of damage and residual propagule bank. Passive restoration yields a community of species that remain on site post-disturbance and species that colonize naturally. Choosing elements to restore is largely a process involved in active restoration. A mixed approach that involves passive restoration of some species and active restoration of others (those that do not readily establish or colonize naturally) may be appropriate for achieving specific restoration goals. Passive restoration techniques emphasize recovery that does not include replanting or other intervention, while active restoration emphasizes fire management, revegetation, and soil treatments.

13.5.1 Passive Restoration

Passive recovery from the seedbank has been evaluated after various disturbances to chaparral, including grazing and physical disturbance to soils, and studies of these disturbances indicate that the rate and pathway of natural recovery of chaparral is generally slow compared to patterns of postfire recovery (Keeley et al. 2006, see Chap. 1). For example, succession on former chaparral land in abandoned agricultural and urban construction soils resulted in alternative trajectories dominated by sage scrub species, such as *Eriogonum fasciculatum* and broom baccharis (*Baccharis sarothroides*) with an understory of non-native annuals in sites up to 70 years post-disturbance (Stylinski and Allen 1999; Stratton 2005, 2009). Deliberate type-conversions in the San Gabriel Mountains in the 1960s were

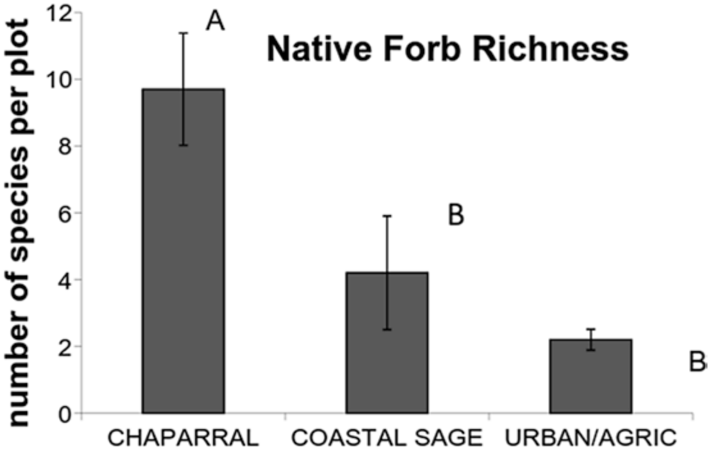

Fig. 13.3 Richness of naturally colonizing (not seeded) native forbs was highest on revegetated roadsides of Interstate 15, San Diego County, when the adjacent vegetation was chaparral rather than coastal sage scrub or urban/agricultural lands (Allen and Heindl 1993, unpublished). The cover of colonizing forbs ranged from 2% to 7% and the cover of planted shrubs averaged 47% across all sites, but no shrub species colonized that were not in the seed mix

invaded by similar species over the following decades. After a fire re-burned the sites in 2002, some chaparral species colonized but at lower abundance than in adjacent undisturbed chaparral (Hubbert et al. 2012b; Corcoran et al. in prep). Removal of domestic grazing animals enabled a degree of natural recovery of chaparral and sage scrub on Santa Catalina and Santa Cruz Islands, but active restoration was needed in some areas to control weeds, reduce erosion, and re-introduce rare species (O'Malley 1991; Halvorson 1994; Beltran et al. 2014). Abandoned roads are also challenging to restore as they have depauperate seedbanks and would need active restoration to re-introduce missing species (Holl et al. 2000). Roadsides actively revegetated with native shrubs were naturally colonized by as many as 10 species of unseeded native forbs, but no species of unseeded shrubs (Allen and Heindl 1993). Along roadsides the highest rate of re-colonization was adjacent to undisturbed chaparral compared to weedy sage scrub or urban development (Fig. 13.3, Allen and Heindl 1993). Where topsoil has been removed or altered, active restoration is usually needed to re-introduce most of the dominant species, otherwise the result could be a novel ecosystem with unusual combinations of native and non-native species (Stylinski and Allen 1999). While such a community may not meet biological diversity goals, it might suffice for other ecosystem services such as soil stability.

13.5.2 Active Restoration

The decision to engage in active restoration typically occurs when soils have been disturbed sufficiently to cause erosion and deplete soil nutrients, microorganisms, and plant propagules, or when vegetation type-conversion results in an alternative stable state. Such sites will not recover quickly enough to meet societal or agency goals for ecosystem services, or will not recover at all without assistance. In this section we review active treatments to promote recovery.

13.5.2.1 Managing Fire Frequency

Altered fire frequencies (see Sect. 13.4.1.4) may present the greatest challenge to land managers intent on restoring the ecosystem services of chaparral. Where shortened fire-return intervals have led to non-native grass invasion, a return to normal fire regimes would allow chaparral to develop a dense and less invasible canopy and also burn sufficiently hot to kill a higher proportion of the non-native seedbank (Keeley and Brennan 2012). Human-related ignitions are responsible for most of the fires in southern California, which usually threaten human developments (see Chap. 12). Use of deliberate type-conversions, fuelbreaks, and fuel modification zones (e.g., mastication of most shrubs with a scattered few left standing) between homes and wildlands may help to protect public safety and may also reduce fire spread from homes to nearby chaparral (Conard and Weise 1998). However, mastication treatments provide only temporary fuel reduction (Keeley et al. 2014; Brennan and Keeley 2015), and non-native herbaceous species may colonize shrub inter-spaces and provide flashy fuel cover (Potts and Stephens 2009). Fuelbreaks require maintenance by animal (grazing) or mechanical means. These vegetation sacrifice areas obviously do not provide the ecosystem services of chaparral but may allow chaparral to persist beyond them. Where chaparral has already been degraded by too-frequent fire, revegetation of areas beyond fuelbreaks may be needed if the return of chaparral ecosystem services is desired (see Sect. 13.5.2.3).

Periodic prescribed fire would prevent ingrowth of trees into fire-suppressed chaparral stands, but air quality concerns (Ahuja 2006) and caution regarding the chance for escapes may limit the use of fire as a vegetation management tool at the wildland-urban interface (Husari et al. 2006). When prescribed fire can be used, the operational window is usually well outside the natural fire season (e.g., during cooler and wetter periods when fire can be more easily be controlled, Green 1981). The resulting fire intensity and effects may be very different from those during the natural summer fire season: burning could be less complete and soils are often wet rather than dry, resulting in less soil heating. As a result, there might be low germination of heat-requiring seeds, such as many *Ceanothus* species (Kelly and Parker 1984; Parker 1990). In southern California these effects may not be extensive (Beyers and Wakeman 2000), but studies in northern California chaparral have generated greater concern (Parker 1987). Implementing prescribed fire in small stands of rare chaparral species, such as that described by Van Dyke et al. (2001), could be difficult. Trees can be removed mechanically to reduce shading in critical situations, but other beneficial effects of time-appropriate fire are hard to simulate.

13.5.2.2 Soil Treatments

Erosion Control

Stabilizing steep hillsides is challenging once shrubs have been completely removed by disturbance and may require an ecological engineering approach that combines surface soil stabilization treatments (erosion control mats, erosion barriers, channel stabilization, re-contouring) with seeding and planting. However, most of the information about the success of these treatments is anecdotal. Erosion barriers or mulch may be applied on small scales to protect urban areas or sensitive species habitat. For instance, mulch increased acorn germination for *Quercus berberidifolia* establishment in the Channel Islands (Stratton 2005). Anecdotal observations after broadcast seeding in utility right of way corridors in the Angeles National Forest suggested that hydromulching was comparable to imprinting (machine-made patterned depressions) for increasing shrub establishment from seed, although both produced only 5%–15% vegetation cover after 1 year (VinZant in prep-b). Effects of hydromulch on erosion and runoff were not measured in these corridors. First year hillslope sediment movement was substantially lower on hydromulched plots than control plots at two burned chaparral sites near Santa Barbara, California (Wohlgemuth et al. 2011). In another study, benefits of hydromulch on erosion control were limited to the first 2–4 months postfire in San Diego County (Hubbert et al. 2012a).

Soil Inoculum

Other soil treatments include use of mycorrhizal inoculum, topsoil, and fertilizer. Although most chaparral species are arbuscular mycorrhizal (AM), the below-ground mycorrhizal community is complex because it includes ectomycorrhizal (EM) oaks and pines, arbutoid mycorrhizal *Arctostaphylos* and other ericaceous species, and *Adenostoma fasciculatum* that can switch between EM or AM under wet or dry conditions, respectively (Allen 1991; Allen et al. 1999). Duff and soil from native oaks improved germination and establishment of acorns because it introduced EM inoculum, while uninoculated nursery transplants of arbutoid Santa Catalina island manzanita (*Arctostaphylos catalinae*) had low survival in former cropland (Stratton 2005). Alternatively, use of fresh chaparral soil added to planting holes for nursery stock of AM shrubs had no effect on their growth (Stratton 2005; VinZant in prep-b). AM shrubs are generalists in terms of their AM fungal associates, and when planted into soils dominated by AM grasses appropriate inoculum is likely to occur in the soil (Nelson and Allen 1993). By contrast, EM oaks and arbutoid manzanitas require their own host-specific inoculum.

Studies from other ecosystems have shown that non-native plant species may cause major shifts in soil microbial species composition and in nutrient cycling (Ehrenfeld 2003; Owen et al. 2013), but the non-native grasses of chaparral had weak feedbacks as there were few differences in microbial communities or nutrient cycling between chaparral and adjacent invaded soils (Dickens and Allen 2014). This suggests that the soil microbial community of Mediterranean-type climate annual grass invaded chaparral will seldom be limiting to plant growth, except in the case of EM or arbutoid fungi that are absent from grassland. In a meta-analysis of 22 restoration studies (none from chaparral), local AM or EM inoculum promoted greater biomass, while commercial inoculum did not (Maltz and Treseder 2015). Some AM plants are highly responsive to inoculation and perform better with inoculation throughout their lifespan. However, mycorrhizal responsiveness of chaparral plants is sparsely studied. Mycorrhizal plants of seasonally dry climates are better able to access water and survive drought (Querejeta et al. 2007), so inoculation is recommended before they are transplanted to the field.

Topsoiling

Where topsoil has been removed for construction purposes, replacing topsoil is recommended, as was tested in a utility corridors in Santa Barbara County (D'Antonio and Howald 1990) and in the Angeles National Forest (VinZant in prep-b). In the latter study, observations indicate that seeded plants had improved growth on topsoil compared to subsoil. Topsoil is seldom available for spreading across large areas, so VinZant (in prep-b) recommends that construction plans include salvaging and re-spreading topsoil. D'Antonio and Howald (1990) documented abundant regeneration of maritime chaparral forbs and early successional shrubs in topsoil that was replaced or conserved (Fig. 13.4). However, soils with a dense non-native seedbank should be avoided for salvaging or solarized (heat-treated by covering with clear or black plastic) during storage to minimize the need for post-restoration weed control. Studies of topsoil handling in *Banksia* woodland restoration in the Mediterranean-type climate region of Australia have revealed that dry-season handling of topsoil yielded much better results than wet-season handling, and that stockpiling soil for even 1 year reduced effectiveness of top-soil replacement. The most effective treatments in the study involved moving topsoil from newly disturbed areas to areas undergoing restoration (Rokich et al. 2000). Similar principles may apply to topsoil handling in chaparral restoration.

Fertilizer

If topsoil is not available to restore severely disturbed sites, soil amendments such as fertilizers and mulch may be beneficial, in addition to mycorrhizal inoculation. An important caveat is that chaparral soils tend to be fairly low in nutrients and

Fig. 13.4 Vegetative cover in second growing season of pipeline revegetation project. Topsoil was conserved during construction and replaced and many unseeded natives recovered from the seedbank. Most species visible here are short-lived perennials or sage scrub species. Surrounding vegetation is 50–60-year old chaparral on sandy soils (D'Antonio and Howald 1990). Burton Mesa region, Santa Barbara County. Photo by Carla D'Antonio

organic matter (Parker et al. 2016), and there is a danger of over-fertilizing and promoting weeds, if careful prescriptions for fertilizer rates are not followed. Non-native grasses have higher growth rates in response to nitrogen fertilizer and any level that is added may promote non-native over native species (Allen et al. 1998). Even in the absence of invasion, elevated soil nutrients will shift native species composition to dominance by species with plastic responses to fertilization, and gradually exclude slower-growing chaparral shrubs (Pasquini and Vourlitis 2010). This suggests that sites with natural topsoil are unlikely to need fertilization, and sites where topsoil is absent may only benefit from fertilizer addition to levels no higher than native soils based on an understanding of plant nutrient needs.

Charate

A soil treatment that could be considered in some settings to benefit plant establishment is burning piles of woody material over the restoration site, especially where a native seedbank is believed to persist. The resulting charred material is then raked over the disturbed area to replace charate that would be released from a natural fire. Such a treatment was used in the Purissima Hills in Santa Barbara County in a pipeline corridor through chaparral and Bishop pine (*Pinus muricata*) stands (D'Antonio and Howald 1990). Species that had not been seeded re-generated in the charate-influenced areas (Fig. 13.5).

Fig. 13.5 Thickleaf yerba santa (*Eriodictyon crassifolium*) seedling growing out of soil in experimental burn pile/charate spread area during second growing season in a pipeline corridor. Photo by Carla D'Antonio

13.5.2.3 Revegetation Treatments

Weeding

Control of non-native species may be sufficient to enable chaparral shrub recovery in invaded areas that have a native propagule bank. Grass-specific herbicide was used to control bulbous canarygrass (*Phalaris aquatica*) on Santa Catalina Island and enabled establishment of chaparral species from the seedbank (Stratton 2005). Weeding accompanied by seeding and planting is the more frequent approach, as heavily grass-invaded chaparral often has a compromised seedbank. For instance, no native shrub seedlings emerged at all from a weeded, formerly grass dominated site in Riverside County (Engel et al. in prep). A summary of weed control activities in invaded utility corridors of the Angeles National Forest included hand weeding and mechanical and herbicide treatments. Although data were not collected from these sites and no unweeded controls were maintained, managers interviewed concluded that weeding was essential for the establishment of seeded shrub species (VinZant in prep-b). Controlled studies showed that seeded black sage (*Salvia mellifera*) and *Eriogonum fasciculatum* (common to both sage scrub and chaparral) had high germination rates and required no seed dormancy-breaking treatment, but only survived in plots hand-weeded of annual grasses, otherwise there was

100% mortality during summer (Schultz 1996). A review of studies on controlling non-native annuals common to chaparral and sage scrub also demonstrated the importance of reduced competition for sage scrub establishment (Allen et al. in prep).

Seeding

Seeding is the preferred method of active restoration because it is less expensive than nursery transplants, but the practice has had limited success in chaparral. Germination rates are notoriously low for many chaparral species, in part because of their specialized dormancy-breaking requirements (see Sect. 13.4.2) and often low percentages of viable seed from wild collections (Wall and Macdonald 2009). Seed application of *Adenostoma fasciculatum, Quercus berberidifolia, Rhus ovata*, and skunk bush (*R. aromatica*) was unsuccessful in spite of appropriate seed treatment (liquid smoke, acid scarification, stratification) and non-native grass control by herbicide prior to seeding, most likely due to severe drought during the study (Engel et al. in prep). Chaparral seeding trials on Santa Catalina Island were less successful than native species emerging from the seedbank of abandoned agricultural fields (Stratton 2005), but there was no mention of seed pre-treatment in these studies. Sage scrub species considered to be early successional in chaparral, such as *Eriogonum fasciculatum*, deerweed (*Acmispon glaber*), and *Salvia mellifera*, established successfully from seed in a pipeline corridor revegetation effort in Santa Barbara County, while seeded chaparral species were largely unsuccessful (D'Antonio and Howald 1990). In a successful chaparral seeding effort, Roy (2009) planted smoke-treated *Adenostoma fasciculatum* seeds in an abandoned rock quarry soil, and observed high germination rates and adequate survival in weeded plots, but poor survival in plots with competition from native and non-native annuals. Summer irrigation reduced dry-season mortality, but establishment occurred in the absence of irrigation when plots were weeded. Taken together, these studies show that drought and competition from non-native annuals are the greatest limiting factors to establishment of chaparral shrubs from seed. Although seed dormancy-breaking requirements have been tested in the laboratory for a wide range of chaparral species (Keeley and Keeley 1987; Keeley and Fotheringham 1998, see Sect. 13.4.2), we could find few references to their use in restoration projects in California in the scientific literature. By contrast, smoke is routinely used as a pre-treatment for seed germination in restoring areas impacted by mines in Australian shrublands (Roche et al. 1997).

Fig. 13.6 Restored abandoned corridor in chamise chaparral on Del Sur Ridge, Angeles National Forest. Container transplants of California sagebrush (*Artemisia californica*), California buckwheat (*Eriogonum fasciculatum*), black sage (*Salvia mellifera*), and Eastwood's manzanita (*Arctostaphylos glandulosa*) were initially irrigated and protected from deer browsing by netting. Photo is taken 2 years after planting. Photo by Jan Beyers

Seed Collection

The preferred seed collection area should be as local as possible to avoid genetic concerns (Vander Mijnsbrugge et al. 2010), although this may be difficult if the degraded area is large and seed sources are distant. US Forest Service guidelines recommend preferably within 8 km (~5 miles) (at most 16 km [~10 miles]) distance and within 150–300 m (492–984 ft) in elevation (VinZant in prep-a). However, these guidelines may need to be relaxed under warming climate conditions. Adapted seed collection zones could be experimentally tested using transplant gardens. Guidelines for southern California chaparral seed processing after collection are summarized in Wall and Macdonald (2009) and Emery (1988) that lists pre-planting seed treatments for many chaparral species.

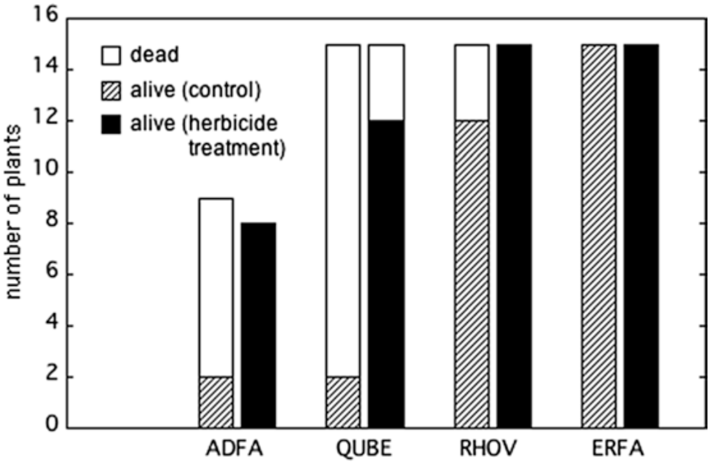

Fig. 13.7 First-year establishment of containerized shrubs was improved for 3 of 4 species (ADFA and QUBE, slight increase for RHOV) by herbicide treatment to control non-native annual grasses and forbs. ADFA = *Adenostoma fasciculatum* (chamise), QUBE = *Quercus berberidifolia* (California scrub oak), RHOV = *Rhus ovata* (sugar bush), ERFA = *Eriogonum fasciculatum* (California buckwheat) (redrawn from Engel et al. in prep)

Nursery Transplants

Chaparral restoration has been most successful from nursery transplants, although this is admittedly a more costly approach than seeding and has only been practiced in small areas (see Chap. 15). Anecdotal reports from utility corridor restoration in the Angeles National Forest indicated that shrub transplants had up to 40% cover the first year, while seeding only resulted in up to 20% (VinZant in prep-b, Fig. 13.6). Some of the successful transplants were hand-watered using deep pipe irrigation that delivers water to the lower rooting zone through a pipe installed at planting (Bainbridge 2007). Deep-rooted (34 cm) transplants of lemonade berry (*Rhus integrifolia*) and laurel sumac (*Malosma laurina*) with consistent irrigation had greater survival than shallow-rooted (10 cm) transplants of the same species (Burkhart 2006). Transplants of 35 shrub species on Santa Catalina Island had marginally greater survival of some species with hand-watering. Interestingly, a water-absorbing polymer decreased survival of some species (Stratton 2005). Stratton (2005) concluded that watering was not required for survival of most of these 35 species in years with average precipitation, but hypothesized that watering would increase the return on investment of transplants of more sensitive species in a drought year. In addition to hand-watering, herbicide treatments reduced annual weed competition and further increased the transplant survival of some shrub species (Engel et al. in prep.), (Fig. 13.7).

In the Mediterranean Basin, transplantation of shrubs was successful if they were pre-conditioned to avoid transplant shock (Vallejo and Alloza 2012), but survival was highly variable depending on the site (Maestre et al. 2006). Polymer gels increased transplant survival in one study (Clemente et al. 2004), and the success of many shrub species was greatest in the presence of nurse plants (Maestre et al. 2001) or tree shelters (Vallejo et al. 2006). Such practices could be implemented in California.

Overall, shrub transplanting is labor intensive. It requires nursery establishment and care for 3–12 months and then watering following transplanting and weeding in the field, in contrast seeded plots are typically only weeded. Nursery transplants are justified where the cost is offset by rapid restoration of ecosystem services, as was demonstrated by reduced erosion from degraded soils in the Mediterranean Basin (Le Houérou 2000; Vallejo et al. 2006). Many chaparral species have not been successfully established from seed in the field, so producing containerized stock from seed or cuttings may be the only option for certain dominant species of chaparral. We are not aware of restoration-scale practical application of transplants from cuttings, but cuttings of some species are routinely taken in botanic gardens. Clean nursery protocols are critically important for restoration projects to avoid pathogen transmission from contaminated nursery transplants (Rooney-Latham et al. 2015). For instance, several species of the pathogenic oomycete *Phytophthora* are spreading into adjacent native chaparral populations from contaminated nursery material (Sims et al. n.d.), a situation that can be avoided by proper sterilization of nursery soils.

13.5.2.4 Establishing Early Successional Species to Initiate Succession

Successional patterns provide information that is useful in ecological restoration (Walker et al. 2007), but this approach to restoring chaparral has not been well explored. Although some obligate resprouters may require the shade of larger shrubs for seedling establishment (Keeley 1992; Pratt et al. 2008), such facilitation plays little role in early stages of postfire chaparral recovery, the context in which chaparral succession is most frequently studied (e.g., Hanes 1971; Kummerow et al. 1985). In restoring chaparral on disturbed or type-converted sites, however, facilitation may play a role. Woody plants that colonize severely disturbed chaparral are often soft-leaved semi-drought deciduous subshrubs of coastal sage (Callaway and Davis 1998; Stylinski and Allen 1999). Where chaparral shrubs have been observed to establish on sites formerly dominated by non-native annuals, they have done so after colonization by soft-leaved shrubs (Callaway and Davis 1993; Brennan 2015). Drought deciduous sage scrub dominants are less likely than evergreen plants to ameliorate the environment of chaparral shrub seedlings during the hot summer drought, and the limited information available suggests that evergreen species may be more effective than drought deciduous species in serving as nurse plants for other woody species (Callaway and D'Antonio 1991). The potential value of using faster-growing evergreen plants to facilitate the establishment of chaparral on disturbed sites requires investigation.

Drought deciduous subshrubs of sage scrub and postburn chaparral (e.g., *Eriogonum fasciculatum*, *Artemisia californica*, and *Salvia mellifera*) are often included in plant palettes used for restoration of disturbed sites (VinZant in prep-a). These species establish more readily and grow more quickly than deeply rooted, slower-growing dominants of chaparral, thereby covering the soil and providing some

Table 13.1 Suggested monitoring criteria over time for chaparral restoration

Criterion	Categories and approaches
Community, measurements:	
Native plant cover (total)	Using transects/plots/cover classes
Native woody plant cover	By life-form, e.g., obligate seeders versus resprouters or sage scrub versus chaparral species
Native woody plant density	By species
Native plant richness	Include all life-forms: forbs, grasses, subshrubs, shrubs
Non-native plant cover (total)	All life-forms in plots or transects
Bird, mammal, herptile usage	Richness, point counts/abundance, nesting, etc.
Invertebrate abundance	Pollinator richness, sweep net counts, pitfall traps
Ecosystem process measurements:	
Net primary production, plant C storage	Biomass accumulation over time, litter accumulation, decomposition rate, % bare soil, NDVI/EVI over time or compared to reference
Soil compaction, nutrients, C storage, microbial functioning	Organic matter accumulation, bulk density, C in profile, microbial biomass
Sediment movement, soil erosion	Soil slips, mudflows, sediment yield
Water yield	Streamflow before/after restoration actions or captured runoff
Nitrate in runoff	In-stream flow before and after restoration actions

degree of soil stabilization more rapidly. Seed mixes for restoration may focus on early seral species that lack the complex germination requirements of many chaparral species (VinZant in prep-a). Whether these species facilitate or retard succession to mature chaparral in restoration settings, however, is not yet known.

13.6 Assessing Restoration Success

The assessment of success in a restoration project depends on the specific goals set during the planning process. These goals should be specific enough to have led to a list of measurable criteria upon which to base decisions for further follow-up actions. Despite the recognized importance of ecosystem processes in most restoration projects, specific measurable criteria for most restoration projects are species based (Ruiz-Jaen and Aide 2005, Table 13.1). They typically include plant cover, richness, and density. Because sites targeted for restoration are likely to be denuded of native vegetation, broad measures of native cover and woody plant density are appropriate. Due to the importance of chaparral for animal diversity, including some estimates of animal presence could be another useful measure of success. Chaparral restoration is frequently oriented to restoration of ecosystem processes such as reducing erosion, so success criteria might also include ecosystem processes such as erosion or nutrient loss (Table 13.1). Plant cover and presence of bare soil can be useful indicators of erosion potential if direct measurements are not made (Robichaud et al. 2000).

A typical restoration project has a reference site against which the measured criteria are compared. Yet in chaparral restoration, the difficulty of restoration toward diverse evergreen sclerophyllous shrubland communities, suggests that success criteria are more likely to be based on broad measurements of native shrub cover and ecosystem functioning rather than comparison to diversity of a reference community. For monitoring ecosystem function, measured outcomes may be compared against both intact reference sites and pre-restoration values using a before/after type statistical design. If restoration sites are not converging toward reference sites or showing directional improvement over time, a need for further restoration planning and actions would be indicated.

13.6.1 Prospects for Long-Term Restoration Success

Active chaparral restoration is too recent and limited in extent to make definitive conclusions about prospects for success on scales larger than road beds or construction pads. The easiest species to establish from seed are herbaceous plants or those shrubs that are considered early successional or are common within sage scrub, such as *Eriogonum fasciculatum*, *Salvia* spp., *Acmispon glaber*, or *Artemisia californica*, whereas true chaparral shrubs are more easily established from container-grown plants (VinZant in prep-b). Most chaparral shrub seeds require pretreatment of some kind to germinate, which is easiest to do in small batches used for propagating nursery stock. In the absence of heat or smoke treatment, only species without a fire-related cue for germination are easy to seed in large quantities or likely to passively colonize restoration sites. In small-scale projects, however, plant cover was most quickly established on restoration sites using chaparral container stock and consistent irrigation (Stratton 2005; VinZant in prep-b).

Sage scrub species have been found to establish from seed without irrigation in years of average precipitation in sage scrub restoration sites (Padgett and Allen 1999), a precipitation regime that is below that for the more mesic regime of chaparral. This suggests they would be suited to chaparral restoration sites in drought years, when true chaparral species would be more difficult to establish, or at the lower elevation limits of chaparral. The decision to replace one vegetation type (chaparral) with another (sage scrub) cannot be made lightly and has become a major debate among conservation and restoration ecologists struggling to preserve species under climate change (Seddon 2010). However, sage scrub species colonize naturally into severely disturbed soil, ecotones with chaparral, and deliberately type-converted areas. They are faster-growing than chaparral shrubs, more drought tolerant, and thus more resilient under frequent fire. The transition from chaparral to sage scrub already occurs in some areas and provides land managers with a suite of species that may perform some desired ecosystem services.

It is unlikely, however, that soft-leaved, deciduous, shallower-rooted species will perform all of the same ecosystem services as hard-leaved, evergreen, deep-rooted chaparral species in the long-term. For instance, sage scrub roots that are 1.5 m (~5 ft) deep should help stabilize surface soils, but they likely will not be as effective at slope stabilization nor sequester as much carbon in the long-term as deep-rooted chaparral shrubs. The comparative roles of sage scrub and chaparral in carbon sequestration is a topic in need of further study. Sage scrub vegetation also does not provide the same habitat structure for animals as true chaparral. For conservation reserves there may be aesthetic issues and diversity shifts in selecting a suite of sage scrub species where chaparral once occurred, but these species may facilitate the establishment of chaparral species (see Sect. 13.5.2.4). Where rapid and more reliable establishment of chaparral is a project goal, the use of nursery transplants, protection from herbivory, weeding, irrigation and soil treatments may be needed in the most severely disturbed sites. Across large public landscapes, trade-offs between the costs of restoration and ecosystem services will need to be assessed. Clearly, additional research on incorporating large scale seed dormancy-breaking treatments, facilitation, and understanding long-term successional processes for chaparral restoration is critically needed.

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