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U.S. Department of Agriculture, Forest Service
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Abstract

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The chaparral restoration workshop held in Arcadia, CA, in 2013 brought together experts, managers and others to discuss alternatives that more effectively protect, manage and even revegetate diminishing shrublands. This publication is a compilation of some of the workshop and oral presentations that update our current knowledge of shrubland ecology and management with respect to its loss and recovery. Shrubland species have evolved strategies that allow them to endure Mediterranean climate extremes. Combinations of often complex factors allow these plants to survive unpredictable and harsh conditions. Determining which elements and conditions are critical for shrubland persistence can provide a framework for re-establishment and management of this resource. Bringing together researchers, managers and advocates who are working on restoring and sustaining chaparral and other shrublands has stimulated exchanges that may lead to improved methods for conserving shrub dominate communities.

Keywords: Chaparral, coastal sage scrub, native plant restoration, shrubland.

Preface

Shrubland use and management has evolved over time and place (Bentley 1967, Cable 1975, Chapline 1919, Green 1981, Myers et al. 2000, Moritz 2003, Rothermel and Philpot 1973, Sidahmed et al. 1981, Wilkin et al. 2013). Periodic reviews of shrubland science, management and policies have documented and addressed changing needs, conditions and knowledge advancements in shrubland communities (Biswell 1974, 1977; Brooks et al. 2004, Conard and Weise 1998, Dunn 1989, Keeley 1989, Lotan 1981, Radtke 1983, Stratton 2004, Tyrrel 1981).

Chaparral and other shrublands have long been appreciated for their resiliency and role in maintaining watershed stability (Bennett and Chapline 1928, Hanes 1971, Heede 1988, Hellmers et al. 1955, Rice and Foggin 1971). Historically, natural regeneration was commonplace and shrub extirpation from disturbance did not typically occur--even if desired. Therefore, management for shrubland post-disturbance recovery usually was not a concern. This is still generally the case for many shrubland locations. However, some areas are now exceptions, particularly in southern California chaparral. Here, both hard and soft chaparral disappeared from previously occupied sites and landscape level efforts were undertaken to reestablish a lost shrub community (Landis 2000, Serrill 2006, Stylinski and Allen 1999).

Shrub loss particularly on steep terrain can cause slope destabilization and resultant debris flows (Hellmers et al. 1955, Rice and Foggin 1971, Sampson 1944,). Shrub loss on the lower foothills can lead to a decline in biodiversity (Duran 2008, Keeley 2005). Conflicting opinions exist on what is the most suitable management scenario for shrublands. For example, mature chaparral is typically considered a highly flammable vegetation type that could be removed for wildland fire hazard reduction--particularly near urban areas (Leisz and Wilson 1980). Conversely, Walter et al. (2005) maintain that ancient chaparral habitat is unique, highly productive and should be protected.

In southern California, soft and hard chaparral shrublands were converted to farms, ranches and urban development during the massive human population surges of the late 20th and early 21st centuries (Calfacts 2000, U.S. Census Bureau 2017). Chaparral has been scarred for fire safety with fuel breaks, deliberately removed for fire hazard reduction (Radtke 1983) and has suffered additional loss from frequent anthropogenic wildfire (Keeley et al. 1999). The wildland urban interface expansion continues to fragment this natural landscape.

Disturbance, from frequent fires, often leads to type conversion with invasive herbaceous plants dominating many historic shrubland sites (Fabritius and Davis 2000, Haidinger and Keeley 1993, Minnich and Dezzani 1998). Shrubland type-conversion has led to increased fire frequency due to the greater ignitability of invasive flashy fuels. This increases erosion from site instability as deep rooted shrubs disappear (Heede 1988, Hibbert 1971, Ingebo 1971, Rice and Foggin 1971).

Changes in fire regime are often accompanied by loss of species diversity especially sensitive flora and fauna (Keeley 2005, Skinner and Pavlic 1994). Deliberate or accidental chaparral conversion to invasive herbaceous species may reduce quality and quantity of ecosystem services such as wildlife habitat or watershed protection.

A growing awareness of valuable ecosystem services that chaparral and other shrublands provide underlines how important it is to properly manage this ecosystem. Chaparral communities provide important ecosystem services (Sampson 1944, Meixner and Wohlgemuth 2003) and contribute to the high biodiversity found in California's floristic province (Myers et al. 2000). We now know that once gone, a generally resilient chaparral shrubland can be difficult to re-establish (Serrill 2006, Stratton 2004, Stylinski and Allen 1999). Fortunately, horticulturalists have developed techniques for nursery propagation and out-planting of many chaparral species (Evans 2000), but success varies. Additionally, revegetation efforts commonly have focused on sensitive species at risk (Hillyard and Black 1987).

Historical context, perspective and reflection on various shrubland sustainability and restoration attempts illustrate how a paradigm shift in shrubland management is occurring. Focus on single species or site recovery is morphing into grander landscape level recovery efforts incorporating multiple species. Methods are now being developed for re-establishing even common shrub species and envelop extensive landscape level wildland plantings. Emergent shrub community management now includes revegetation using common species (e.g. *Eriogonum fasciculatum, Encelia farinosa*, *Adenostoma fasciculatum*, etc.) to resurrect damaged or missing shrubland components (Wilkin et al. 2013).

The work presented here documents some emerging knowledge on the forefront of shrubland restoration practices. Mediterranean ecosystem plants have survival tactics that reveal evolutionary adaptations appropriate for historic drought and possible global climate change extremes. These include but are not limited to variability in: reproductive strategies such as seeders and sprouters, summer drought dormancy, and shallow and/or deep roots. Untangling critical lifecycle elements at a species as well as landscape level and transferring these into useful applications are a prerequisite to improve management and restorations for these unique communities (Bowler 1990, Cerdà and Robichaud 2009, Stylinski and Allen 1999).

Shifting the management paradigm from historical shrubland removal to current landscape scale maintenance and revegetation proffers new viewpoints with expanded possibilities. This paradigm shift encompasses broad scale shrub community restoration efforts using multi-species combinations that intermix planting seeds and started plants (Serill 2006, Stratton 2004). If fully embraced and implemented, an adjustment in perspective may secure improved environmental quality to a vast expanse of the American West and other shrub dominated ecosystems worldwide.

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Contents

Long-Term Prospects for Restoration of Coastal Sage Scrub: Invasive Species, Nitrogen Deposition, and Novel Ecosystems[1]

Edith B. Allen[2], Christopher McDonald[3], and Bridget E. Hilbig[4]

Abstract

Coastal sage scrub (CSS) is one of the most endangered ecosystem-types in California and is undergoing extensive restoration efforts. Major threats to CSS include agriculture and urban development, fragmentation, invasive species, frequent fire, and high levels of anthropogenic nitrogen (N) deposition that increases exotic species productivity, further increasing fuel for fire. In this review we compare a range of techniques that have been used with varying success to restore CSS, using examples from published and unpublished sources. Techniques that treat large scales and reduce the exotic seedbank are the most effective, such as herbicides or solarization, but each may also have drawbacks. Other methods include mulch, seeding or planting species with competitive functional traits, grazing, and fire. Regardless of method, invasive species recolonize to varying extents following restoration, and periodic treatment is needed. CSS in sites receiving more than the critical load of 11 kg ha^{-1} yr^{-1} of N deposition may become type-converted to exotic annual grassland in the absence of other disturbance such as fire. Such sites are not good candidates for restoration. Inability to control exotic species reinvasion and restore diversity of native forbs results in novel ecosystems with reduced conservation value and ecosystem services.

Keywords: Coastal sage scrub, invasive plant species, fire, functional traits, nitrogen deposition, critical load, restoration, smokewater, solarization

Introduction

Restoration of some ecosystem types has proved quite challenging for various reasons, including the degree of impact they have received or the extent of invasion by exotic plant species (Allen et al. 2000). Ecosystems that cannot be easily restored may have exceeded a threshold and moved into another stable state and will require major effort to return to original condition (Hobbs and Norton 1996). This might be the case where invasive species have formed a dense seedbank or soils have become eutrophied so that native plants adapted to low-nutrient soils cannot compete (Cione et al. 2002, Yoshida and Allen 2004, Rao and Allen 2010, Fenn et al. 2010). The effort required to restore these ecosystems to a predisturbance state may be higher than available resources allow, or in some cases impossible with any level of effort.

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These have been defined as novel ecosystems, those that have no antecedents in any local vegetation types because they have a species composition that is completely or partially different from pre-disturbance communities (Hobbs et al. 2009). Another term, hybrid ecosystems, has been applied when the resultant vegetation is a mix of native and non-local species (Hobbs et al. 2009). When the goal of restoring a pre-disturbance ecosystem cannot be met, managers may have to settle for a hybrid ecosystem that retains important conservation values, such as habitat for rare species, and major ecosystem dynamics, such as an appropriate fire regime, but may include non-local species.

Coastal sage scrub of California is among the most endangered ecosystem types in the United States (Noss et al. 1995) because it is limited in extent and includes many federally listed, state listed and sensitive species. Conservation and protection measures for these species and their habitats have made CSS the object of major restoration research and management efforts (Allen et al. 2000, and studies reviewed below). CSS is also among the most challenging vegetation types to restore. It has been extensively converted to, and fragmented by, urban development and agriculture and subject to other impacts such as historic grazing, invasions by Mediterranean annuals, frequent fire, and high levels of anthropogenic nitrogen (N) deposition (Talluto and Suding 2008, Cox et al. 2014). Domestic grazing declined by the 1930's with increasing urbanization, and sites that were historically grazed were able to recover (Fleming et al. 2009, Cox et al. 2014), but exotic annual invasions continued to expand (Minnich 2008). Nitrogen deposition originates from automobile, industrial, and agricultural emissions, and began to increase in the 1950's. Current levels are above critical load (10 to 11 kg N ha^{-1} yr^{-1}) for 33% of CSS vegetation statewide (Fenn et al. 2010, Cox et al. 2014). By comparison, background N deposition in unpolluted regions is 2 kg N ha^{-1} yr^{-1}. CSS areas above critical load are highly invaded by Mediterranean annuals with a loss of native plant diversity. Some areas of CSS have been entirely converted to exotic annual grassland (Cox et al. 2014), including areas within Forest Service lands and conservation reserves that are being restored, or where restoration planning is under consideration (Western Riverside County Habitation Conservation Plan, n.d.).

The objectives of this review are to discuss techniques that have been used for restoration of invaded CSS, including some previously unpublished studies; to review projects and techniques that have been successful, and discuss factors that contributed to their success; and to help managers make decisions about techniques to use and sites to select to maximize the success of CSS restoration. In many cases exotic annuals cannot be permanently controlled, and managers will need to make decisions about how much effort to put into maintaining the site. In some cases, novel or hybrid ecosystems may be the product of restoration (Hobbs et al. 2009).

Restoration Techniques for CSS

The techniques used for CSS restoration depend on the degree and kind of disturbance. With the implementation of Habitat Conservation Plans in several southern California counties, abandoned agricultural areas, including citrus orchards, small grain fields and pastures, are being converted back into native habitat. These typically have little or no native seedbank, have been extensively invaded by Mediterranean annual grasses and forbs, and require the most intensive efforts to restore (Allen et al. 2005, Marushia and Allen 2011). Sites that have been burned frequently also have a depleted native seedbank (Cione et al. 2002), although their soils have not been disturbed by tillage so soil nutrients and microorganisms are still largely intact (Dickens et al. 2013, Dickens and Allen 2014). In contrast, soils of a former citrus orchard had a legacy of fungal pathogens that selectively reduced growth of native forbs but not exotic grasses (Hilbig and Allen 2015). Sites that receive high levels of N deposition also have high concentrations of soil nitrogen, which promote exotic grass growth to the detriment of native species (Padgett et al. 1999, Yoshida and Allen 2001, 2004, Rao and Allen 2010). Increased grass biomass contributes to frequent and larger fires, further degrading native shrublands and promoting vegetation-type conversion. Each of these disturbance types can be addressed with multiple approaches to restoration.

CSS is one of the most invaded ecosystems in California, so restoration often begins with control of invasive species. Recovery by natural succession may occur (DeSimone 2011), but is infrequent in highly invaded or severely disturbed CSS (Stylinski and Allen 1999). Exotic annual species tend to displace native shrub seedlings and native annuals through competition and high propagule pressure. Propagule density of exotic species can be quite high even in CSS in relatively good condition. In invaded CSS with shrub cover, Cox and Allen (2008a) found 5000 seeds/m^2 of exotic grasses and forbs, 800 seeds/m^2 of native forbs, and only 4 seeds/m^2 of shrub species. Exotic seed density was even higher in former CSS that was converted to exotic grassland, with lower densities of native seed. The exceptions are constructed soils such as cut banks on roadsides or other urban construction sites where topsoil was removed. These soils are initially devoid of any seed, and planted seeds would be able to establish without competition from exotic species. However, in most degraded CSS lands, propagule pressure and the density of the exotic seedbank is a major obstacle to success following seeding. In general CSS shrub seedlings are poor competitors with exotic annuals, while established shrubs and perennial grasses fare better (Eliason and Allen 1997). Roots of exotic annuals tend to be shallow compared to mature CSS plants, but CSS seedlings have shallow roots that reduce their competitive ability (Wood et al. 2006, Eliason and Allen 1997). CSS plants at different life history stages compete with exotic annuals with varying success. Because of this, seedlings of CSS species need reduced competition by creating openings in the exotic annuals. Whether naturally occurring, caused by small mammal disturbances or by land managers, these openings are crucial to establishing native vegetation (DeSimone and Zedler 1999, Moyes et al. 2005). Any restoration of invaded lands would need to begin with controlling existing exotic species and controlling the exotic seed bank. Restoration researchers and practitioners have used a variety of techniques to restore degraded CSS, including fire, solarization, grazing, mowing, herbicides, and followed by seeding where the native seedbank is depleted. Effectiveness and limitations (Table 1) of these techniques are discussed

Fire

Fire is used as a restoration tool only when there are few or no native shrubs remaining, as managers are understandably reluctant to remove existing native vegetation. Fire is effective for controlling exotic annual grasses when it is timed in spring or early summer just before seed drop to reduce the contribution to next year's seed bank (Gillespie and Allen 2004, DiTomaso et al. 2006). This is an effective technique for control of grasses with very short-lived seedbanks (1-2 years) such as red brome (*Bromus rubens*; Salo 2004), but additional control is needed for longer-lived grass seeds such as ripgut brome (*Bromus diandrus*) or longer-lived seeds of forbs such as storksbill or mustard (*Erodium* or *Brassica* spp.). Fire is often best used as part of an integrated pest management (IPM) plan (DiTomaso et al. 2006), particularly because post-fire fuel loads are often not sufficient to carry fires in successive years, unless precipitation is above average for multiple years. Thus, fire will need to be coupled with a second alternative treatment. Wildfires burn through CSS at intervals of 20-40 years (Cleland et al. 2016). When return intervals are sufficiently long, fire by itself is not detrimental to CSS stability, nor does infrequent fire by itself promote type conversion of CSS (Cox et al. 2014.).

Solarization

Solarization is the most effective method for controlling exotic seedbanks, but only on soils where vegetation has been removed for best contact with the plastic cover, such as on abandoned agricultural fields or urban construction sites (Moyes et al. 2005, Marushia and Allen 2011). In a comparison of winter solarization (using black plastic, also called tarping) mowing, grass-specific herbicide, and early season disking, solarization was the most effective technique in reducing the seedbank and promoting establishment of native plants (Marushia and Allen 2011). Summertime application of clear plastic is more effective because soils will heat to high temperatures providing greater weed seed mortality, but only in moist soil (Elmore et al. 1993). Since irrigation water is typically limiting, summer solarization is seldom an option. A comparison of winter and spring solarization (taking advantage of residual soil moisture and the warmer temperatures of spring) showed that spring was more effective in controlling the weed seedbank (Weathers 2013). However, all of these studies were conducted in small research plots. While solarization is common for agricultural seed bank control, few efforts have been made to extend it to large-scale restoration (Stapleton and Jett 2006). Solarization can be costly on a large scale, but the resulting depletion of most of the seed bank can also be highly rewarding (Stapleton 2000).

Grazing

While grazing studies are common in grasslands (e.g. Weiss 1999), no published studies address grazing as a technique for control of exotic grasses in CSS restoration. The removal of intensive grazing enabled CSS recovery on Santa Cruz Island (Yelenik and Levine 2010), but control of exotic grasses often requires limited, timed grazing. One previously unpublished study that addressed effects of controlled grazing for CSS restoration was conducted at the Lake Skinner Western Riverside County Multispecies Habitat Reserve in Lopez Canyon, an area grazed through the late 1980's (Allen unpublished). The site had sparse CSS shrubs (<20% cover), with an understory of red brome, ripgut brome, and filaree as the dominant exotic species, and some 70 native species (mainly annual forbs) per ha. Further background information for this study is in Allen et al. (2005). Twelve 1-ha plots were arranged in three replicate blocks with each of three treatments and a control per block: 1) Fusilade II ® (grass-specific herbicide)—February/March 1999 and 2000, using hand-held applicators at the lowest level of the manufacturer's recommended dose; 2) Dethatching plus herbicide—November 1998 using hand-held

Table 1—Effectiveness and limitations of restoration treatments to control invasive plants and seed with native species. Extent: small 100 m^2 - 1 ha, medium 1-100 ha, large >100 ha

Treatment	Effectiveness	Limitations	Extent	Citations
Fire	Spring burns are effective to control exotic annual seedbank. Typically applied in degraded CSS with few or no shrubs.	Managers are reluctant to burn healthy shrub stands, even with invaded understory. Many CSS shrubs are mainly reseeders with a sparse seedbank. Would need to reseed shrubs. Many CSS shrubs do not tolerate frequent fire (>1/decade).	Large scale	Gillespie & Allen 2006, DiTomaso et al. 2006
Solarization	Highly effective in reducing or eliminating weed seed bank if applied on moist soil with high temperatures and bare soil.	Natural soil moisture is high only in winter rainy season, but temperatures are too cool. Laying down plastic after a spring rain, and allowing it to remain in place until hot summer temperatures, is possible. Used mainly in agriculture, few applications in actual restoration.	Small to medium	Elmore et al. 1993, Marushia and Allen 2011, Moyes et al. 2005, Stapleton 2000, Weathers 2013
Grazing	Controls exotic grass seedbank when properly timed before annual seed production. Shrubs will survive browsing. Useful in rugged or rocky terrain.	Grazers are not selective of exotic vs. native annuals, but native annuals have long-lived seedbank. Timing is difficult in dry years when exotic grasses go to seed early, grazers will not consume sharp seeds. Results vary with livestock species.	Large scale	Weiss 1999, Kimball and Schiffmann 2003, Allen unpubl. (see text)
Mowing	Effective in controlling weed seebank when properly timed before seed production.	Useful in degraded CSS with few or no shrubs on relatively level land.	Medium to large scale	Allen et al. 2005, Marushia and Allen 2011
Hand weeding, line trimmers	Effective for smaller sites needing complete weed control, e.g. to protect listed species or around visitor centers.	Labor intensive, must be applied periodically. Best when volunteers can be assembled.	Small to medium	Allen et al. 2005, Hasselquist et al. 2013, Marushia and Allen 2011
Mulch	Good control when applied before weed seed germinates and when native seed applied after mulching. Also immobilizes excess soil N, reducing competition from nitrophilous weedy species	Mulch adds organic matter to soils that are natually low. Arid land plants and soil microorganisms may not be adapted to elevated soil moisture. May need reapplication if weeds need continual control.	Small to medium	Zink and Allen 1999, Cione et al. 2002, Wolkevich et al. 2009
Herbicide	Effective when used properly, may be selective depending on target species. Often most cost efective weed control option.	Some agencies or managers are reluctant to use herbicides based on the "precautionary principle." Some potentially effective herbicides are not permitted on all potential lands or land uses (i.e. grazing).	Medium to large scale	Allen et al. 2005, Bell et al. 2015, Marushia and Allen 2011, Cione et al. 2002, Holl et al. 2014, Kimball et al. 2014,
Smoke stimulation	Many CSS species germinate without smoke treatment. Wildfires provide natural treatment, or seeds can be treated in the lab or nursery prior to seeding.	No known effective direct field application of smoke or smoke water treatment, other than actual fire.	Small to large	Dixon et al. 1995
Seeding	Essential when seedbank is depleted of native seeds	Seeds may not emerge in first year, or not at all due to drought, herbivory, or unknown causes	Small to large	Dickens and Allen 2014, Marushia and Allen 2011, Cox and Allen 2008
Nursery transplants	Used when a rapid vegetation stand is desired, or to avoid the seedling stage of species that are poor competitors with exotic species.	Expensive, time consuming, weed control surrounding young plants increases survival	Small to medium	Bowler 2000, Eliason and Allen 1997
Functional traits	Native plants with traits that make them more competitive (larger, higher growth rates) or have greater longevity than invasive annuals (shrubs vs. annuals) are better candidates for successful restoration.	A limited pool of native species is available to select from, so not all needed traits that are superior to invasive plants are available. Invasive species may have certain traits such as earlier germination that trump all other native traits.	Small to large	Cleland et al. 1993, Hilbig 2015, Kimball et al. 2014, Talluto et al. 2006

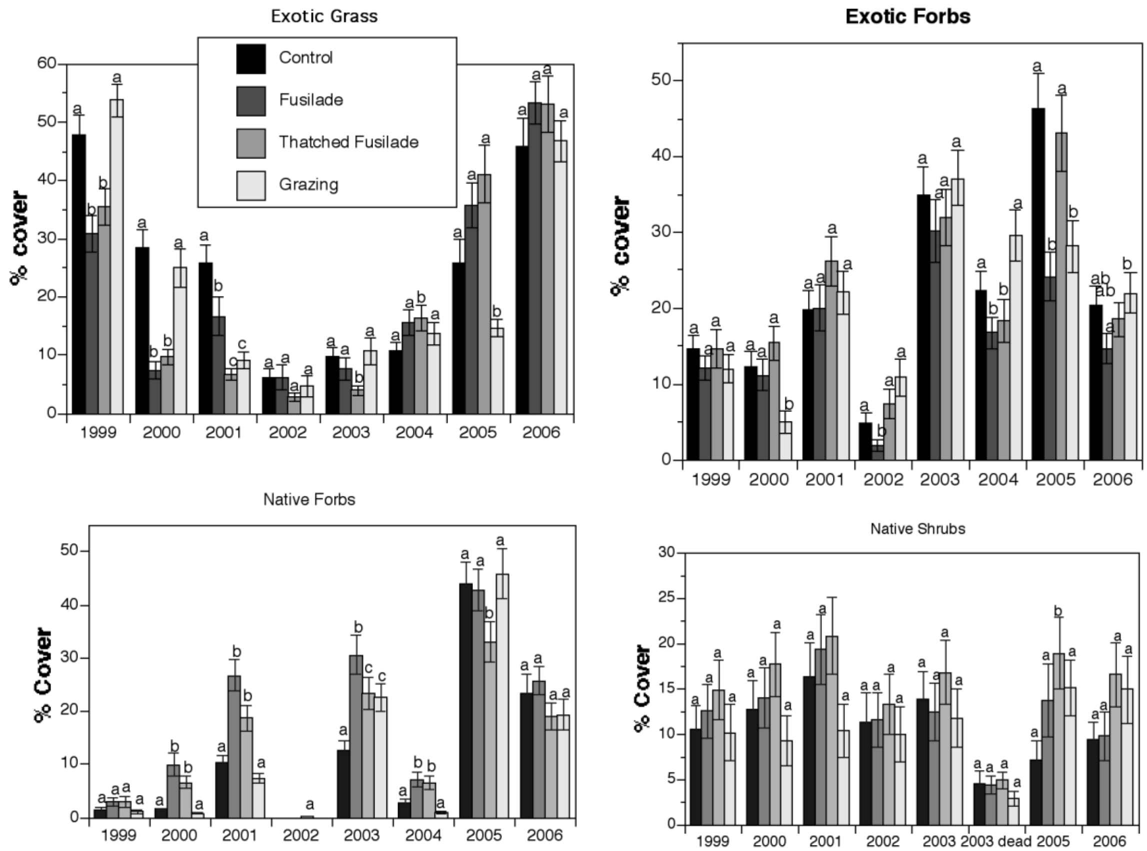

Figure 1—Responses of exotic grass, exotic forbs, native forbs, and native shrubs to three weed control treatments. Letters show significant differences within a year. Native forb responses to treatments were still significant through 2005. Dead shrub cover is shown for 2003, following the 2002 extreme drought year (Allen, unpublished).

weed trimmers, followed by Fusilade II ® in February/March 1999 and 2000; 3) Grazing—March/April in 1999, 2000, and 2001, using 200 sheep per hectare plot for 48 hours; and 4) control. Vegetation percent cover was sampled in 1 m x 0.5 m quadrats, 20 in each 1-ha plot, yearly in late April/early May. Data were analyzed using one-way ANOVA with blocking. Grass-specific herbicide alone and in combination with dethatching significantly reduced grass cover (mainly red brome and ripgut brome) in 1999-2001 compared to controls (fig. 1). During and after the extreme drought year of 2002 (5 cm, average is 26 cm), there was no longer a significant effect of herbicide on grasses. Grazing was not effective until the third year, 2001, when exotic grass in grazed plots was lower than in control plots. Exotic forbs (mainly filaree species) did not respond positively to grass reduction during 1999-2001, and their reduced cover in formerly-herbicide treated plots in 2004-2006 is difficult to explain. Alternatively, native forbs responded to reduced grass cover after herbicide control with increased cover in 2000-2004 and to reduced grass cover from grazing in 2003. The response of native forbs (some 60 species) to grass removal treatments after the 2002 drought is remarkable, suggesting the seedbank may have been affected by the treatments with a carryover to 2003, which had average rainfall. Shrubs were not significantly affected by any treatment. Drought in

2002 caused considerable mortality as measured by dead shrubs in 2003, but they recovered to pre-drought levels.

While grazing was considered an effective strategy for restoration of Central California grasslands (Weiss 1999) that have higher precipitation and productivity, timing was difficult in the drier and variable climate of southern California. Exotic grass productivity is relatively low in most years, and there is only a 2-3 week window for grazing before brome grasses produce seeds. Sheep will not consume grasses with these sharp seeds, and graze native shrubs and forbs instead. Native forbs were also preferentially consumed by cattle in a Carrizo Plain grassland, and grazing was not considered to be an effective restoration treatment (Kimball and Schiffman 2003). In CSS grazing may be effective in a wet year with sufficient exotic grass production for a reasonably long window of time for grazing, but not in a dry year. Short duration timing of grazing is critical even in a wet year to assure that animals consume exotic grasses without excessive trampling of native plants.

Mowing, Weed Trimming, and Hand Weeding

Machine mowing to restore CSS is effective in relatively flat landscapes and that lend themselves to mowing such as abandoned agricultural fields that have no remnant shrubland, but it cannot be used in steep or rocky terrain. Annual mowing has allowed stands of CSS to spread into adjacent abandoned agriculture at the Johnson Ranch in Riverside County and into weedy parkland at Mt. Rubidoux Open Space Park in Riverside City (Cione et al. 2002, Marushia and Allen 2011, Allen, personal observations). Mowing at these sites ceased after shrubs were tall enough to interfere with the mower, at which time they were also tall enough to overtop exotic grasses (e.g., > 50 cm). Weed trimming (weed "whacking" with a line trimmer) is typically used for individual plants, in smaller areas such as research plots in small to medium sized infestations or in terrain inaccessible with a mower, and can be effective if applied annually or more times per season (DeSimone 2007, Marushia and Allen 2011). It has been used to control weeds in the interspaces between CSS shrubs in areas of several hectares, such as at the Western Riverside County Habitat Conservation Reserve, by using teams of workers. Hand weeding is done for small plots around high-visibility areas such as near park visitor centers, to protect sensitive species, or for research studies. Many exotic annual grasses have densities >500/m^2, so this is challenging work but very effective. After the first year the seed bank density begins to decrease, so subsequent years require only light maintenance. The Theodore Payne Foundation has several hectares of hand-weeded CSS maintained by staff and volunteers to provide a public exhibit of native vegetation (theodorepayne.org/visit-us/our-grounds/wildflower-hill/). Hand weeding provided the best exotic grass control to promote recovery of the endangered *Ambrosia pumila*, better than Fusilade II® although not practical on a large scale (Hasselquist et al. 2013). Persistent hand removal provides the best results for weed-free research plots as controls, and has been used in CSS restoration studies (Eliason and Allen 1997, Cox and Allen 2011, Dickens and Allen 2014).

Mulch

Application of a mulch barrier to prevent the establishment of annual plants can be highly effective at restoring CSS. Mulch (wood chips, bark or straw) reduces the ability of annual species to establish allowing for reduced competition when potted perennials are installed (Zink and Allen 1998). Mulch is also used to immobilize excess soil N and improve soil moisture and subsequent seedling establishment and plant growth. In a study to restore CSS, bark mulch was more effective than straw, and both were better than no mulch for reducing soil mineral N to reduce competition from exotic grasses and restore seeded California sagebrush (Zink and Allen 1998). Bark mulch reduced soil N in another CSS restoration seeding study but was ineffective in promoting native shrub growth because it was applied after exotic grasses had germinated (Cione et al. 2002). In a mature, invaded stand of CSS, exotic grass litter addition increased both CSS shrub growth and exotic grass production by improving soil moisture (Wolkevich et al. 2009). Whether mulch is used to immobilize excess soil N or to reduce exotic seed germination by smothering, timing of mulch application before germination is critical. Small native seed may in some cases need to be broadcast after mulch application, but thick layers of mulch may reduce seed contact with soil. In this case, planting nursery stock may be the best option. Mulch application is not practical on a large scale, and is typically used in sensitive areas or near visitor centers. It is not a solution for mitigating landscape-scale N deposition, which will continue to elevate soil N after restoration. Furthermore, CSS soils have low organic matter (typically 1-2%), so adding mulch may artificially alter soil chemistry and moisture (Wolkevich et al. 2009). Additionally, mulch barriers eventually decay or disperse and need to be replenished if long-term control of invasive species is not attained by one application.

Herbicides

Herbicide treatments can be highly effective at reducing weed cover and if applied for several successive years can ultimately eliminate the seed bank of short-lived weed seeds. Few herbicide evaluations have been conducted in CSS on a large-scale or over a long-term, but results from small plots show weed populations can be significantly reduced by single or repeated applications (Allen et al. 2005, Cox and Allen 2008). In general, herbicide treatments will reduce exotics the most when treatments last longer than the longevity of dormant exotic seeds. Many exotic annual species in California have a sufficiently short-lived seed bank where repeated annual applications of herbicide can be effective at weed extirpation. For instance, red brome seed survives 24 months (Jurand et al. 2013, Salo 2004), ripgut brome lasts 2-3 years (Kleemann and Gill 2009), wild oat survives 3-5 years or longer (Conn and Werdin-Pfisterer 2010), short-podded mustard survives 4+ years (Chadoeuf et al. 1998) and tumbleweed 1-3 years (DiTomaso and Healy 2007),. While herbicides, mulching and solarization can provide significant weed control, herbicide treatments are the most cost-effective treatment when compared to mulching or solarization (Holl et al. 2014, Bell et al. 2015). Herbicide or other treatments are used for on-site control, but propagule pressure from surrounding untreated areas is a constant threat to restoration.

Herbicide success will depend on the active ingredient used and thus the herbicide mode of action, the rate at which it is applied, timing of application and efficacy of initial and follow up treatments. In general, the efficacy of using a broad combination of herbicides is well developed in agricultural systems; it is less commonly used in the restoration of wildlands. Future research on a variety of herbicide uses in wildlands could significantly improve CSS restoration and management. For example, in areas where exotic grasses dominate, broad spectrum and grass-specific herbicides as well as pre-emergent herbicides could control annual exotics throughout the growing season. In addition, herbicide combinations could allow for management of exotics while planting nursery stock. Most published studies of CSS restoration strategies have relied on only two herbicides, glyphosate and/or fluazifop (Kimball et al. 2014, Marushia and Allen 2011, Cox and Allen 2008b, Cione et al. 2002) despite a variety of available products on the market.

Smoke-induced Seed Bank Stimulation

While the suppression of invasive plants and weeds is critical to establishment of CSS, there remains much to learn about the stimulation of seeds by smoke, both native and exotic species. Smoke can be used to break dormancy of seeds in Mediterranean-type ecosystems (Dixon et al. 1995, Keeley and Fotheringham 1998, Egerton-Warburton 1998), and is a natural process of seed bank stimulation in burned Mediterranean-type shrublands. Smoke treatments may be used to "flush out" weeds and/or native species that are dormant in the seed bank in areas where fire cannot be applied (Jefferson et al. 2014). A preliminary study of smokewater application to the soil surface showed marginal (most P values > 0.05) increases in native CSS species from the seedbank (Egerton-Warburton unpublished). Many CSS seeds will germinate in the absence of fire or smoke (Keeley 1987, DeSimone and Zedler 1999). However, germination of slender wild oats (*Avena barbata*) and possibly Mediterranean splitgrass (*Schismus* spp.) in southern California was significantly higher when stimulated by smokewater (Engel 2014). If the weed seed bank is depleted by long-term control, smoke could be applied to stimulate longer-lived and dormant CSS species. These potential methods require additional research.

Seeding, Planting, and Seedbanks

Many efforts to restore invaded CSS include seeding native species, with variable success depending on environmental and biotic conditions. Native species often survive in the seedbank, so seeding or planting may not be needed (Allen et al. 2005, Bell et al. 2015), but is required where the seedbank is depleted (Cione et al. 2002, Dickens and Allen 2014). Native seedbank loss occurs when soils are disturbed by agriculture or urban construction (Montalvo et al. 2002), but also with frequent fire followed by post-fire invasion and competition from high-density invaders (Cione et al. 2002). Seedbank density is also dependent on naturally occurring soil pathogens that decrease native seedbank density and subsequent seed germination, more so under high than low moisture conditions (Mordecai 2012). Plant community composition is an insufficient indicator of seedbank composition. In stands of mixed native and exotic plants, seed banks may be especially depleted of native seed (Engel 2014, Cox and Allen 2008a), while in other trials germination of native species from the seedbank was high after exotic species control (Cox and Allen 2008b, Cox and Allen 2011). Where the exotic seedbank was sparser, native plants established through natural colonization without weed control (DeSimone and Zedler 1999, DeSimone 2011). Preliminary tests of seedbank density, or germination responses to invasive plant control need to be done prior to restoration planning efforts.

Seeding is preferred to planting nursery-grown materials because of the reduced cost and labor. CSS seeds will establish readily in years of near average to above precipitation (Padgett et al. 2000). Nursery transplants require additional moisture to establish initially (Eliason and Allen 1997), that can be supplemented with irrigation or hand watering where available. Salvaged plants have also been used with success, as in a Habitat Conservation Plan site in Orange County (Bowler 2000). The advantage of transplants is that they are past the initial drought-sensitive stages of seedlings. If they can survive transplant shock and establish, they will quickly overtop exotic grasses to develop a mature shrub stand. Nevertheless, exotic grass control improves success of nursery transplants (Eliason and Allen 1997, Bowler 2000).

Functional Traits

Recent studies have tested the concept of limiting similarity to restore native species in invaded vegetation. This theory predicts that coexisting species exhibit different resource-use traits to avoid competitive exclusion (Funk et al. 2008, Price and Pärtel 2013). In these studies, native species with traits similar to, or superior to, invaders were chosen for restoration. Common plant traits considered include plant height and biomass, growth rate, root depth, or some other morphological or physiological trait. Cleland et al. (2013) were able to establish native forb species common to CSS when early active forbs were used to match the phenology of exotic grasses. However, the native species chosen to represent functional groups in this study were not all locally native species, such as *Aristida adscensionis,* a non-local, warm season grass, so this was not technically a restoration. A similar approach was used by Talluto et al. (2006), who seeded a mix of native annual forbs and thereby reduced establishment of exotic annuals in a shrub understory and increased overall stand diversity. In another study, planting with shrub plus forb functional groups of native species within CSS communities resulted in greater competition for decreased available resources. This implies a successful approach would be to plant them sequentially, but exotic annuals would also need to be controlled (Kimball et al. 2014). Even when the best efforts are made to select native plants with superior traits, there are a limited number of species to choose from, and invasive species have many traits that make them successful. One trait is early germination, which gave the invasive annuals *Bromus diandrus* and *Erodium cicutarium* a competitive edge over any native species regardless of other traits such as size or growth rate (Hilbig 2015). Wainwright et al. (2012) took advantage of early germination phenology to control seedlings of exotic species and deplete the seedbank before native seeds germinated. Although there are some examples from other vegetation types (Price and Pärtel 2013), the promise of using competitive traits to fully restore CSS diversity based on limiting similarity has not been fulfilled, and at this time controlling exotic species will most often be necessary to assure restoration success.

Restoration Success and Reinvasion

The most successful seeding trials for CSS restoration have been done in soils with limited weed propagule pressure. A study in constructed soil where topsoil had been removed at the Diamond Valley Reservoir, Riverside County, and seeded using hydroseeding and drill seeding had high native shrub, grass and forb density regardless of seeding technique (Montalvo et al. 2002). Little weeding was required during the two-year experiment, and most seed colonized from adjacent areas. After the research was completed and weeding ceased, colonizing invaders were able to establish, and the site is now a shrubland with a sparse understory of native and exotic herbs (Allen, personal observations). Similarly, solarization of abandoned agriculture in former CSS greatly reduced the exotic seedbank and enabled seeded native forbs and shrubs to establish at high density and cover (Marushia and Allen 2011). However, after the study was completed the site was recolonized by exotic grasses and forbs, likely both from the seedbank and adjacent surrounding weedy areas (Allen, personal observations). Grazing and herbicides were used in 1-ha plots at the Shipley Reserve (fig. 1 and Allen et al. 2005) and observed for 5 years. By the fifth year plots began to homogenize such that treatments were no longer significantly different from controls, suggesting that weed control must be done at least every 5 years to maintain increased native forb cover. There was no significant increase in shrub cover in these studies. Weed control in small, 5 m^2 plots at Mt. Rubidoux Park in Riverside enabled establishment of native shrubs (Cione et al. 2002). However, a wildfire burned the shrubs 10 years after establishment, exotic grasses quickly recolonized the plots, and post-fire native shrub establishment was < 1 shrub per 5 m^2 plot (Allen, personal observations, (fig. 2). One of the hallmarks of successful CSS establishment is that the site will regenerate naturally after a fire (Bowler 2000). Clearly this was not a long-term successful restoration. This was a

2A

2B

2C

Figure 2—Restoration at a site with high N deposition and frequent fire in Riverside (Cione et al. 2002). A.) CSS shrubs were established in 5 X 5 m plots in spring 1998 using grass-specific herbicide and hand cultivation (June 2005 photo). B.) Wildfire killed shrubs in October, 2008. C.) Shrub recovery was sparse in May 2010 (green vegetation in plots is exotic forbs), exotic grasses and forbs recolonized the site. (Photo credits: E.B. Allen).

site with high N deposition (20 kg N ha^{-1} yr^{-1}; Cione et al. 2002), and coupled with high exotic grass productivity and small plots that allow more rapid grass recolonization, the restoration had little prospect of long-term success without continual weed management. By contrast, a 2007 burn at a CSS restoration site at Siphon Reservoir in Orange County recovered its shrub component beginning only three years after the fire, and two other Orange County roadside sites that also recovered from fires in 2007 (Margot Griswold personal communication), as well as the Orange County site described by Bowler (2000). At this time the information on post-fire recovery of restored CSS is mainly anecdotal. Data on exotic seedbank density and exotic plant productivity and competition with native species are absent, so conclusions regarding mechanisms of post-fire recovery are difficult to make. One important difference is the lower N deposition in coastal Orange County, which has cleaner air than inland regions (see conclusions). A successful CSS reseeding effort occurred at the San Jacinto Wildlife area in 6.8 ha of abandoned farmland that had been cleanly cultivated to reduce the weed seedbank (fig. 3). Aerial photographs from the mid-1930s showed the site had been CSS, and was subsequently farmed. A mix of native forbs and shrubs (fig. 3) was drill-seeded in 2003, established successfully, and persist to the present. While no data were collected for this project, the photos (fig. 3) indicate that CSS can be restored with some degree of success However, the native forbs that were abundant in the early years declined, and exotic grasses, especially red brome, colonized the site. Three of the initial shrub species declined, and the site is dominated by *Eriogonum fasciculatum*,suggesting this species is best suited to the local conditions. There may be multiple reasons for success at this site compared to the other inland CSS sites reported above: no fires have occurred, the initial seedbank had sparse weed seed (based on field observations, Fig. 3B), the restored area is large so the dispersal of weed seed from adjacent sites was slower than into small plots, and the region has moderate N deposition of 11 kg N ha^{-1} yr^{-1}. This value is at the critical load for vegetation-type conversion (Cox et al. 2014), so long-term restoration success is not guaranteed.

Conclusions

Restoration research in California grasslands has shown that controlling invasive annual grasses promotes establishment of native species, but that within 3-5 years the treated and untreated sites homogenize, with equal proportions of native and exotic species (Larios et al. 2013, Holl et al. 2014). Resistance to restoration is similar in invaded CSS, where restored shrublands become dominated by understories of exotic grasses and forbs after 3-5 years (Allen et al. 2005, and post-publication observations of studies by Cione et al. 2002, Marushia and Allen 2011). The initial success of restoration is determined by the extent to which aboveground competition as well as the exotic seedbank can be controlled. Solarization has the potential to be most successful under appropriate conditions (bare, moist soil) because of the high seedbank mortality, but these conditions are typical only for abandoned agriculture. Solarization has been used experimentally in small plots, but seldom, if at all, on a large scale for actual restoration. Other large-scale techniques, such as fire or herbicides, are temporary because invasive species will eventually recolonize. Thus periodic treatment must be part of any long-term CSS restoration plan.

3A

3C

3B

Figure 3—Restoration of abandoned farmland at San Jacinto Wildlife Area. A.) After drill-seeding with native species, March 2003. B.) Establishment of native forbs and shrubs, sparse invasive species, March 2004. C.) Dominance of California buckwheat, invasive red brome, and sparse native forbs in 2013. Initial seed mix for this site was *Encelia farinosa* (brittlebush), *Artemisia californica* (California sagebrush), *Eriogonum fasciculatum polifolium* (California buckwheat), *Lotus scoparius* (common deerweed), *Amsinckia menziesii* (common fiddleneck), *Hemizonia fasciculata* (clustered tarweed), *Lasthenia californica* (California goldfields), *Layia platyglossa* (coastal tidytips), *Lepidium nitidum* (shining pepper-grass), *Lupinus succulentus* (arroyo lupine), and *Marah macrocarpus* (wild cucumber). (Photo credits: E. B. Allen).

Even after employing best techniques for seedbank control in large areas, long-term restoration success will depend on site factors. Nitrogen deposition is a major driver of exotic grass productivity, and CSS sites that had N deposition above 11 kg ha^{-1} yr^{-1}converted to exotic grassland even in the absence of recent fire (Cox et al. 2014). An additional factor driving conversion was the composition of surrounding vegetation, as CSS stands surrounded by exotic grass were more likely to convert to exotic grassland even in sites with lower N deposition. Such sites are poor candidates for successful restoration of CSS. Some 33% of CSS in California occurs above the critical load of N (Fenn et al. 2010), especially in inland regions where N deposition tends to be higher. Coastal regions have lower N deposition, and include a number of successful efforts at restoration of CSS (Bowler 2006, DeSimone 2013). Restoration will be most successful in sites below critical load of N deposition. This information can be used to inform air quality legislation to improve environmental health of impacted ecosystems such as CSS (Pardo et al. 2011).

With some exceptions (DeSimone 2011), even the best-conserved CSS sites have been invaded, and restoration challenges range from stands with very low invasion to degraded sites dominated by exotic species. Given the high effort and often high cost of controlling exotic plants to restore native species (Bell et al. 2015, Kimball et al. 2015), managers must make decisions on which sites will be able to resist invasion in the long-term, and which will be quickly reinvaded. The latter include sites with high N deposition above critical load of 11 kg ha^{-1} yr^{-1}, continual disturbance impacts from humans or domestic grazers, frequent fire, or surrounding sources of invasive species in the landscape. If sites subject to high reinvasion need to be restored to meet conservation goals, such as to enhance populations of endangered species, then periodic efforts will be needed to control exotic species, and long-term planning is essential (Wilson et al. 2011). Even with the best restoration efforts, the outcome may best be classified as hybrid ecosystems that retain characteristics of the original ecosystem but include invasive species. However, this is an improvement over stands dominated by invasive species that are novel ecosystems with reduced ecosystem services and conservation value (Hobbs et al 2009).

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Shrub Recruitment 10 Years Following Fire on Type-Converted and Native Chaparral Watersheds of San Dimas Experimental Forest, California[1]

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Abstract

Following the 1960 Johnstone wildfire, some areas on the San Dimas Experimental Forest, CA were type-converted from native chaparral to non-native grasses. In 2002, the Williams fire re-burned much of the same area. In 2012, vegetation was measured to document post-fire chaparral recruitment and recovery in the presence and absence of these previous type-conversions. Six 1-3 ha (2.5-7.5 ac) watersheds were studied: three type-converted and three native chaparral. Of 59 plant species identified, 49 were native. Mean cover values for sub-shrubs and grass were significantly greater in type-converted watersheds compared to native chaparral. In contrast, shrub, litter, total live and total cover values were significantly greater in native chaparral watersheds than type-converted. Tree, forb and bare soil cover values were similar among all watersheds. Shrubs and sub-shrubs combined provided 76 percent cover on type-converted watersheds and 114 percent on chaparral watersheds. Type-converted grass cover was mostly *Ehrharta calycina* with values between 4 and 40 percent compared to 6-8 percent in native chaparral watersheds. Over 52 years after type-conversion and 10 years following fire, results show that sub-shrubs and woody shrubs re-established in both type-converted and native chaparral watersheds. While all watersheds were mostly soft and hard chaparral species, two of three type-converted watersheds had a significant component of non-native grass cover. Future disturbances such as close interval wildfire or climate change may further contribute to non-native annual and perennial grass expansion, possibly changing the community recovery dynamics. Further research is needed to identify how and if these historical disturbances will continue to affect this unique landscape and its associated plant assemblages.

Keywords: hard chaparral, non-native plants, post-fire vegetation recovery, soft chaparral, type-conversion, watershed management, wildfire.

Introduction

Type-conversion of fire-adapted chaparral vegetation to non-native herbaceous grasslands has become a major concern in southern California, especially in wildland urban interface zones that burn frequently (Jacobson et al. 2004, Keeley et al. 2005). Non-native species dominate many areas previously occupied by chaparral shrublands (Keeley and Brennan 2012). Furthermore, climate change characterized

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by increased temperatures and erratic precipitation may favor proliferation of invasive species over native vegetation on disturbed sites (D'Antonio 2000). For these and other reasons, disturbances such as close-interval fire can threaten the ecological integrity of California chaparral by reducing native species distributions and diversity while altering ecosystem processes which favor non-native species.

Following fire, chaparral typically progresses initially from herbaceous forbs to "soft" chaparral (Conrad et al. 1986, Paysen et al. 1980) which is composed of shallow rooted, drought-deciduous, semi-woody (subligneous) sub-shrubs such as *Acmispon glaber* (deerweed), *Eriogonum fasciculatum* (California buckwheat), or *Salvia mellifera* (black sage). As time after disturbance increases, "hard" chaparral species composed of longer-lived, woody, deep-rooted sclerophyllous shrubs such as *Adenstoma fasciculatum* (chamise), *Ceanothus* spp. (California lilac), *Quercus* spp. (oak), and *Arctostaphylos* spp. (manzanita) normally dominate southern California chaparral ecosystems (Horton and Kraebel 1955). Generally, closed canopy conditions are associated with mature "hard" chaparral. However, close-interval fires or other disturbances such as mechanical manipulation of the vegetation have been known to change the trajectory of ecological processes where communities do not recover to a pre-disturbed state within typical succession time (D'Antonio and Vitousek 1992, Mack and D'Antonio 1998, Stylinski and Allen 1999). Hence, permanent type-conversion of native plant communities can occur.

Following the 1960 Johnstone wildfire, many watersheds on the San Dimas Experimental Forest (SDEF), Angeles National Forest, CA were type-converted from mixed chaparral shrublands to non-native annual and perennial grasses in order to study resource and watershed management alternatives. To aid the establishment of seeded grasses, various chemical (herbicide) and mechanical methods were used throughout SDEF to remove native shrubs during the first three years following the fire (Corbett and Green 1965, Rice et al. 1965). Species such as *Bromus mollis* (Blando brome), *Agropyron* spp. (pubescent and intermediate wheatgrass), *Lolium rigidum* (Wimmera ryegrass), *Phalaris tuberosa* var. *stenoptera* (Harding grass), *Poa ampla* (big bluegrass), and *Oryzopsis miliacea* (smilo grass) were deliberately seeded in many watersheds (nomenclature follows Corbett and Green 1965). Some objectives of chaparral landscape manipulation during that era included developing techniques to improve water yield, erosion control, wildlife forage, and to assist wildland fire management at the WUI by replacing extremely flammable native shrubs with fast growing, seeded non-native grasses (Bentley 1967).

In 2002, the Williams fire burned about 16,000 ha (40,000 ac) of the Angeles National Forest, including both 42 year old native chaparral and type-converted watersheds on the SDEF. A Joint Fire Science Program study was funded in 2003 (JFSP Project Number 03-2-3-13) in part to investigate if the previously mentioned watershed management actions affected vegetation recovery after the 2002 Williams fire. In 2003, mean species richness by watershed was 35 in type-converted and 33 in native chaparral (Wohlgemuth et al. 2008). Four years post-fire, average species richness decreased for type-converted watersheds to 20 and native chaparral to 22. Initially, grass and forb cover dominated both watershed types, but the majority of cover shifted from herbaceous and grasses to sub-shrubs and shrubs by 2006. Four years post-fire, type-converted watersheds had a combined shrub and sub-shrub cover of 38 percent compared to 46 percent in native chaparral watersheds. In 2006, average grass cover for type-converted watersheds was 13 percent and 3 percent for native chaparral. Less shrub and forb cover and more sub-shrub and grass cover were found in type-converted compared to native chaparral watersheds. By the end of the

2006 measurement period, none of the watersheds had become woody, closed canopy chaparral.

In 2012, the six watersheds from the JFSP study were re-measured to determine whether shrub establishment and community development had changed in type-converted or native chaparral watersheds. More specifically, the intent of this re-assessment was to document if the overall cover of seeded non-native grasses or other non-native species had increased in type-converted watersheds or spread into native chaparral areas following the Williams fire, and if this grass cover corresponded to a decreased sub-shrub and woody shrub cover. Additionally, changes in species richness from 2003 to 2012 were compared between watershed types. Possible long-term vegetation disturbance scenarios in these burned native chaparral and type-converted watersheds include: 1) remain type-converted, 2) remain or return to chaparral, 3) become type-converted from native chaparral to weedy grassland, or, 4) become something in-between.

Methods

This study was conducted on the San Dimas Experimental Forest (SDEF), located in the San Gabriel Mountains, approximately 45 km (27 mi) northeast of Los Angeles, CA (figure 1). SDEF was established in 1933 and is administered by the U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station (Dunn et al. 1988). This reserve consists of over 7,000 ha (17,000 ac) and is centered at a latitude of 34^{o} 12’ N and longitude of 117^{o} 46’ W. It has a typical Mediterranean type climate with hot, dry summers and cool, moist winters. The area generally has southeast facing aspects with mean slopes of 68 percent and elevations between 450 and 1,700 m (1,500-5,500 ft.) (Dunn et al. 1988). The study area is underlain by crystalline metamorphic and intrusive igneous bedrock that typically produces steep slopes and poorly defined soil profiles with non-cohesive sandy loam soils (Wohlgemuth and Hubbert 2008). Soil depths ranged among all study watersheds from 0.15 to 0.24 m (0.39-0.78 ft.) (Unpublished data on file at Riverside, CA). Vegetation in SDEF is dominated by chamise-chaparral and mixed-chaparral, with woodland and riparian vegetation in canyon bottoms (Riggan et al. 1988).

This study documents post-fire vegetation growth 10 years following the 2002 Williams fire in six small watersheds (1-3 ha (2.5-7.5 ac)) which are located southeast of Glendora Ridge Road near Tanbark Flat and Forest Service Roads 1N14 and 1N10. Three of the watersheds were type-converted using herbicides and seeded with non-native grasses following the 1960 Johnstone wildfire (Corbett and Green 1965, Williamson et al. 2004). The other three had 42 year old mixed chaparral at the time of the 2002 Williams fire and served as untreated controls. Prior to the 2002 Williams fire, type-converted watersheds were predominantly *E. fasciculatum* and *S. mellifera* with a large component of the perennial non-native grass *Ehrharta calycina* (African veldt grass) (personal observations). All watersheds were mostly surrounded by native chaparral.

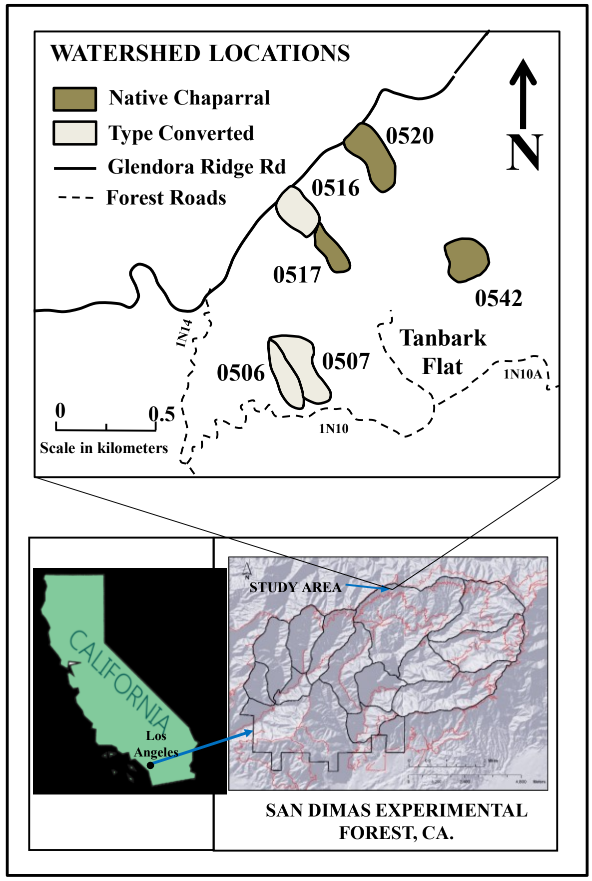

Figure 1—Study area location. The shaded relief map shows the perimeter of San Dimas Experimental Forest with major watershed boundaries, roads and trails within the Angeles National Forest, CA. Upper inset shows the relative locations of six small watersheds used for this study; three type-converted (0506, 0507 and 0516) and three native chaparral (0517, 0520, and 0542).

During July and August 2012, vegetation composition and cover were measured in each of six watersheds using 30 or 40 previously established 10 m (32 ft.) transects (Wohlgemuth et al. 2008). Line intercept sampling (Canfield 1941) was used to quantify plant cover and species richness along each transect (figure 2). Plants were identified to species (nomenclature follows Baldwin et al. 2012). Each species was assigned to a physiognomic growth form (tree, shrub, sub-shrub, forb or grass). The shrub category refers to deep-rooted, evergreen "hard" chaparral species (Sawyer and Keeler-Wolf 1995). Sub-shrubs include drought-deciduous, semi-woody plants sometimes referred to as "soft" chaparral (Conrad et al. 1986, Paysen et al. 1980). Forbs included herbaceous plants other than grasses or grass-like plants. All species were classified as native or non-native (USDA, NRCS Plants Database 2013).

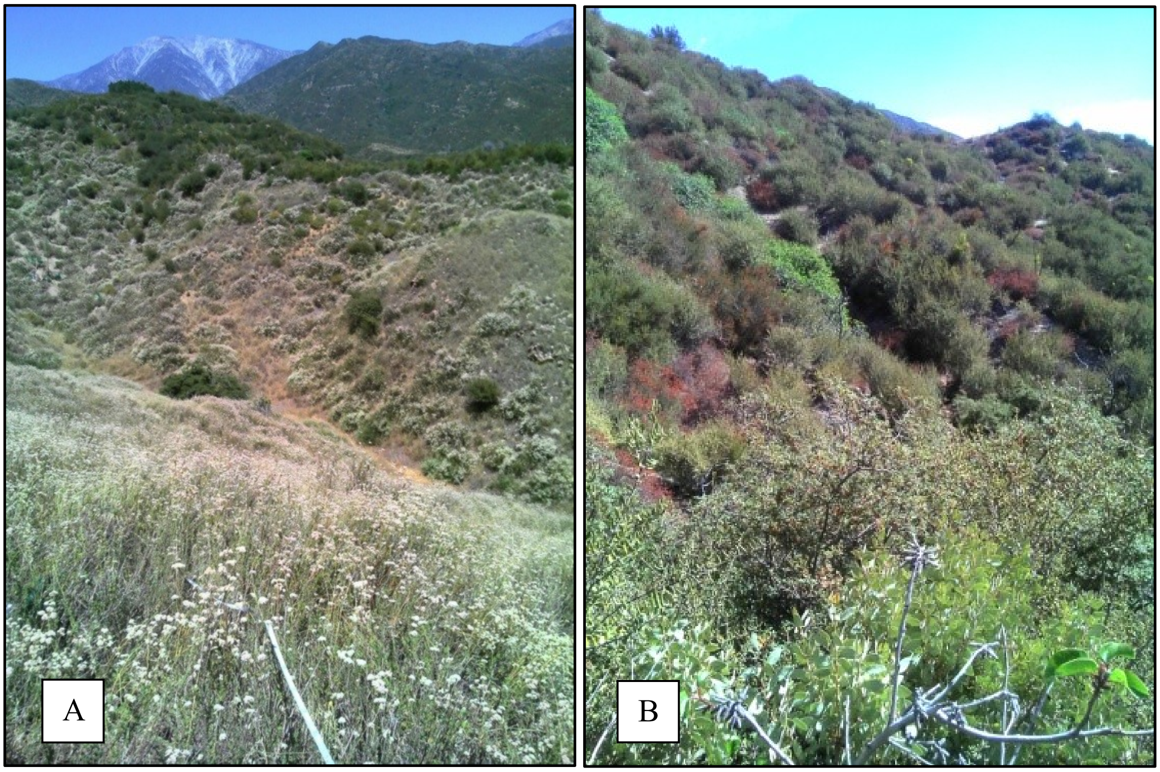

Figure 2—Vegetation for two watershed types were measured in 2012, 10 years after the 2002 Williams fire in the San Dimas Experimental Forest, CA. (A). Dominant live plant cover in type-converted watersheds included the sub-shrub *Eriogonum fasciculatum*, the non-native perennial grass *Ehrharta calycina* and some native shrubs: *Ceanothus crassifolius, Adenostoma fasciculatum,* and *Quercus berberidifolia.* (B). Native chaparral watersheds had an overlapping live plant cover that was predominately shrubs: *A. fasciculatum, C. crassifolius, Arctostaphylos glandulosa* and *Q. berberidifolia*. Photos: USDA Forest Service.

Areas without live vegetation were recorded as cover categories of plant litter or bare soil--which included rocks. If litter was observed in the understory, then litter cover was recorded as well as the plant cover above it. Additional species observed outside designated line transects were documented to identify future seed sources and overall species richness. Species not encountered along line transects (but within 20 m) were noted as "nearby" species. However, cover of these "nearby" species was not measured.

Cover was calculated separately by species and physiognomic growth form for each transect. Mean species cover and richness were compared in the presence and absence of the historic type-conversions for all watersheds and between watershed types to determine the extent of plant community development since the 2003-2006 JFSP study (Wohlgemuth et al. 2008).

To identify significant differences in cover of physiognomic growth forms between watershed types, a linear mixed model analysis was performed using SAS with watersheds a random variable and transects a sub-sample within watersheds. Least-squares means (the mean of the watershed means) were estimated for each watershed type. F-tests were used to test for equality of means between the three native chaparral and three type-converted watersheds. Furthermore, physiognomic growth form cover differences were compared among all watersheds with both watershed and watershed type fixed to identify if there were any significant differences within watersheds irrespective of watershed type. Values were considered significant at $P < 0.05$.

Results

In 2012, 10 years following the Williams fire, a total of 59 species of plants were identified on line transects and "nearby" areas for all six watersheds. As reported in table 1, 44 species were tallied on all transects which included 3 trees, 14 shrubs, 8 sub-shrubs, 13 forbs and 6 grasses.

Of species measured on transects, 80 percent were native. Non-native species included 1 tree (out-planted), 2 forbs, and 6 grasses. Type-converted watershed transects had 9 non-native species compared to 4 in native chaparral. Species richness on type-converted transects was 35 in 2012 and included 2 trees, 10 shrubs, 8 sub-shrubs, 9 forbs, and 6 grasses. Native chaparral watersheds had a species richness of 30 in 2012 and included 1 tree, 11 shrubs, 8 sub-shrubs, 6 forbs, and 4 grasses. Five tree species were observed in total, but very little tree cover was encountered on line transects in any watershed.

Species composition and cover varied between watershed types (table 1; table 2). Of all native species observed, 34 occurred in type-converted watersheds and 35 in native chaparral. Of all non-native species observed, 10 occurred in type-converted and 6 occurred in native chaparral watersheds. Three tree, 4 shrub, 1 sub-shrub, and 7 forb species were found in the type-converted watersheds that were not found in native chaparral. One tree, 6 shrub, and 7 forb species were found in the native chaparral watersheds, but not the type-converted.

In 2012, non-native species cover in type-converted watersheds was predominately the non-native perennial grass, *E. calycina*, followed by *Festuca myuros* (rattail fescue) (table 1). Two type-converted watersheds had 33 and 39 percent *E. calycina* cover, while the third had less than 2 percent (unpublished data on file at Riverside, CA). For native chaparral watersheds, non-native species cover consisted of 5 percent *Bromus madritensis* ssp. *rubens* (red brome) and 2 percent was *E. calycina*. Trace occurrences of six other non-native species in both watershed types included *Avena fatua* (wild oats), *Bromus diandrus* (ripgut brome), *Bromus tectorum* (cheat grass), *Centaurea melitensis* (tocalote), and *Hirschfeldia incana* (short pod mustard). Non-native species and cover were measured on 76 percent of transects in type-converted watersheds and observed "nearby" 52-100 percent of the time (unpublished data on file at Riverside, CA). This contrasts with native chaparral watersheds which had non-native species cover occurring on an average of 45 percent of transects and non-native species were observed "nearby" between 47 and 67 percent of the time.

Table 1—Mean percentage live cover on line-transects by species for three type-converted and three native chaparral watersheds 10 years following the 2002 Williams fire in the San Dimas Experimental Forest, CA. Mean values are averages over all transects within a watershed type. Physiognomic growth forms are defined in Methods section of text. Note: (n) = number of 10 m transects; (--) indicates zero plant cover; (*) indicates non-native species.

Genus and species	Common name	Mean percentage live cover by species: Type-converted n=99	Native chaparral n=100	Physiognomic growth form
Acmispon glaber	Deerweed	1.50	0.70	sub-shrub
Adenostoma fasciculatum	Chamise	3.88	18.40	shrub
Arctostaphylos glandulosa	Eastwood manzanita	0.68	0.90	shrub
Arctostaphylos glauca	Bigberry manzanita	--	0.20	shrub
*Avena fatua**	Wild oats	0.14	0.40	grass
Baccharis salicifolia	Mule fat	0.46	--	shrub
*Bromus diandrus**	Ripgut brome	0.10	0.10	grass
Bromus madritensis ssp. *rubens**	Red brome	0.78	4.30	grass
*Bromus tectorum**	Cheat grass	0.02	--	grass
Calochortus splendens	Splendid Mariposa lily	0.11	--	forb
Ceanothus crassifolius	Hoaryleaf ceanothus	10.84	35.70	shrub
Ceanothus integerrimus	Deer brush	0.20	0.40	shrub
*Centaurea melitensis**	Tocalote	0.05	--	forb
Cercocarpus betuloides	Mountain mahogany	--	0.40	shrub
Cryptantha intermedia	Common cryptantha	0.02	--	forb
Cuscuta sp.	Dodder species	0.02	--	forb
Ehrendorferia chrysantha	Golden ear-drops	--	0.10	forb
*Ehrharta calycina**	African veldt grass	23.39	1.10	grass
Erigeron sp.	Fleabane	0.02	--	forb
Eriodictyon trichocalyx	Smooth leaf yerba santa	7.96	4.20	sub-shrub
Eriogonum fasciculatum	California buckwheat	44.01	18.80	sub-shrub
Eriophyllum confertiflorum	Golden yarrow	0.69	0.50	sub-shrub
*Festuca myuros**	Rattail fescue	1.32	--	grass
Garrya veatchii	Silktassel bush	--	0.20	shrub
Hazardia squarrosa	Sawtooth goldenbush	0.37	0.30	sub-shrub
Helianthemum scoparium	Common rush rose	2.26	0.90	sub-shrub
Helianthus gracilentus	Slender sunflower	0.12	0.30	forb
Hesperoyucca whipplei	Chaparral yucca	0.38	0.30	sub-shrub
Heteromeles arbutifolia	Toyon	0.29	0.20	shrub
*Hirschfeldia incana**	Short pod mustard	0.09	0.40	forb
Malacothamnus fasciculatus	Bush mallow	--	0.10	shrub
Malosma laurina	Laurel sumac	1.03	--	shrub
Marah macrocarpus	Wild cucumber	--	0.70	forb
Mimulus cardinalis	Scarlet monkeyflower	0.06	--	forb
Pellaea mucronata	Cliff brake	--	0.01	fern/forb
Phacelia cicutaria	Caterpillar phacelia	0.01	--	forb
Pinus sp.*	Pine species	0.06	--	tree
Prunus ilicifolia	Hollyleaf cherry	0.42	--	shrub
Pseudognaphalium biolettii	Bioletti's everlasting	--	0.02	forb
Quercus agrifolia	Coast live oak	--	0.30	tree
Quercus berberidifolia	Scrub oak	0.36	4.00	shrub
Rhus ovata	Sugar bush	0.75	12.80	shrub
Salix gooddingii	Black willow	0.06	--	tree
Salvia mellifera	Black sage	5.05	15.10	sub-shrub
Total Number of Species:	44	35	30	

Table 2—Species observed "nearby" (within 20 m), but not on the 10 m line transects for three type-converted (TC) and three native chaparral (NC) watersheds 10 years following the 2002 Williams fire in the San Dimas Experimental Forest, CA. Physiognomic growth forms defined in Methods section of text. Note: (n) = number of 10 m transects; (*) indicates non-native species; (+) indicates species presence "nearby"; (--) indicates species absence "nearby".

Genus and species	Species occurring nearby Common name	TC n=99	NC n=100	Physiognomic growth form
Calystegia macrostegia	California morning glory	--	+	forb
Camissoniopsis bistorta	California suncup	--	+	forb
Ceanothus oliganthus	Hairy ceanothus	+	+	shrub
*Cirsium vulgare**	Bull thistle	--	+	forb
Ericameria parishii	Parish's goldenbush	+	--	shrub
Penstemon sp.	Penstemon species	+	+	forb
Plantanus racemosa	Western sycamore	+	--	tree
Pseudognaphalium californicum	California everlasting	+	+	forb
Quercus chrysolepis	Canyon live oak	+	+	tree
Rhamnus ilicifolia	Holly-leaf redberry	--	+	shrub
Rhus aromatica	Basket bush	--	+	shrub
Salvia columbariae	Chia	+	+	forb
*Spartium junceum**	Spanish broom	+	+	shrub
Toxicodendron diversilobum	Poison oak	+	--	sub-shrub
Trichostema lanatum	Woolly bluecurls	--	--	sub-shrub
Total number "nearby" species:	15	9	11	

F-tests showed significant differences in mean cover of some growth forms between the two watershed types (table 3). Means for shrub, forb, litter, and total cover were significantly less in type-converted watersheds compared to native chaparral. Mean live cover for type-converted watersheds included a combined shrub and sub-shrub cover of 76 percent (table 3). Live cover in type-converted watersheds was predominately *E. fasciculatum* and *E. calycina*, followed by *A. fasciculatum*, *C. crassifolius*, *Eriodictyon trichocalyx* (smooth leaf yerba santa) and *S. mellifera* (table 1). In contrast, mean live cover for native chaparral watersheds contained an

Table 3—Percentage of mean cover by physiognomic growth form between three type-converted and three native chaparral watersheds 10 years following the 2002 Williams fire on the San Dimas Experimental Forest, CA. P-values from F-tests show whether mean cover values between watershed types were significantly different at the 0.05 percent level. Note: (n) = number of watersheds.

	Mean percentage of growth forms between watershed types		
Physiognomic growth Form	Type-converted n=3	Native chaparral n=3	*P-value*
Tree	0.10	0.27	0.5966
Shrub	19.35	72.66	0.0018
Sub-shrub	56.59	41.16	0.1315
Forb	0.51	1.55	0.0152
Grass	26.66	6.64	0.1561
Litter	15.13	44.04	0.0420
Bare/Soil	15.82	16.24	0.8878
Total Live	102.59	122.42	0.0945
Total Cover	133.74	182.69	0.0380

overlapping shrub and sub-shrub cover totaling 114 percent (table 3). Predominant cover for native chaparral watersheds included *Ceanothus crassifolius* (hoaryleaf ceanothus), followed by *E. fasciculatum* and *A. fasciculatum.* Additional shrub cover in native chaparral watersheds also included *S. mellifera*, *Rhus ovata* (sugar bush), *Quercus berberidifolia* (scrub oak) and *E. tricocalyx* (table 1).

Significant site-specific differences in mean cover by physiognomic growth form were found among the six watersheds regardless if type-converted or native chaparral (table 4). Shrub cover in all type-converted watersheds was significantly lower than in all native chaparral. Sub-shrub cover in one type-converted watershed was significantly higher than another type-converted watershed. No significant differences were found for sub-shrub cover among native chaparral watersheds. Between watershed types, sub-shrub cover was significantly higher in two of three type-converted watersheds compared to two of three native chaparral watersheds. Mean grass cover (predominately *E. calycina*) in one type-converted watershed was similar to native chaparral. Litter cover did not vary among type-converted watersheds, and two of three native chaparral watersheds had significantly greater litter cover compared to type-converted. Total live cover among type-converted watersheds and one native chaparral did not differ significantly. The two remaining native chaparral watersheds had significantly greater live cover overall compared to type-converted. This same trend was also observed for total cover. There were no significant differences found among all watersheds for tree, forb, or bare ground cover.

Discussion

Over 52 years after type-conversion and 10 years since the 2002 Williams fire, the most widespread plant growth forms in native chaparral watersheds were shrubs and sub-shrubs. In contrast, there was a significant component of the non-native perennial grass *E. calycina* in two of the three type-converted watersheds, and "hard" chaparral shrub cover was significantly less in all type-converted watersheds compared to native chaparral (table 3). The substantial proportion of perennial grass cover in these watersheds raises concerns for the passive recovery of native plants and associated habitat, especially if the grass component continues to increase. Chaparral shrublands are important for the preservation of ecosystem services such as wildlife habitat, hydrological function, carbon storage, oxygen production, and hillslope stability, especially in southern California's WUI. Loss of resilient chaparral ecosystems due to type-conversion could jeopardize the ecological integrity of California's unique natural landscape (Allen-Diaz 2000, Lambert et al. 2010).

Native shrub and sub-shrub recruitment and success is an essential part of the post-disturbance recovery process for chaparral. After the 2002 Williams fire, mean shrub and sub-shrub cover increased in type-converted watersheds from 38 percent in 2006 (Wohlgemuth et al. 2008) to 76 percent six years later in 2012. In comparison, native chaparral watershed mean shrub and sub-shrub cover increased from 46 percent in 2006 (Wohlgemuth et al. 2008) to 114 percent in 2012. The substantial sub-shrub cover with increasing woody shrub cover suggests that these watersheds may be in transition to becoming a closed-canopy mixed-chaparral community given more time. However, non-native perennial grass cover is still persisting in over one third of the landscape in two type-converted watersheds compared to about 6 percent in native chaparral (table 4).

Table 4—Tukey test comparisons among all watersheds showing statistical significance of differences in physiognomic growth form cover means. Cover values were analyzed using both watershed type and watershed as fixed effects. Means for the same variable (shrub, sub-shrub, grass, litter; total live, and total cover) with the same letter are not significantly different from each other at the five percent level. No significant differences were found among watersheds for tree, forb, or bare/soil cover (not shown). Note: (n) = number of 10 m transects sampled; (TC) indicates type-converted; (NC) indicates native chaparral; (-) indicates no significant differences; (*) indicates significant differences (*P*<0.05).

	Significant differences of mean percentage cover among watersheds								
Physiognomic growth form			Type-converted			Native chaparral			
	Watershed	% Cover	0506 n=29	0507 n=30	0516 n=40	0517 n=30	0520 n=40	0542 n=30	Tukey significance
Shrub			-	-	-	*	*	*	a
	TC 0507	19.38	-	-	-	*	*	*	a
	TC 0516	25.49	-	-	-	*	*	*	a
	NC 0517	69.47	*	*	*	-	-	-	b
	NC 0520	64.69	*	*	*	-	-	-	b
	NC 0542	84.83	*	*	*	-	-	-	b
Sub-shrub	TC 0506	59.04	-	-	-	*	-	-	ab
	TC 0507	44.30	-	-	*	-	-	-	bc
	TC 0516	65.83	-	*	-	*	-	*	a
	NC 0517	31.92	*	-	*	-	-	-	c
	NC 0520	49.46	-	-	-	-	-	-	abc
	NC 0542	41.53	-	-	*	-	-	-	bc
Grass	TC 0506	39.92	-	-	*	*	*	*	a
	TC 0507	36.33	-	-	*	*	*	*	a
	TC 0516	3.94	*	*	-	-	-	-	b
	NC 0517	5.62	*	*	-	-	-	-	b
	NC 0520	7.79	*	*	-	-	-	-	b
	NC 0542	6.50	*	*	-	-	-	-	b
Litter	TC 0506	12.36	-	-	-	*	*	*	a
	TC 0507	16.47	-	-	-	-	*	*	ab
	TC 0516	16.54	-	-	-	-	*	*	ab
	NC 0517	26.95	*	-	-	-	*	*	b
	NC 0520	44.58	*	*	*	*	-	*	c
	NC 0542	60.58	*	*	*	*	*	-	d
Total Live	TC 0506	111.67	-	-	-	-	-	*	ab
	TC 0507	100.51	-	-	-	-	*	*	a
	TC 0516	96.09	-	-	-	-	*	*	a
	NC 0517	107.98	-	-	-	-	-	*	ab
	NC 0520	124.06	-	*	*	-	-	-	bc
	NC 0542	135.10	*	*	*	*	-	-	c
Total Cover	TC 0506	138.31	-	-	-	-	*	*	ab
	TC 0507	136.88	-	-	-	-	*	*	ab
	TC 0516	126.22	-	-	-	*	*	*	a
	NC 0517	154.82	-	-	*	-	*	*	b
	NC 0520	184.49	*	*	*	*	-	*	c
	NC 0542	208.72	*	*	*	*	*	-	d

Differences in species composition between watershed types was observed (table 1; table 2). Average species richness on type-converted transects increased from 20 in 2006 (Wohlgemuth et al. 2008) to 22 in 2012. Native chaparral watersheds had an average species richness of 22 in 2006, which decreased to 20 in 2012. Chaparral species community composition can be related to stand age (Keeley 1992, Patric and Hanes 1964). In addition, micro-site differences may support different community compositions among the various watersheds. The seed sources of native trees and woody shrubs identified on transects and "nearby" show that native species abundance and cover has the potential to increase over time in these watersheds.

Non-native cover (grasses and forbs) was much higher on type-converted watersheds compared to native chaparral (table 1). Schultz et al. (1955) showed that non-native annual grasses can competitively exclude chaparral seedlings. It is unknown what role the perennial grass *E. calycina* may play in possible future colonization of "hard" chaparral species. Additional monitoring of physiognomic growth form cover and diversity, especially non-native herbaceous and perennial grass cover at these sites, could determine if type-converted watersheds remain mixed "soft" chaparral and grasses, or if species composition will shift towards a predominately "hard" chaparral shrub community given more time. Furthermore, it would be informative to know if non-native grass cover will continue to increase or if it will move into intact native chaparral watersheds given the close proximity to type-converted areas.

In 2012, mean non-native grass cover in type-converted watersheds was 27 percent, almost four times greater than the 7 percent found in native chaparral (table 3). This was a two-fold increase in grass cover for both watershed types since 2006 as reported by Wohlgemuth et al. (2008). Interestingly, of the three type-converted watersheds, the one with the lowest grass cover also had the highest cover of shrubs and sub-shrubs. Keeley et al. (2005) found that the most critical factor influencing non-native plant population dominance in chaparral is the rapid return of the sub-shrub and woody shrub cover. Because the cover of the perennial grass *E. calycina* has increased considerably from 2006 to 2012, and it was accompanied by less shrub and sub-shrub cover in the watersheds where it was abundant, its presence, a remnant of the type-conversion manipulations that occurred in the 1960s, may be an enduring component in the chaparral ecosystems of SDEF.

Relationships between native and non-native plant species diversity have been suggested to play a determining role in non-native plant invasions (Elton 1958). The perennial grass *E. calycina* has successfully persisted in SDEF since it was introduced in the 1960s. This grass, and other non-native species observed, could serve as potential seed sources for future expansion of non-native plant populations. Shifts in vegetation assemblages of native chaparral species to non-native annual and perennial grasses may increase fire return intervals, affect soil water availability, alter carbon storage, and change community population dynamics (D' Antonio 2000, Facelli and Pickett 1991, Jacobson et al. 2004, Keeley et al. 2005, Keeley and Brennan 2012).

Shallow, fibrous root masses of many non-native grasses are believed to inhibit native shrub seedling establishment (Eliason and Allen 1997). In SDEF, the size and distribution of root masses in the soil profile under non-native grasses have been shown to vary considerably from that of native chaparral plant species (Williamson et al. 2004). Non-native forbs and grasses can alter soil-water utilization because root depth, root number and root size are different from native species (Holmes and Rice 1996, Perkins and Nowak 2013, Williamson et al. 2004). Compared to chaparral vegetation, soil temperatures under the perennial grass *E. calycina* have been shown to be warmer from September to March and cooler from June to September (Williamson 2004). In addition, organic matter in soils under *E. calycina* grass tussocks was greater, A-horizons were thicker, and water content was less compared to under chaparral, indicating that pedogenic processes have been altered under type-converted vegetation compared to chaparral (Williamson et al. 2004). Effects of the perennial grass *E. calycina* on soil water availability and soil formation could perturb chaparral population dynamics, particularly if this perennial grass cover continues to increase. This is of further concern when considering unknown future climatic changes such as erratic rainfall and higher temperatures.

Changes to soil properties that are caused by plants, which in turn influence the performance of microbiota and soil fauna populations as well as other plants, are termed plant-soil feedbacks (e.g. allelopathy or carbon fixation) (Ehrenfeld et al. 2005, Perkins and Nowak 2013, Van der Putten et al. 2013). Physical, chemical, and biological changes to the soil environment by non-native plant species can profoundly alter many ecosystem processes (Ehrenfeld 2010, Vitousek et al. 1990). Through changes in the demography of plant and microbiota populations, and/or the physiological activity of individuals, non-native species can influence the dynamics of coexistence, invasion, and the restoration success of an ecosystem (Ehrenfeld 2003, Van der Putten et al. 2013). Consequences of different mycorrhizal fungi diversity associated with grasses compared to shrubs on plant-soil feedback mechanisms have been shown to influence plant biodiversity, productivity, variability, and stability, all of which are critical for ecosystem functioning (Egerton-Warburton and Allen 2000, Eliason and Allen 1997, van der Heijden et al. 1998). Therefore, it is possible that non-native grasses in type-converted watersheds may be adversely affecting successional trajectories by changing the most basic building blocks of the trophic system, thereby altering the natural transition toward "hard" chaparral species dominance by displacing the native flora as a result of competition for light, nutrients, and water (Dahlin et al. 2013, Eliason and Allen 1997). Further examination into whether there is some threshold density of cover by non-native grasses in chaparral ecosystems at which ecosystem function breaks down would be informative.

The perennial grass component in type-converted watersheds may be even more problematic if future disturbances such as close-interval fire occur. *E. calycina* accumulates a large and persistent seed bank, readily resprouts after fire, and has been associated with severe soil water repellency (Smith et al. 1999, Williamson 2004). Once established in an area, non-native grasses may shorten fire return intervals by providing highly ignitable flashy fuels (Keeley and Brennan 2012, Zedler et al. 1983). Absence of fire for long periods of time is necessary for obligate seeders to reproduce in chaparral ecosystems (Jacobsen et al. 2004). Air pollution (nitrogen deposition) effects on "soft" chaparral vegetation can decrease cover and biomass of native sub-shrubs, forbs and mycorrhizal fungi while increasing cover and biomass of non-native grass species (Allen et al. 2005, Egerton-Warburton and Allen 2000, Perkins and Nowak 2013). Alteration of disturbance regimes can have profound effects on a functional group of species (Mack and D'Antonio 1998), threatening the resilience of a healthy ecosystem. Some or all of these factors may have contributed to the differences found in watershed species composition, which included significantly higher grass and lower shrub cover in type-converted compared to native chaparral watersheds.

Mean litter cover was significantly greater in native chaparral compared to type-converted watersheds (table 3). Litter plays a critical role in plant species composition which can mediate nutrient, water, and carbon cycling within and among different populations of flora and fauna (Dahlin et al. 2013, Ehrenfeld 2003, Facelli and Pickett 1991). The importance of the type, as well as the amount, of litter present can also affect chaparral seedling survival (Keeley 1992, Patric and Hanes 1964). Differences in litter composition, structure, and mass associated with grasses compared to shrubs may contribute to chaparral germination success or failure. In native chaparral watersheds, the robust litter layer may provide a more suitable seedbed for germination and seedling survival for many native chaparral species compared to the grass and sub-shrub litter of type-converted watersheds.

Mean forb cover was greater in native chaparral compared to type-converted (table 3), but very low values were observed in both watershed types. The low cover and low species richness of forbs measured may have been due to time since fire, the presence of grasses, or perhaps timing of sampling in late summer (Beyers et al. 1998, Hubbert et al. 2012, Keeley and Keeley 1989, Muller et al. 1968, Williamson 2004). Sampling during the active growing season may increase the detection of forbs.

Bare ground cover was similar among all watersheds and it has not changed since 2006 as reported by Wohlgemuth et al. (2008). Canopy gaps are important for insolation stimulation of germination and growth of some native species (Baskin and Baskin 1998). These gaps of bare soil may allow different species to expand their distributions into available spaces. Keeley (1992) found that in arid, open landscapes, fire-persisters (*Quercus, Rhamnus, Heteromeles*) do poorly and fire recruiters (*Adenostoma, Arctostaphylos,* and *Ceanothus*) dominate. In time, as these watersheds age and litter accumulates in bare gaps, the potential exists for a greater variety of chaparral species to re-establish and flourish.

Conclusions

Ten years after fire were sufficient for native chaparral watersheds on the San Dimas Experimental Forest to return to shrub dominated conditions, although they were not yet closed-canopy. In contrast, live plant cover in type-converted watersheds was substantially different, in that it was predominately sub-shrubs and non-native grasses with some shrubs. Less "hard" chaparral shrub cover and more grass cover in type-converted compared to native chaparral watersheds is an enduring effect of the deliberate type-conversion that occurred 52 years prior to this study. These differences in vegetation growth form cover demonstrate how unaided post-fire chaparral species recovery has occurred in previously type-converted and native chaparral watersheds in southern California. Additional monitoring would document whether past type-conversion permanently shifted the plant community to a new, stable species composition consisting of sub-shrubs and the non-native perennial grass *E. calycina*, or if "hard" chaparral species can recolonize and eventually establish dominance in these grassy watersheds. Site-specific conditions may affect both chaparral species composition and re-establishment success and possibly the time interval required for succession to occur following disturbance. Effective resource and watershed management of deliberately type-converted shrublands at the wildland-urban interface may require pro-active management interventions to eradicate non-native species in order to maintain the rich native plant diversity unique to southern California's chaparral.

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The Feasibility of Chaparral Restoration on Type-Converted Slopes[1]

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Abstract
Increased fire frequency, livestock grazing, and other disturbances have converted many chaparral stands in southern California to non-native annual grassland. Competition from annual grasses interferes with the establishment of chaparral shrubs, presenting a challenge to restoring chaparral in type-converted areas.
In this study we examined the feasibility of restoring chaparral by exploring several methods of restoration on type-converted slopes in San Timoteo Canyon, Riverside County, California. We assessed the effectiveness of a broad-spectrum herbicide (glyphosate) and a grass-specific herbicide (fluazifop-p-butyl) in facilitating shrub establishment. In addition, we compared the success of seeding and planting containerized shrub seedlings as methods of restoration and examined the soil seed bank to see if there was a relict seed bank that could be manipulated to promote restoration. The broad-spectrum herbicide application significantly increased soil moisture and the successful establishment of transplanted seedlings. The grass-specific herbicide, applied later in the season, was less effective. Seeding was unsuccessful, most likely due to low rainfall during the study, and few germinable seeds of native chaparral species were detected in the grassland soil.

Keywords: chaparral, restoration, type-conversion, glyphosate, fluazifop, grass competition, seed bank.

Introduction

In southern California, type conversion of chaparral to non-native annual grassland has occurred on many landscapes due to disturbances such as frequent fire and livestock grazing. Deliberate conversion of chaparral to grassland for purposes such as improving grazing land and wildlife habitat is a long-standing practice that may pre-date European settlement (e.g., Keeley 2002). The USDA Forest Service pursued type-conversion beginning in the late 1950s for these and other purposes (e.g., providing fuel breaks; Bentley 1967, Tyrrel 1982). Fire frequency increased during the 20th century due to increases in human-caused ignitions, especially at the wildland-urban interface (Keeley et al. 1999, Rundel and King 2001, Keeley and Fotheringham 2003). Short fire-return intervals can extirpate chaparral shrub species

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by killing obligate seeders before they mature, and extremely short fire-return intervals reduce the ability of resprouters to recover (Zedler et al. 1983, Halsey 2005 and references therein). Colonization of burned or disturbed areas by non-native annual grasses consequently leads to a positive feedback cycle that further increases fire frequency, increases competitive suppression of natives, and alters ecosystem structure and function (Zedler et al. 1983, D'Antonio and Vitousek 1992, Minnich and Dezzani 1998, Stylinski and Allen 1999, DiTomaso 2000, Cione et al. 2002, Syphard et al. 2006).

It has been recognized that converting chaparral to non-native grassland may incur costs. On steep slopes, grasses are less effective than shrubs at preventing land slips after heavy rains (Corbett and Rice 1966). Chaparral shrubs vary in rooting habit (Helmers et al. 1955), but because they are generally much more deeply rooted, they are more effective at preventing slope failure than the annual grasses (Rice et al. 1969). From a different perspective, increasing recognition of threats to global biodiversity and of the unique native flora of California has led to many attempts to restore native systems. Restoring slow-growing shrubs in areas dominated by fast-growing annual grasses, however, can be challenging

In California, non-native annual vegetation often outcompetes shrub and tree seedlings for limited summer soil moisture, thereby preventing woody plant establishment (e.g., Gordon et al. 1989, Williams and Hobbs 1989, Eliason and Allen 1997). As a result, research into restoring shrublands, such as coastal sage scrub, has often incorporated tests of methods for reducing grass competition. These methods have included weeding, mowing, disking, soil solarization, and employing grass-specific herbicides (Allen et al. 2000, Cione et al. 2002, Cox and Allen 2008, Marushia and Allen 2011). Grass-specific herbicides offer attractive options in restoration because their specificity allows managers to treat areas without threatening most restoration plantings. They do not control invasive broad-leaf weeds, however, which may also suppress restoration plantings. In contrast, common broad-spectrum herbicides, such as glyphosate, cannot be applied easily after desirable shrubs or forbs emerge or are planted. We explored a combined approach to suppressing non-native competitors by using a broad-spectrum herbicide prior to planting and a grass-specific herbicide after planting to control late-germinating grass cohorts.

The overall goal of this study was to explore the feasibility of restoring chaparral on type-converted slopes by (1) comparing the effectiveness of a grass-specific herbicide (fluazifop) to that of a broad-spectrum herbicide (glyphosate) with a fluazifop follow-up in facilitating chaparral restoration, (2) comparing the effectiveness of seeding and planting containerized seedlings as methods of restoration, and (3) determining whether a relict chaparral seed bank existed on the site that could be manipulated for restoration.

Study Area

Research was conducted on an ecological preserve owned by the Riverside Land Conservancy in San Timoteo Canyon, Riverside County, California (33° 58.252' N, -117° 3.986' W). Historic land use of the San Timoteo Canyon has dramatically altered the natural habitats in the canyon and the preserve. Prior to European colonization, the Cahuilla tribe occupied areas of the canyon (Christian 2002). Early settlers, in the 1850s, used the canyon for raising and grazing livestock, grain and hay farming, dairy operations, beekeeping, and citrus groves. The grazing and agricultural practices led to type-conversion from native shrubs to non-native annual grasslands and cropland. Historic aerial photographs show that many hillsides in the canyon were once covered by chaparral; now they are covered primarily by non-native annual grasses. Agricultural operations continued late into the 20th century, until the land was purchased and subdivided into residential development projects, state parks, and county parks (Knecht 1971, Christian 2002).

On the preserve where this study was conducted, agricultural practices began in approximately 1875 and ceased in 2003 when the Riverside Land Conservancy obtained the property (Jack Easton, Riverside Land Conservancy, pers. comm. 18 Nov. 2013). The study site supports remnant chaparral stands on ridge tops that are dominated by *Adenostoma fasciculatum* Hook. & Arn., *Eriogonum fasciculatum* Benth., *Artemisia californica* Less., *Rhus aromatic* Aiton, *Rhamnus crocea* Nutt., *Rhus ovata* S. Watson, and *Quercus berberidifolia* Liebm. Our experiments were conducted on west-facing grass-dominated hillsides with slopes of 35-60 percent, at an elevation of approximately 650 m.

Methods

Effectiveness of various manipulations on chaparral shrub establishment was investigated in a factorial experiment. Treatments consisted of four restoration methods (no restoration treatment, smoke-water application, applying seeds of chaparral shrubs, and planting container stock) across three herbicide applications (no herbicide, glyphosate with a follow-up fluazifop application [glyphosate + fluazifop follow-up], and only fluazifop). The smoke-water treatment was applied in an attempt to stimulate the germination of smoke-responsive seeds in the soil seed bank at the site (Roche et al. 1997). The resulting 12 treatments were replicated three times in a randomized block design. Treatments were applied in plots measuring 2 x 2 m, except for plots in which containerized seedlings were planted, which measured 2 x 5 m.

Herbicide Application

Glyphosate plots were treated with Ranger® PRO[1] (Monsanto Co.) at a rate of 2.34 L ha^{-1} (1.123 kg glyphosate ha^{-1}). For plots seeded in mid-December, glyphosate was applied within 24 hours of seeding; for other plots glyphosate was applied on 23 Jan 2013. The fluazifop-only and the glyphosate + fluazifop follow-up plots were treated with Fusilade®II (Syngenta, Inc.) after restoration treatments were applied (below) and when grasses were more active, on 6 Mar 2013, at a rate of 1.32 L ha^{-1} (316 g fluazifop-P-butyl ha^{-1}) with 0.5% surfactant (Activator 90). Herbicide was applied via backpack sprayer with an 8004 vs nozzle.

Seed Application

Seeds of chaparral shrubs (*Adenostoma fasciculatum* Hook. & Arn., *Gutierrezia sarothrae* (Pursh) Britton & Rusby, *Quercus berberidifolia* Liebm., *Rhus aromatica* Aiton, and *Rhus ovata* S. Watson) were purchased from Wild California, a local

[1] The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

wholesale nursery. Additionally, seeds of two shrub species more characteristic of coastal sage scrub or early successional environments (*Artemisia californica* Less. and *Eriogonum fasciculatum* Benth.) were purchased and included in the seed mix for comparison. Four of these species were pre-treated by Wild California staff. *Adenostoma* seed received 24 hour soak in liquid chamise smoke, which was prepared by Wild California staff. *Quercus* received cold stratification, and both *Rhus* species received an acid wash.

Eight acorns were planted in each seeded plot, with other species being sown at 4-9 kg seed ha^{-1} to yield an overall rate of 35 kg pure seed ha^{-1}. Seeding rates from 2.2-33 kg ha^{-1} have been used in restoration: the seeding rate is dependent on species, size of seed, and the number of species being used (e.g. DeSimone 2011; CalTrans 2012). Seeds were sown on 14 Dec 2012. Seeds of the two *Rhus* species and the oak were distributed into a shallow ditch and then tamped down. Soil "packing," or tamping, guarantees good seed-soil contact and maintains soil moisture (Morgan 1997). The other, smaller seeds were broadcast over the plot by hand.

Containerized Seedling Installation

Containerized seedlings of *Adenostoma fasciculatum, Eriogonum fasciculatum, Quercus berberidifolia,* and *Rhus ovata* were purchased from Wild California, Recon Native Plants, Rancho Santa Ana Botanic Garden, and Riverside-Corona Resource Conservation District. Plants were in 1-gallon pots except for *Rhus ovata* seedlings, which were only available in 2-gallon pots, and the *Quercus* seedlings, which were only available in deep bullet pots.

There were two planting dates due to the timing of herbicide application. Plots scheduled to receive fluazifop only (a grass-specific herbicide) were planted as early in the season as the onset of winter rains allowed. Plots scheduled to be treated with glyphosate (a broad-spectrum herbicide) were not planted until after glyphosate application. The fluazifop and no-herbicide plots were planted on 20 Dec 2012. The glyphosate + fluazifop follow-up plots were planted on 30 Jan 2013. Each 2 x 5 m plot was planted with three *Adenostoma* seedlings (aside from one plot that was planted with 2 due to limited seedling availability), five *Eriogonum* seedlings, five *Quercus* seedlings and five *Rhus ovata* seedlings. Shallow basins were dug adjacent to each seedling to facilitate subsequent watering. The plots that received seedlings were watered once or twice a month as needed, at a rate of 1 L per plant, throughout the remaining winter and spring season.

Plots were surveyed every 2-3 weeks for seedling survival. At the same time, the height and width of each plant was measured. Live plant canopy volume was then calculated as the volume of a spheroid from the height and width measurements.

Effects of Herbicide and Planting on Soil Moisture

To assess the effect of herbicide treatments on soil water status, soil samples were collected in April 2013 from all three herbicide treatments in plots with no restoration treatment and in plots planted with containerized seedlings. The latter plots had received supplemental water due to periodic irrigation of seedlings and had last been watered nine days prior to soil sampling. Within each plot, duplicate samples were collected from each of three depths: 0-5 cm, 5-15 cm, and 15-30 cm. Each sample was homogenized separately, then a 117 cm^3 aliquot was taken from the homogenized sample and placed in an air-tight soil tin. Each sample was weighed prior to drying at 105°C for 8 days and immediately after drying.

Seed Bank Assessment

To determine whether a relict chaparral seed bank remained on the site, soil samples were collected from the research site in November 2012, prior to any winter rains. A soil core tool (6 cm in diameter) was used to obtain the samples to a depth of 4 cm. Soil samples were collected at 1-meter intervals along transects between plots. The samples from each block of plots were homogenized and passed through a 4 mm sieve to remove large rocks and pebbles.

To stimulate the germination of as many species as possible, soil was subjected to four treatments. One quarter of the soil from each block was exposed to heat (105°C for 5 minutes) then smoke water, one quarter was treated with smoke water only (no heat), one quarter was treated with 500 ppm gibberellic acid, and one quarter received deionized water (no treatment). Smoke water was generated by bubbling smoke through 20-liter jugs of water for one hour (Roche et al. 1997). Aliquots of approximately 120 cm^3 of soil were spread to a depth of 3 mm in flats on top of a 50:50 mixture of sand and seed starting mix and watered with either 55 mL of 10% smoke water, 55 mL of 500 ppm gibberellic acid, or 55 mL of deionized water. After 24 hours, flats were placed under an automatic mist system, receiving a mist of deionized water for 3 minutes every 8 hours.

As seedlings germinated, representatives of each species were transplanted into 4-inch pots for further growth and identification. Unknown species were identified using *T*he *Jepson Manual, Vascular Plants of California, Second Edition* (Baldwin et al. 2012).

In the field, smoke water was applied to the soil in an attempt to stimulate the germination of smoke-responsive seeds from any relict chaparral seed bank. Smoke water was applied to the soil at a rate of 1 L m^{-2} (Roche et al. 1997) on 17 Dec 2012.

Statistical Analyses

Differences in seedling survival among herbicide treatments were tested with a series of pairwise comparisons using Fisher's Exact Test. Species were analyzed separately. Due to low sample sizes, data from all three blocks were combined for this analysis.

Effects of block and herbicide treatment on plant size of survivors were analyzed with a two-way ANOVA (GLM ANOVA, MiniTab® 16 Statistical Software). Species were analyzed separately. To better test for herbicide effects on growth, we then compared sizes of only the ten largest plants of each species in each treatment. This approach eliminates the potential bias introduced by differential survival of individuals of different sizes (i.e., the tendency of smaller plants to die in more stressful environments and survive in more benign environments). Because this approach limits sample size, data from all three blocks were combined for each species and subjected to one-way ANOVAs followed by Tukey's tests (MiniTab® 16 Statistical Software).

Differences in soil water content across blocks, across herbicide treatments, and between restoration treatments (planted vs. no restoration treatment) were analyzed with three-way nested ANOVAs (GLM ANOVA, MiniTab® 16 Statistical Software).

Depth increments were analyzed separately, and duplicate samples were nested within plots.

Results

Impacts of Herbicide Treatment on Seeding Success

Seed application was not effective. Casual observation of animal disturbance indicated that some seeds, especially of *Quercus*, were lost to seed predation. The one seedling that was observed (an oak) died within a month of emergence. This seedling emerged in a fluazifop-only plot. No other seedling emergence was observed for any of the other species during the spring after seeding.

Impacts of Herbicide Treatment on Containerized Seedling Success

By mid-May 2013, mortality of *Adenostoma* and *Quercus* was lower in the glyphosate + fluazifop follow-up treatment than in the control or in the fluazifop-only treatments (fig. 1). Mortality among seedlings of *Rhus ovata* and *Eriogonum* was still quite low by mid-May, but effects of the glyphosate + fluazifop follow-up treatment were apparent in the growth of *Eriogonum* (fig. 2).

Because so few individuals of *Adenostoma* and *Quercus* survived in the control treatment, their growth was not analyzed. For all survivors of *Eriogonum* and *R. ovata*, two-way ANOVAs revealed no effect of block on plant size and no interaction between block and herbicide treatment. Comparison of the ten largest plants of *R. ovata* in each treatment revealed no significant effect of herbicide treatment on plant size ($p=0.421$, $df= 2$, $F= 0.89$; fig. 2). However, for *Eriogonum*, live canopy volume was much larger for the ten largest plants in the glyphosate + fluazifop follow-up treatment than for the ten largest plants in any other treatment ($p= 0.00$, $df= 2$, $F= 49.31$; fig. 2).

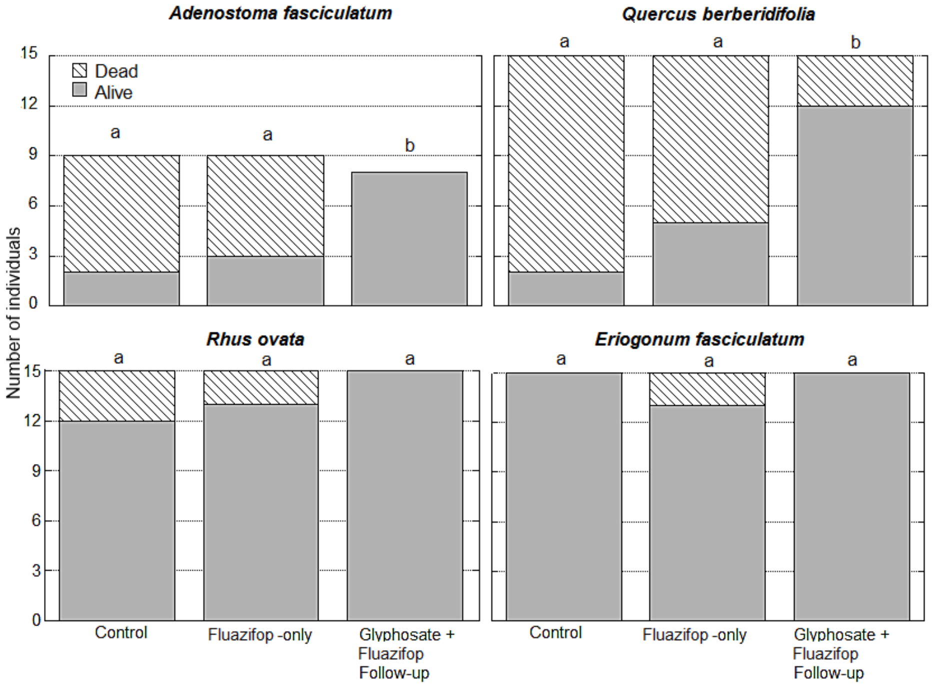

Figure 1–Survival of containerized seedlings in the different herbicide treatments. For each species bars sharing the same letter are not significantly different at p<0.05 according to Fisher's exact test.

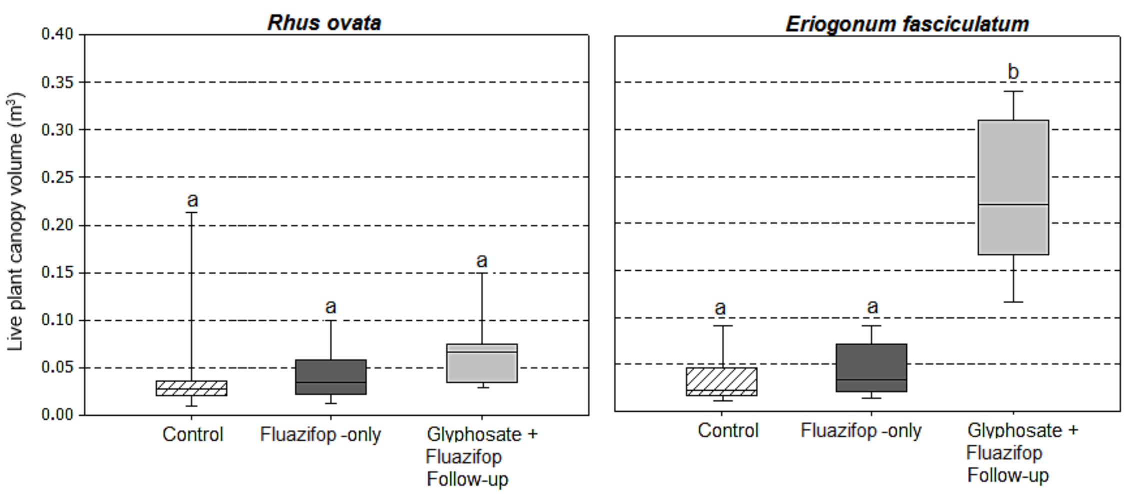

Figure 2–Live plant canopy volume (m^3) in mid-May of the 10 largest individuals from each treatment for *Rhus ovata* and *Eriogonum.* The medians, upper and lower quartiles, and ranges are shown. For each species bars sharing the same letter are not significantly different at p<0.05 according to Tukey's test.

Impacts of Herbicide on Gravimetric Soil Water Content

Gravimetric soil water content in the glyphosate + fluazifop follow-up treatment was significantly higher than in the control and in the fluazifop-only treatments (p<0.01; fig. 3, table 1). There were no significant differences in soil water content between control and fluazifop-only plots at any soil depth ($p < 0.05$, Tukey's test). There was no significant difference in gravimetric soil water content between control plots and those that were planted and given supplemental water at any soil depth (table 1).

Table 1—Three-way nested ANOVA results for gravimetric soil water content. Depth increments were analyzed separately

		0-5 cm		5-15 cm		15-30 cm	
Source	df	F	P	F	P	F	P
Block (B)	2	3.34	0.06	18.22	0.00	27.65	0.00
Planting (P)	1	0.00	0.99	0.27	0.61	0.00	0.95
Herbicide (H)	2	17.99	0.00	145.89	0.00	166.43	0.00
B x P	2	0.40	0.68	1.26	0.31	0.91	0.42
P x H	2	0.10	0.90	0.86	0.44	4.01	0.04
B x H	4	3.22	0.04	1.02	0.43	3.42	0.03
B x P x H	4	0.38	0.82	1.70	0.19	3.08	0.04
Error	17						
Total	34						

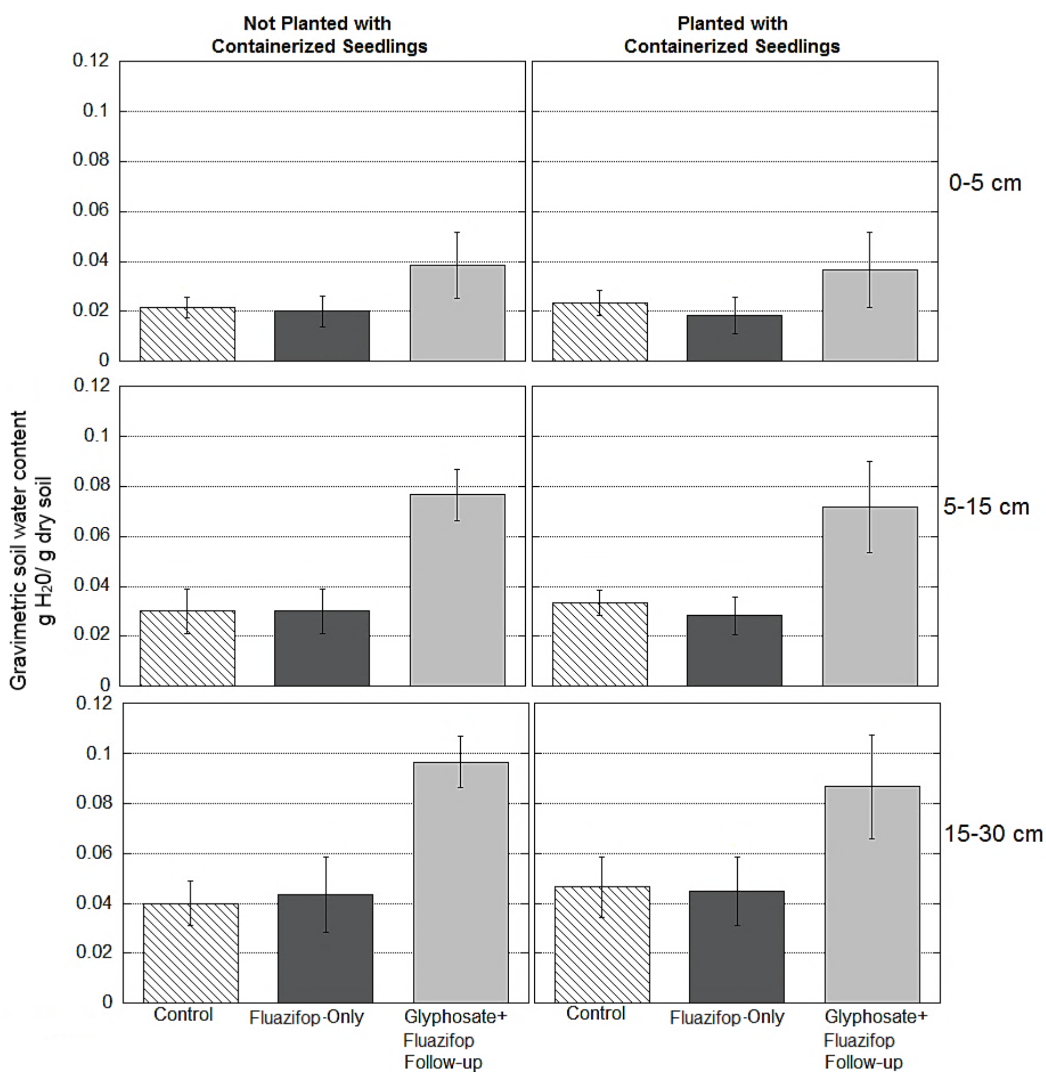

Figure 3–Gravimetric soil water content from soil samples collected in April 2013. Means and standard deviations are shown.

Seed Bank Assessment

The seed bank was dominated by non-native species (fig. 4). Those present were *Bromus diandrus* Roth*, Schismus barbatus* (L.) Thell., *Hirschfeldia incana (L.)* Lagr.-Fossat*, Erodium cicutarium* (L.) Aiton. and *Centaurea melitensis L*. Ruderal native species such as *Croton setigerus* Hook*, Amsinckia menziesii* (Lehm.) Nelson & J.F. Macbr. and *Gnaphalium* sp. were also present in the seed bank. Only two native shrubs, *Artemisia californica* Less. and *Eriogonum fasciculatum* Benth. were present in the seed bank. The weedy non-natives made up 58 percent of the assay. The ruderal natives comprised 24 percent, and other native herbs comprised 15 percent. Native shrubs comprised only 3 percent of the assay.

Germination of some species clearly benefited from soil treatments (smoke water, heat and smoke water, gibberellic acid), but no single treatment benefited all species (fig. 4). Application of smoke water to soil in the field did not result in the emergence of chaparral shrub seedlings or change the composition of the plant community appreciably.

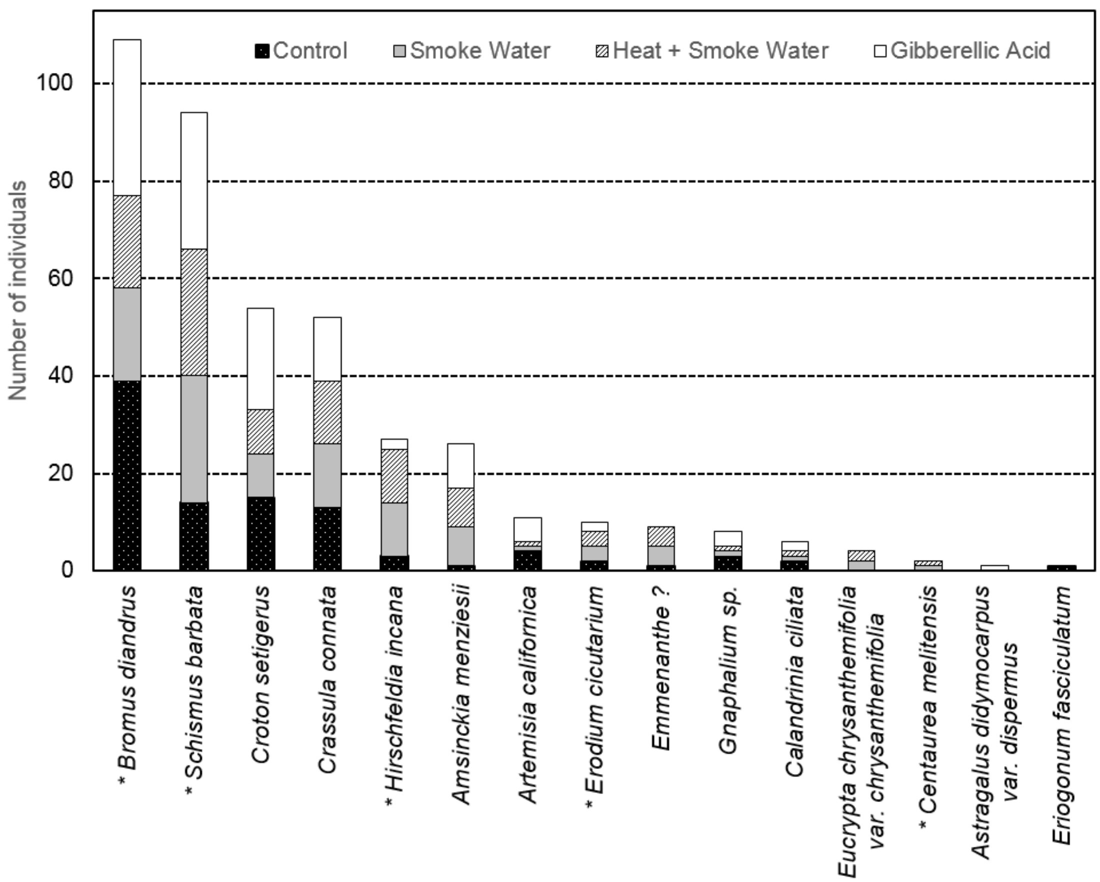

Figure 4–Seedling emergence from seed bank assay. Asterisk (*) indicates a non-native species.

Discussion

For chaparral restoration to be successful, weed populations must be controlled relatively early in the season. Glyphosate applications, applied early in the growing season, had many beneficial effects that fluazifop, applied later in the season, did not exhibit. Glyphosate-treated plots had higher soil moisture, which probably contributed to the higher survival and/or growth exhibited by shrub seedlings transplanted into these plots. Fluazifop alone, which was applied later in the year (March rather than December) did not result in higher springtime soil moisture content or transplant performance over control levels. Whether earlier fluazifop application would have increased soil water content or simply allowed broad-leaf weeds to use soil moisture is unknown. However, the timing of this study's fluazifop application was similar to that employed in previous studies in coastal sage scrub systems (Cox and Allen 2008, Marushia and Allen 2011), and this study's glyphosate treatment followed by a fluazifop treatment was more effective than the fluazifop-only treatment at promoting transplant success.

Although the glyphosate treatment increased the growth and/or survival of *Eriogonum*, *Quercus*, and *Adenostoma*, early-season growth and survival of *Rhus ovata* were uniformly high across all treatments. Whether the higher success of *Rhus ovata* is characteristic of the species or was due to the fact that transplants were from 2-gallon pots, rather than smaller 1-gallon pots, is unknown. However, some research has shown that survival of outplanted seedlings from deeper pots with longer roots is higher than that of seedlings from smaller pots (Burkhart 2006). Production of container stock of larger seedlings requires more time, effort, and advance planning however, so the effect of size on transplant success in restoration practice merits further investigation.

Seeding was highly unsuccessful compared to container transplants. Several factors may have contributed to that failure. Rainfall was limited: from the time of seed sowing to mid-May, precipitation was 41 percent of normal. Additionally, the presence of grass thatch may have contributed to the failure of the seeding treatment. Eliason et al. (1997) recommended that, for coastal sage scrub restoration, seeding should be accompanied by reduction in grass cover and thatch removal. In our study, thatch was not removed; its presence could have prevented seeds from making good contact with the soil or decreased light levels needed for germination. Whether seeding success would have been higher in a wetter year or with different site treatment is unknown. However, in this study, applying seed was much less successful than transplanting seedlings, a result that has previously been found in coastal sage scrub restoration (Eliason et al. 1997).

The relict native seed bank appeared to be insufficient for restoration of chaparral shrubs at our study site. As a result, attempts to manipulate that seed bank (e.g., with the application of smoke water to the soil in the field) were futile. This finding is consistent with previous work in coastal sage scrub that has shown that seed banks can be depleted rapidly by frequent fire, disturbance, and proximity to non-native annual grasslands (Cione et al. 2002, Cox and Allen 2008). Our study area had been degraded for a long period of time (estimated at 70+ years), and relict chaparral seed banks may be more intact in areas that have not been type-converted for as long. Geophytes present in the grassland at our study site (e.g., *Calochortus plummerae* E. Greene, *Dichelostemma capitatum* Alph. Wood, data not shown) merit preservation as understory components of chaparral. While their presence could constrain site treatments and the timing of herbicide application, early-season grass control (before geophytes emerge) may prove beneficial as it did for the chaparral shrub transplants in this study.

Conclusions

Chaparral restoration of type-converted areas may be feasible if the non-native annual grass competition is eliminated early in the season. Although it is too early to tell if transplanted seedlings will become fully established once supplemental water ceases, initial results suggest that transplanting shrub seedlings into herbicide-treated areas is more successful than seeding or attempting to manipulate a residual chaparral seed bank under the conditions of our study. Our study site had been occupied by non-native grasses for a substantial length of time, a factor that likely contributed to the paucity of a native soil seed bank. The year of our study was exceptionally dry, a factor that may have contributed to seeding failure and exacerbated early-spring grass competition. Under these conditions, a winter glyphosate application with a springtime fluazifop follow-up was more effective at eliminating non-native annual grasses and promoting transplant success than springtime fluazifop application alone. This combination of herbicide treatments, followed by planting of containerized seedlings, appears to be a promising approach to re-establishing chaparral shrubs on type-converted slopes.

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Shrubland and Woodland Restoration in the Mediterranean Basin[1]

V. Ramón Vallejo[2,3] and J. Antonio Alloza[2]

Abstract
Landscapes and ecosystems in the Mediterranean basin (MB) have been profoundly modified after a long history of land use and deforestation, which has caused extensive land degradation. In the last third of the 20th century, extensive land abandonment occurred in European Mediterranean countries, which dramatically increased fuel load and continuity in landscape and, consequently, large wildfires. As a response to land degradation, the forest services of MB countries already promoted large afforestations in the late 19th century. After the breakout of large wildfires and the new social perception of wildlands, new approaches for land restoration were called for. We present an approach and decision support system for assessing post-fire restoration. Short-term rehabilitation is considered to mitigate post-fire soil degradation and excessive runoff, while short- and mid-term restoration focuses on recovering keystone species. Long-term restoration is considered to recover the integrity of reference ecosystems and their services, together with a reduced fire hazard. The protocol and decision support system proved applicable to most of the characteristic vegetation types in the MB. However it is still uncertain how applicable they can be to other Mediterranean type ecosystems, such as the Californian chaparral.

Keywords: Mediterranean basin, resprouters, seeders, restoration, post-fire, shrubland, plantation, drought.

Introduction

Landscapes and ecosystems in the Mediterranean basin (MB) have been profoundly modified by long-term land use (Thirgood 1981), which often causes irreversible degradation. The main land transformation causes include cultivation, grazing (often overgrazing), fuel wood collection, forest overexploitation, mining and forest fires. Fire is known to form an essential part of Mediterranean plant evolution and natural ecosystem dynamics (Keeley et al. 2012). Therefore, most Mediterranean plant species present specific adaptations to withstand fires. However, excessively high fire frequency and extreme fire severity may overcome ecosystem fire resilience, especially when other disturbances contribute to ecosystem degradation. Recent widespread land abandonment in European Mediterranean countries has triggered wildfire occurrence (Pausas and Fernández-Muñoz 2012). Burned forests and shrublands are now the main subject of restoration projects in the MB. For that reason, this paper mostly focuses on post-fire restoration.

Afforestation of degraded lands is quite an old practice in the MB (fig. 1), and

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has become especially significant since the 19th century (Vallejo and Alloza 2012). The main afforestation objectives were watershed protection, increased forest areas and timber productivity, dune fixation and providing employment to marginal regions (Vallejo and Alloza 1998). The traditional management strategy of burned and other degraded areas in the Mediterranean region was based on afforestation with conifers, mostly pines. This strategy assumed that the restoration of degraded areas first required the introduction of pioneer conifers, followed by the introduction of late-successional hardwoods (Pausas et al. 2004). This traditional approach has been applied by default for decades in the MB to restore degraded ecosystems. However, the high cost to completely implement this strategy, and the changes in fire regime in the last few decades of the 20th century, have strongly compromised the effectiveness of this strategy.

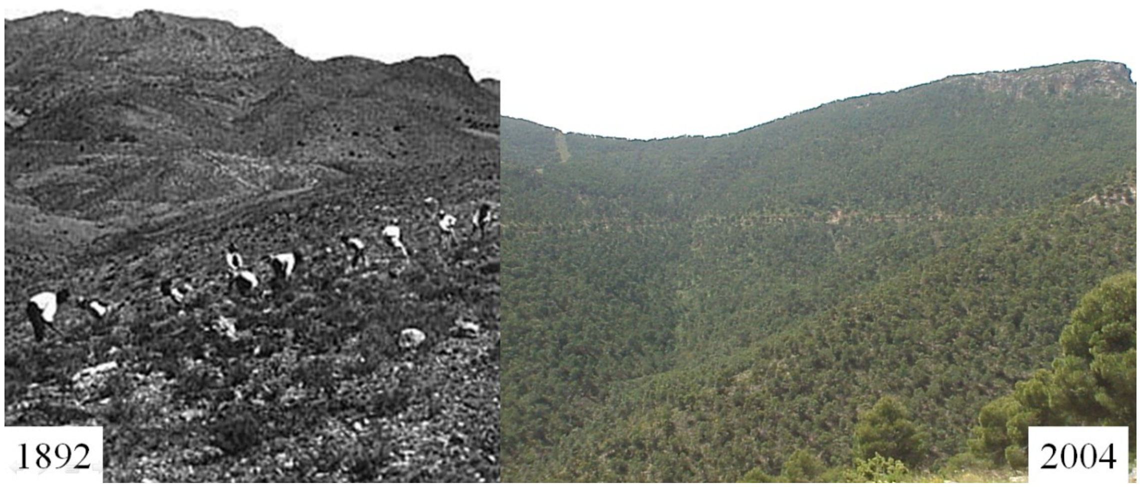


Figure 1—Example of old afforestation in Sierra Espuña (SE Spain). The project started in 1892 (left) after devastating floods caused by head-waters catchment deforestation. Pine plantations were successful in the long-term (right). Vallejo 2005.

In the present-day, new objectives for land restoration are added to traditional ones, such as increasing biodiversity, combating desertification, carbon fixation, fire prevention, and landscape recreational, cultural and aesthetical values (Vallejo and Alloza 2012). These new forest restoration goals should be reflected in the restoration strategies and techniques to be applied. In many Mediterranean countries, especially lowland forests with low timber productivity, fire prevention has become the first forest management priority. Advances in fire and restoration ecology in recent decades, along with the new social demands for preserving and improving ecological values, have led to new approaches in forest and shrublands management in general, and in post-fire restoration in particular. In this context, the definition of a restoration approach for a burned area must consider not only the expected ecosystem responses on a local scale, which will be determined by ecosystem type and by fire severity, but also the management objectives for the burned area (Moreira et al. 2012).

Ecosystem responses to fire are dependent on the regeneration strategy of dominant plant species (Vallejo and Alloza 1998). However, plant communities' response to fire is also dependent on the characteristics of the fire itself (e.g., fire intensity and frequency) and, what is even more unpredictable, the post-fire weather conditions. For one same vegetation type, different response patterns are expected for distinct fire intensities and severities (e.g., Bond and van Wilgen 1996, Belligham and Sparrow 2000, Moreira et al. 2009).

Specific forest management objectives can be extremely diverse depending on local ecological and socio-economic conditions. However, assuming post-fire land use changes is neither allowed nor considered, few baseline management objectives can be taken as a reference for most burned ecosystems in the MB (Vallejo and Alloza, 1998; Vallejo & Alloza 2012): 1) Soil conservation and water regulation. 2) Increasing fire-resilience in fire-prone ecosystems; 3) Recovering natural forests and tall shrublands.

In this article, we summarize the rationale and approaches for post-fire assessment and restoration in MB ecosystems developed by CEAM since the early 1990s in close collaboration with regional and national forest services. The background research has been mostly carried out in Eastern Spain (the Valencian Region) in dry sub-humid to semi-arid Mediterranean climates, including areas that have suffered large, frequent wildfires. The approaches have been also tested in other MB countries (Moreira et al. 2102, Vallejo et al. 2012).

Main Characteristics of Fire-Prone Vegetation in the MB

In the MB, wildfires affect mostly shrublands (more than 50% of the burned area), and pine woodlands and forests (Moreno et al. 2013). Most shrublands are successional, except those thriving in a semi-arid climate and in extreme habitats. Figure 2 provides a simplified overview of the main MB ecosystems types and their response to fire.

Hardwoods and sclerophyllous tall shrubs (fig. 3) are resprouters. Resprouting vigour can be very diverse depending on species, and sometimes on age and fire regime (Reyes and Casal 2008, Lloret and Zedler 2009). Most sclerophyllous MB trees and tall shrubs are extremely vigorous resprouters under current, and even severe (high intensity or frequency), fire regimes. Pine forests and woodlands are abundant in landscapes because of their good colonization ability and the extensive artificial plantations carried out since the 19th century. Most Mediterranean pines are obligate seeders, except *Pinus canariensis,* which is a vigorous resprouter and originally from the Canary Islands. Some pine species are very sensitive to crown fires (e.g.*, Pinus nigra, P. sylvestris, P. pinea*). Serotinous pine species *(P. halepensis, P. brutia, P. Pinaster,* dependent on provenance) usually regenerate after crown fires when trees are mature, but are sensitive to crown fires with short fire-return intervals (around 15 years, or shorter) that imply an immaturity risk; that is, the regenerate does not have time to reach sexual maturity and there is no seed bank for regeneration after the second fire (Pausas et al 1994).

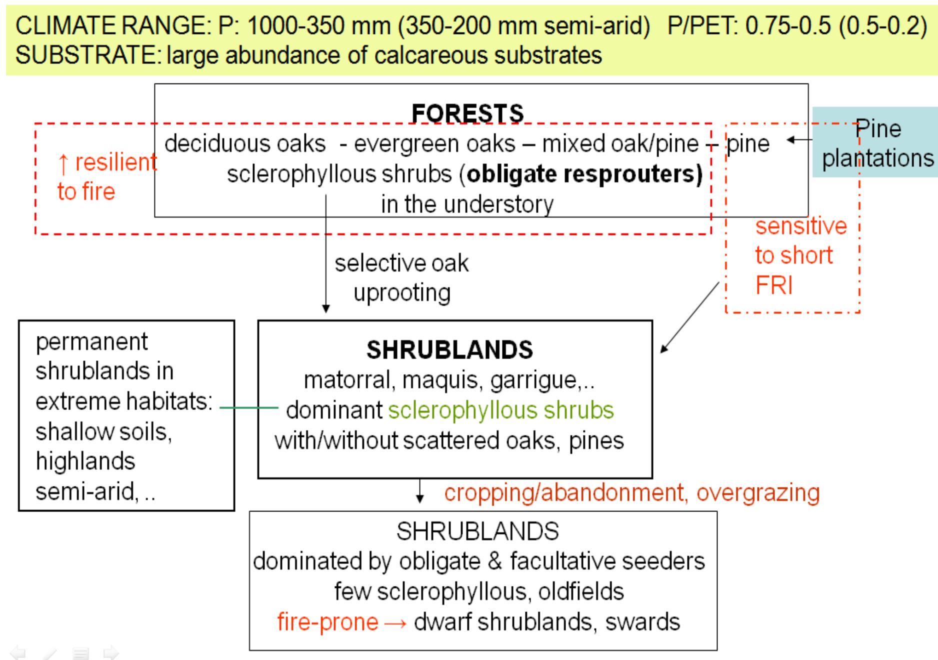


Figure 2—Simplified overview of the main ecosystem types affected by wildfires in the Mediterranean Basin and their dynamics. P: precipitation. PET: Potential Evapotranspiration. P/PET: Aridity index (UNESCO 1979). FRI: Fire Return Interval. More details in the text.

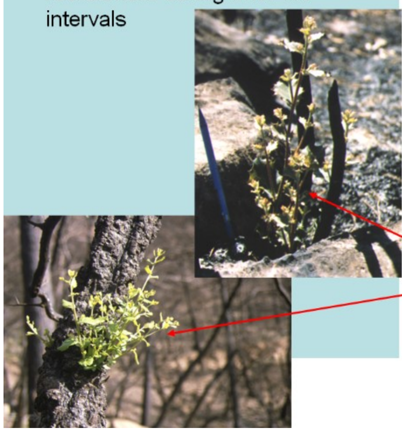
Permanent shrublands growing in a semi-arid climate are seldom affected by fire because of insufficient fuel load and continuity. If the climate is dry sub-humid, successional shrublands can group in those dominated by sclerophyllous species (fig. 3), maquis, kermes oak garrigue), which are vigorous resprouters, from those dominated by obligate seeders which are fire-prone and provide little protection to soil shortly after fire (fig. 4), Vallejo and Alloza 1998). These latter shrub communities especially develop in old fields (fig. 5) and may trigger short-term fire cycles, leading to ecosystem degradation loops. Therefore, recurrent fires may trigger catastrophic shifts in ecosystem state and resilience, especially in old fields and erodible soils.

Less disturbed, and never cultivated oak forests (in a dry sub-humid climate), are often considered reference ecosystems for restoration. Some, especially holm oak forests, were traditionally called "chaparral" in Spanish from Spain (deriving from the Basque language word txaparro, RAE 2001). The term chaparral refers to the low stature and shrubby form of trees, often resulting from coppicing and/or canopy fires.

In summary, knowledge of plant species' post-fire regeneration strategies is essential to predict fire impacts and to assess species selection in restoration programs which aim to increase ecosystem fire-resilience. A database of fire-related traits for species in the MB is already available (Paula et al. 2009) and a larger initiative is ongoing (Kattge et al., 2011).

MEDITERRANEAN BASIN SCLEROPHYLLOUS SPECIES

- **vigorous** post-fire obligate resprouters
- recruitment during inter-fire intervals



Genera/sp	Family
Arbutus	Ericaceae
Olea	Oleaceae
Phillyrea	Oleaceae
Pistacia	Anacardiaceae
Quercus coccifera	Fagaceae
Q. ilex	Fagaceae
Q. suber	Fagaceae
Viburnum	Caprifoliaceae
Rhamnus	Rhamnaceae

Figure 3—Representative MB sclerophyllous genera and species (for *Quercus*). Several of these genera are also native in North America. *Quercus suber* (cork oak) is the only species in the list to have insulating cork and to show epicormic resprouting after crown fires. The rest resprout from stumps and/or roots.

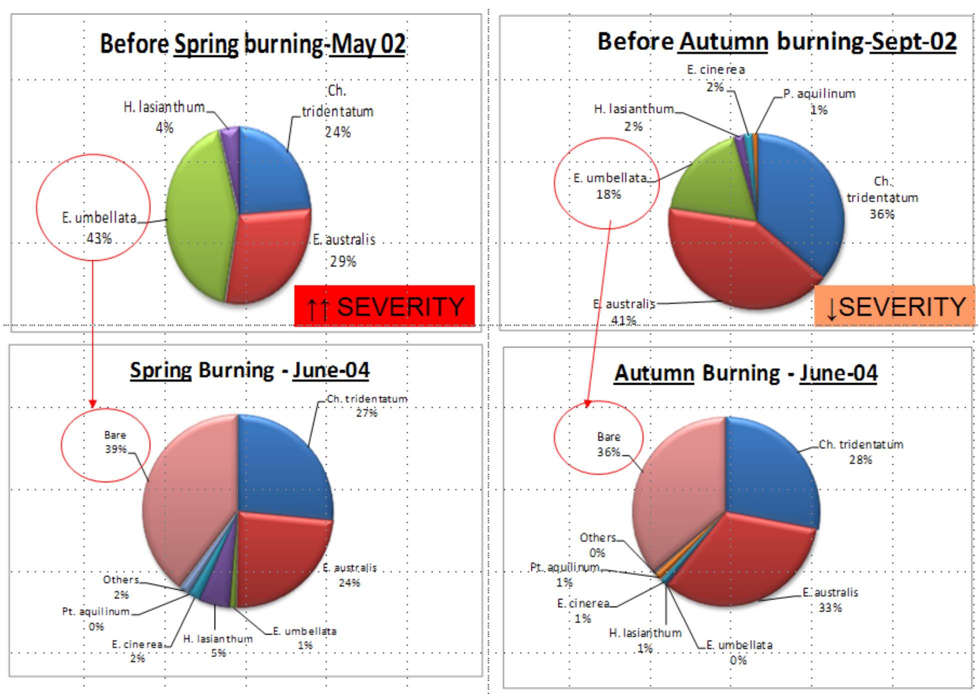


Figure 4—Shrubland regeneration after experimental fires set up in spring (high severity) and autumn (low severity), Gestosa (Portugal). The dominant resprouter shrubs (*Chamaespartium tridentatum* and *Erica australis*) quickly recovered their pre-fire plant cover. *Erica umbellata*, an obligate seeder, did not recover its pre-fire plant cover after 22-24 months since burning, which largely contributed to the persistence of bare soil (hence erosion risk) for both fire severities. Serrasolses (unpublished).

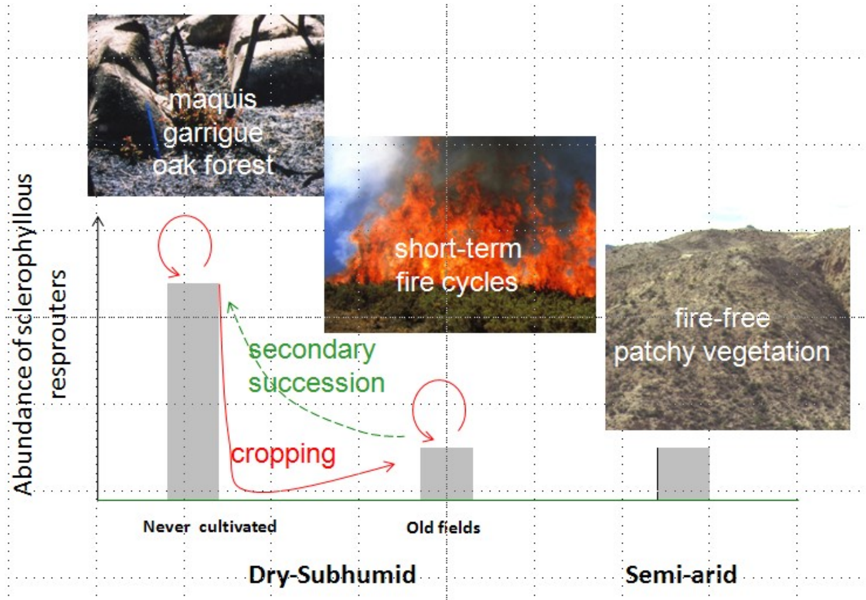


Figure 5—The role of cropping-abandonment in the abundance of sclerophyllous species in shrublands, woodlands and forests. Cropping drastically reduces the abundance of sclerophyllous species, which are poor colonizers to old fields. The dominant succession model after fire is autosuccession (Hanes 1971, Trabaud 1994); i.e., recovery of the same plant community after a relatively short time, a few years.

Assessing Post-fire Restoration Needs

Current restoration practices very much depend on the media impact of a given fire, which is related to fire size and social damages, and also on public budget availability. In an attempt to provide a comprehensive post-fire restoration strategy, and to optimise the investment of limited available economic resources, we have developed a protocol and a Decision Support System to help post-fire management decision making (Vallejo and Alloza 2012).

The main steps considered are:

- Defining management objectives, such as avoiding damage (erosion, flash floods), increasing fire-resilience and ecosystem/landscape biodiversity, and preventing new fires. According to the socio-economic and biophysical conditions of the burned area, more site-specific objectives can be considered.
- Identifying fire-vulnerable ecosystems
  - ⇒ Predicting runoff and soil erosion risk
  - ⇒ Predicting the regeneration capability of dominant species (resilience, regeneration rate) according to fire severity and to land and ecosystem characteristics
- Recommending specific techniques to mitigate degradation and to assist regeneration at different time steps.

The operational protocol includes four time steps according to timing of risks (fig. 6), Vallejo and Alloza 2012): 1) Preliminary GIS assessment of the vulnerable areas based on topography, vegetation and soil erosion risk maps; 2) Short-term assessment of soil erosion and excessive runoff risk based on Step 1 and the field survey immediately after fire occurrence – recommendation of emergency rehabilitation actions; 3) Short-term assessment of the regeneration of keystone species and possible recommendations to assist natural regeneration; 4) Long-term assessment of ecosystem restoration, often forest restoration, by considering the provision of ecosystem services and fire prevention. Long-term restoration should include climate change projections according to the state-of-the-art knowledge on the projected changes in climate and the fire regime, and also on the species adaptation potential to change. All the operational steps, from 2 to 4, include the quality control of restoration works (stewardship during implementation), and monitoring and assessment according to the set objectives. The objectives, performance standards and protocols for monitoring and for data assessment should be incorporated into restoration schemes before a project starts. Post-fire monitoring and assessment is essential to gain an understanding of forest ecosystems' post-fire successional pathways and, accordingly, to plan appropriate restoration actions. It will also allow the re-direction of restoration actions in an adaptive management framework (Vallejo & Alloza, 2012). The participation and involvement of relevant stakeholders in all the project phases is critical to help incorporate local knowledge and to stimulate adoption.The protocol has been fully developed using standard data collection forms and general guidelines to assist forest managers in post-fire management (Alloza et al. 2014, in Spanish).

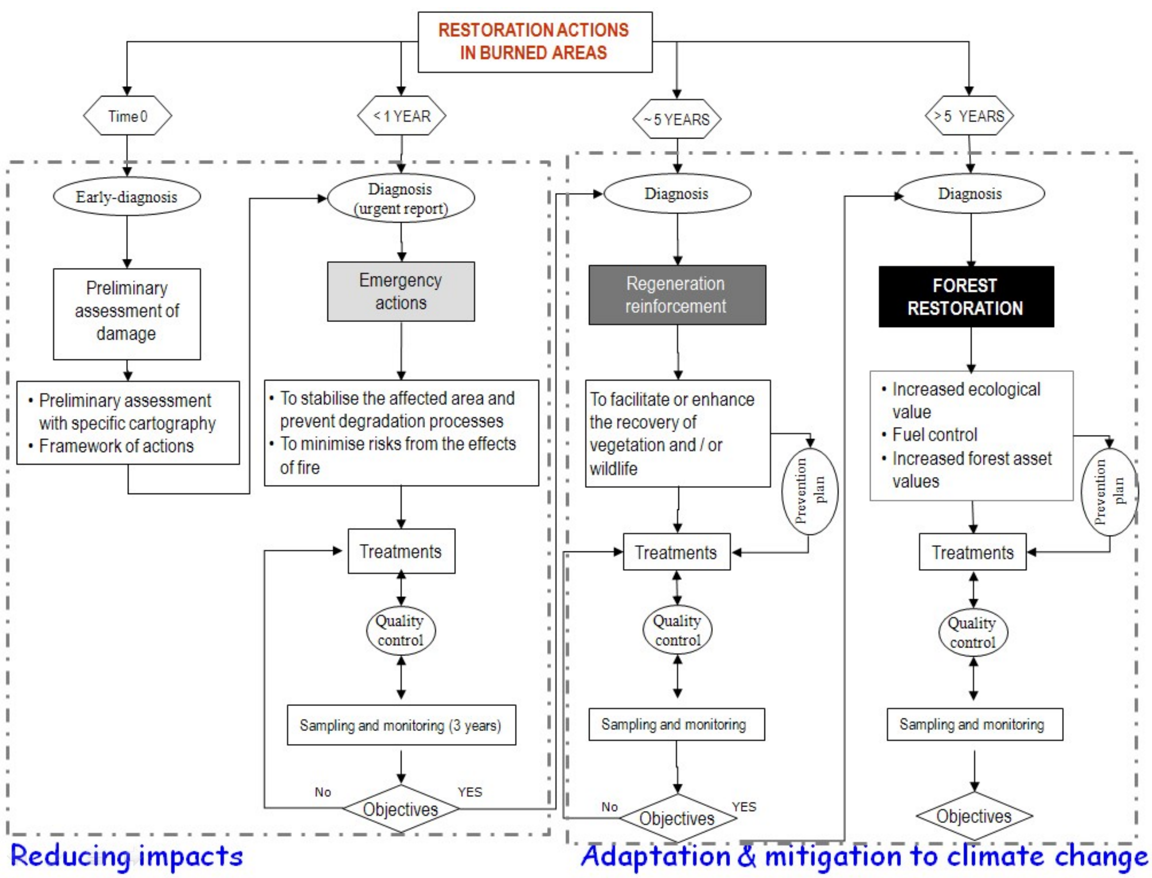


Figure 6—Scheme for assessing post-fire restoration in the Mediterranean Basin. Details in the text.

Post-fire Decision Support System (POSTFIRE-DSS)

Based on the previous protocol, we have developed a Decision Support System to help post-fire decision making for MB conditions (Vallejo and Alloza 2012). The application, called the POSTFIRE-Decision Support System, includes four steps: 1) Identifying vulnerable areas, 2) Assessing fire impacts and emergency interventions, 3) Short- and mid-term planning and 4) Long-term planning. All the steps are not necessarily required depending on the stakes focussed on, the general objectives and level of risk,

Identifying Vulnerable Areas

Forest and land managers need tools to identify priority areas for fire prevention and post-fire intervention. Using a Geographic Information System (GIS) and thematic cartography, vulnerable areas can be mapped on the basis of assessing not only the potential regeneration capacity of the vegetation, but also the post-fire degradation risk (Alloza and Vallejo 2006; Duguy et al. 2012, (fig. 7). The main factors considered are: 1) Estimating the vegetation regeneration capacity by using the combination of autosuccession potential (the ability to recover the pre-fire vegetation type) and the plant recovery rate, which determines how quickly plant cover will recover to protect soil against excessive erosion and runoff risk; 2) Assessing the short-term degradation risk based on the potential soil erosion risk; 3) Combining the regeneration capacity and soil degradation risk to produce a map of ecosystem vulnerability that enables the identification of priority areas for pre-fire prevention and post-fire intervention when a wildfire occurs. This type of approach can be used on different scales; for example, in the European Union (Duguy et al, 2013) or on a regional scale (Duguy et al. 2012).

Assessing Fire Impacts and Emergency Interventions

The ecological impact of a fire partly depends on fire severity. It is critical to assess severity levels as soon as possible after fire. This can be done by either field surveys or using high resolution remote sensing, or a combination of both. Several published guidelines can be used to quickly assess fire severity and to identify areas for emergency interventions (e.g., USDI, 2003; Napper, 2006; Lutes, 2006; Pike & Ussery, 2006; Stella et al, 2007; Robichaud and Ashmun, 2012). The field survey protocol has been developed using standard forms (Alloza et al. 2014) to assess fire impacts. Depending on site conditions and fire impacts, emergency interventions may be required.

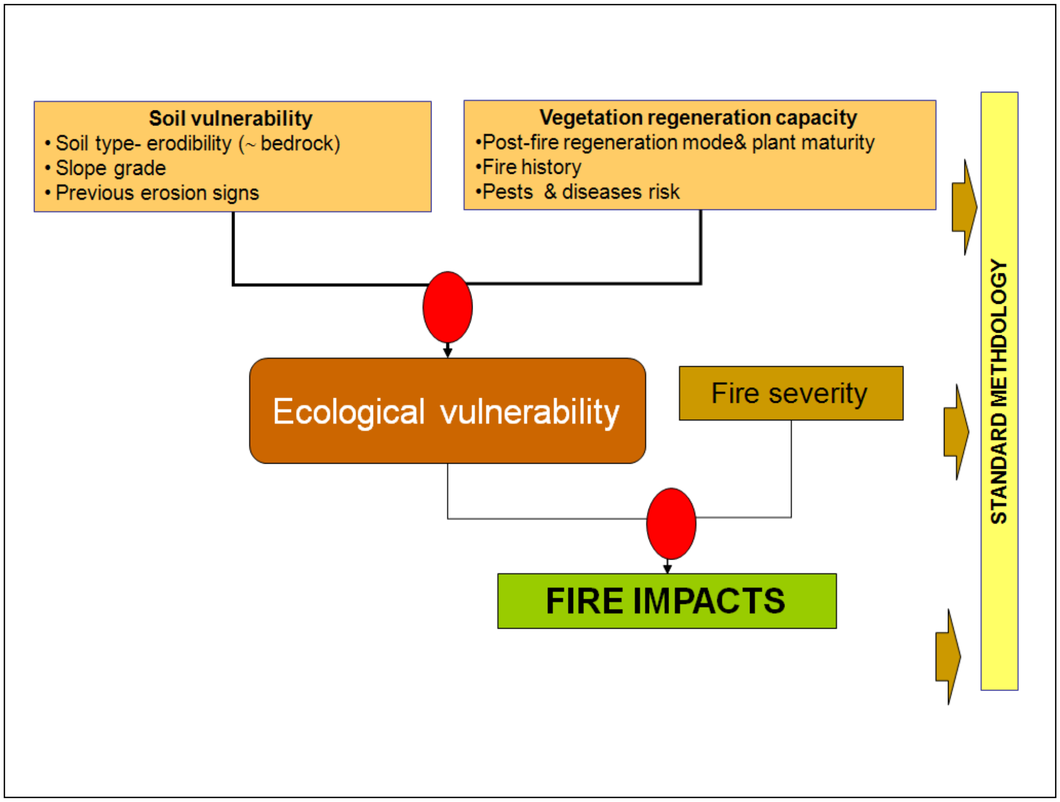


Figure 7—Outline of the criteria used to assess ecological vulnerability and fire impacts.

Emergency interventions, sometimes called first-aid rehabilitation, aim to stabilize the affected area, to prevent degradation processes and to minimize risks for people (Robichaud et al. 2000). They focus on soil protection to avoid erosion and to decrease water runoff and risk of flooding, to reduce risks to people and property (e.g., hazard from falling burned trees), and/or to prevent tree pest and disease outbreak. They should be carried out as soon as possible, in a few weeks, or a few months after the fire at the most, and preferably before the first (often heavy) autumn rains in the Mediterranean region. In principle, early emergency approaches should not essentially be modified because of climate change as only moderate rain intensities immediately after fire may produce already serious erosion, whereas mid- to long-term restoration should very much consider climate change projections.

The Decision Support System displays a screen with the variables needed to characterize site conditions and fire characteristics. Variables are grouped into four blocks: site conditions, vegetation (composition and cover) before fire, fire severity, and early post-fire soil conditions (fig. 8). The most unfavourable site conditions are: 1) high risk of heavy autumn rain; 2) poor resprouters cover before fire; 3) erodible soils; 4) steep slopes. These risk situations can be exacerbated for high severity fires.

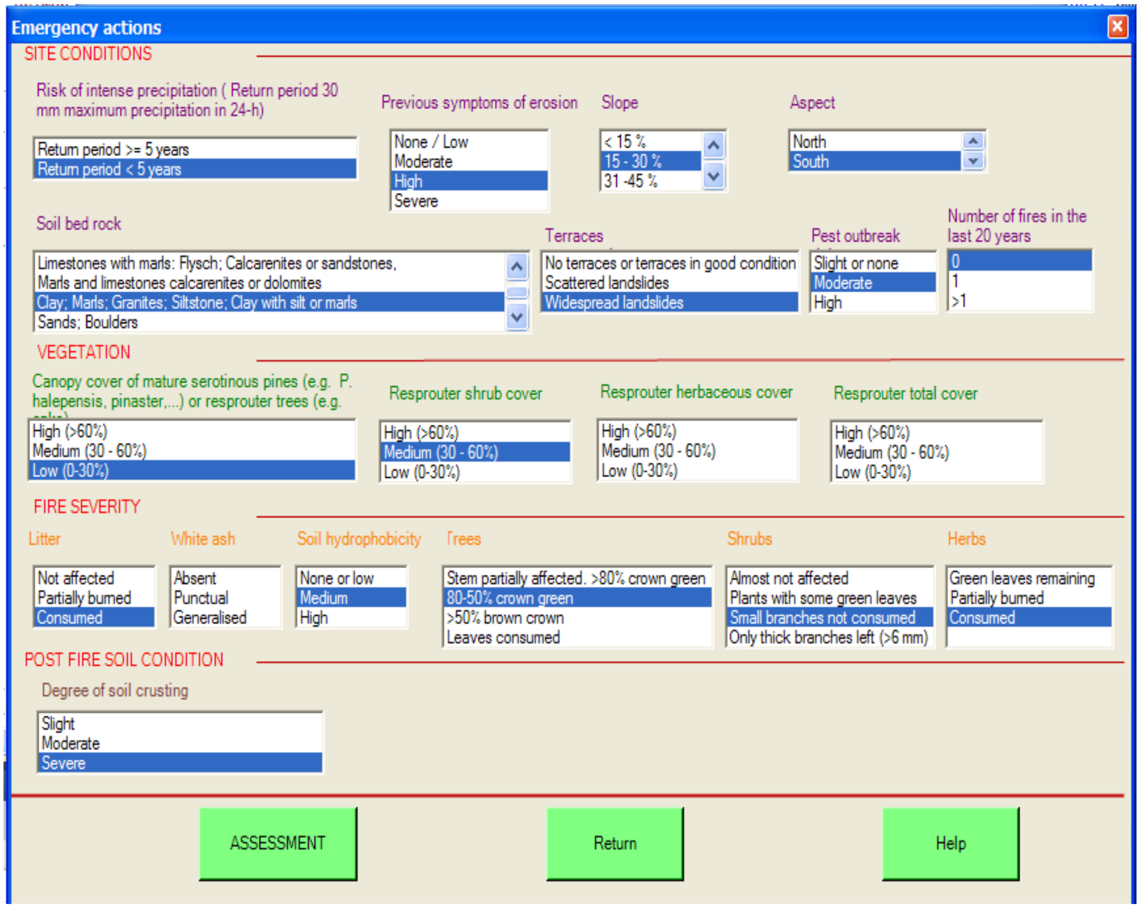


Figure 8—Screen showing the variables and ranges considered for the assessment of short-term fire impacts and the need for emergency rehabilitation actions (POSTFIRE-DSS).

Short- to Mid-term Planning (2-5 yrs.)

Short- to mid-term interventions aim to facilitate the natural regeneration of keystone species (trees and tall shrubs) and to reintroduce the key species that have been eradicated by fire or their abundance has been severely reduced. The application requests information about the presence and abundance of resprouter species, and on the density of natural regeneration of tree species. The criteria used in the assessment are shown in table 1.

Long-term Planning (>5 yrs.)

This is related to long-term ecosystem restoration, in accordance with the established management objectives (Vallejo et al., 2009). In general, long-term restoration actions are needed when the ecosystem's resilience capacity has been vastly altered and cannot be naturally recovered. In the MB, this is often caused by an unprecedented combination of fire regime and other disturbances. The long-term perspective should attempt to recover ecosystem integrity in accordance with ecological restoration concepts. In addition, as fire hazard is inherent in the

Table 1—Criteria and recommendations used in the regeneration assessment

Regeneration	Cover/Density	Assessment	Recommendation
Woody resprouters cover (shrubs and trees)	> 60%	Good plant cover regeneration expected	No vegetation reinforcement required for increasing fire resilience
	30-60%	Intermediate plant cover regeneration	Reinforcement with woody resprouter plantation
	< 30%	Poor plant cover regeneration	Containerized seedling plantation with resprouter species.
Tree density	> 3000 seedlings/ha	Overstocking: density control with selective thinning.	In pine forests, thinning to reduce fuel accumulation, to increase seed production and limit crown fire propagation. In broadleaved forest (coppices), cleaning stools to promote high forest structure
	1000-3000 trees/ha	Good tree species regeneration	Do nothing.
	200-1000 trees/ha	Poor tree species regeneration	Open woodland regeneration.
	< 200 seedlings/ha	Very poor tree regeneration	Containerized seedling plantation of tree species, often hardwoods and conifers combined.

Mediterranean, fire prevention principles should be incorporated into post-fire restoration strategies on ecosystem, and especially, on landscape scales. Landscape ecology principles should be considered when designing new plantations.

This long-term planning can have diverse objectives in accordance with the specific socio-economic and ecological conditions of the affected area. Depending on the situation, restoration may include type-conversion to other forest types, afforestation or reforestation and, in general, promoting ecosystem services in the socio-ecological systems context.

Implementation

Wildlands in the Mediterranean have been negatively selected by humans for centuries because the most productive soils were devoted to agriculture and pasture. Soils in wildlands are usually shallow, stony and develop on steep slopes and crests (Vallejo et al., 1999). Only in recent times have abandoned terraced lands presented better, deeper soils, made available to recolonize natural vegetation and for restoration purposes.

The restoration of degraded lands should take into account poor soil productivity and the characteristic drought occurrence of the Mediterranean climate, which is expected to worsen in the near future (IPCC 2012). Therefore, when the reintroduction of keystone plant species is considered in restoration, containerized seedling plantation should be designed to overcome transplanting shock. This applies to both post-fire restoration and the restoration of degraded lands caused by other disturbances. Indeed, drought is the main cause of seedling mortality for plant recruitment and in plantations (Vallejo et al., 1999).

Technical options to reduce water stress in plantations include (Chirino et al. 2009; Vallejo et al. 2012b; Vallejo et al. 2012c): 1) Species and provenances selection, including seed collection and conservation quality criteria; resprouting sclerophyllous tall shrubs and trees is preferred because of their high post-fire regeneration capacity, and also because of their low rate of risky fuel upload as compared to seeder shrubs. Species and provenances are selected by considering their water use efficiency. 2) Nursery cultivation techniques to produce high quality seedling acclimated to overcome transplanting shock. In our experience, the most successful techniques were drought preconditioning, the use of deep containers for species developing tap root (e.g., *Quercus* species), and the addition of hydrogel in the culture substrate to provide extra water supply immediately after plantation. 3) Soil preparation and amendment focus on improving water availability to seedling (Valdecantos et al., 2014). 4) Plantations using several native species, in contrast to traditional mono-specific pine plantations; 5) Tree-shelter for shade-tolerant seedlings. Tree shelters reduce transpiration demands, protect seedlings from small and domestic herbivores, and generate a shaded environment during the first years after outplanting. 6) Extant vegetation treatment: facilitation or competition? Frequently, extant shrubs have a nurse effect on introduced seedlings (for example, see Castro et al. 2002). However, this is not always the case and contrasting results may be obtained depending on site conditions (Maestre and Cortina 2004; Maestre et al. 2003). 7) Spatial configuration of plantations according to the spatial distribution of microsites, and also to landscape ecology and fire prevention principles; 8) Post-plantation care: in principle, it should be reduced to as much as possible to minimize costs.

Concluding Remarks

The post-fire restoration assessment approach presented herein has proven applicable to representative fire prone ecosystems in the Mediterranean basin. It is still uncertain how applicable it might be to the other Mediterranean type ecosystems (MTE) in the world. In spite of the many common climate characteristics and plant adaptations in the various MTEs (Keeley et al. 2012), particularly between the MB and California, both of which share common Laurasia plant lineages, many differences appear in relation to land use history, the impact of disturbances affecting the fire regime and degradation processes (Vallejo et al. 2012b) and, hence, affect restoration needs and the associated technologies. For example, the long-term extensive land use history of the MB, and the recent widespread land abandonment in the European MB, are causing an unprecedented modification of ecosystems, fuels and the fire regime. Therefore, given the direct impacts on landscapes and ecosystems, and the indirect impacts on the fire regime, land use history can be considered the key driver to primarily shape MB landscapes. This is quite different in at least three of the other MTEs, but is no so different in Chile. Conversely, invasive exotic species are a major issue in most MTEs, but not so much in the MB, especially in relation to fire regime: MB fire-prone ecosystems are generally slightly invasive (so far) (Vallejo et al. 2012a). Finally, MB countries share a long-standing tradition of active and extensive afforestation to improve degraded lands, which could somehow be assimilated to the modern ecological restoration concept. This tradition is still probably influencing the proneness of MB policy makers and forest services to promote restoration projects, including plantations.

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Restoration in Type-Converted and Heavily Disturbed Chaparral: Lessons Learned[1]

Katie VinZant[2]

Abstract

A wide range of disturbances including recreation, utility corridors, too frequent fire and invasive plants have led to the type conversion of native chaparral habitats to non-native grasslands on the Angeles National Forest. These often compounding disturbances have led to the recent planning, implementation and monitoring of a wide variety of restoration techniques across the forest. Restoration techniques have included imprinting, hydroseeding, container planting and aggressive weed control with a variety of results. Factors influencing these results are adequacy of planning, timelines, weather patterns, hydrology and soil conditions, pre-project invasive plant infestations, availability of materials and personnel, and as always, cost. The highest performing restoration sites appear to be those that either had a low presence of non-natives beforehand or had aggressive weed control in the first two/three years of restoration, combined with the use of enough genetically appropriate, early seral native plants introduced to the site as container plants at the correct time in the early fall and watered consistently.

Keywords: chaparral restoration, Angeles National Forest, topsoil salvage, soil decompaction, invasive plant removal, hydroseeding, container planting

Introduction

The Angeles National Forest (ANF) is home to over roughly 2000 different plant species, spread across roughly 80 vegetation types, with the dominant type being chaparral (Stephenson and Calcarone 1999). In addition to this plant biodiversity, the ANF is crisscrossed by hundreds of miles of pipeline and powerline rights of way (ROW's), utility and recreational roads, and trails. All of this infrastructure, which services Los Angeles and its 10 million residents (U.S. Census Bureau 2014), has contributed to vegetation disturbance in the form of invasive plant proliferation, soil erosion and compaction, vegetation cutting or removal, dust pollution, increased fire intervals and increased cross-country unauthorized vehicle travel. The effects of these disturbances, especially when combined, have led to the vegetative type conversion of native chaparral habitats into stands of non-native annual grasses or forbs.

Prior to 2007 very little historical evidence can be found for chaparral vegetation restoration activities occurring on the ANF to mitigate the impacts of this infrastructure construction and maintenance (J.Nickerman, personal communication, November 6, 2014). However, around 2007 it was recognized that the habitat fragmentation and degradation caused by infrastructure across the ANF needed to be mitigated in the form of onsite vegetation restoration. The first challenge was

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creating a comprehensive restoration plan that would document the strategy for enacting restoration principles on the ground. A template for such a restoration plan has now been developed.

The intent of this paper is to describe the processes of chaparral restoration planning on the ANF by detailing the methods employed and the lessons learned over 100 different sites across the Castaic and San Gabriel Mountain ranges from 2009 to 2014. Approximately 40 of the sites had restoration started in 2009, 10 in 2010, 10 in 2011, 10 in 2012, 25 in 2013 and 5 in 2014. Sites vary in size from 0.06 to 5 acres, cumulatively amounting to approximately 75 acres. Elevations range from 2000 to 7000 feet, with annual precipitation ranging from 10-15 inches in the Castaic range sites and 30-35 inches in the San Gabriel range sites (Stephenson and Calcarone 1999). The sites primarily occur in vegetation dominated by chamise chaparral, semi desert chaparral, California buckwheat (*Eriogonum fasiculatum*) scrub, scrub oak chaparral, mixed chaparral, oak woodland and non-native annual grassland. However, some work was also done in cottonwood/willow riparian woodlands as well. All of the sites were scheduled to be restored due to utility or road construction or maintenance disturbances.

Restoration Methods and Lessons Learned

Restoration sites were evaluated through the use of aerial photographs and multiple field visits. The seed palettes (described in the paper “Guidelines for Choosing a Plant Palette and Collecting Native Plant Materials” of this General Technical Review (GTR)), erosion control best management practices (BMPs), and restoration methods were tailored to the soil conditions, slope, aspect, elevation, hydrology, vegetation and severity of disturbance at each site. The following headings highlight the main components that must be considered when planning for chaparral restoration. Each heading is divided into two sections: the “Methods” portion and the “Lessons Learned” portion. The headings are discussed in the order in which they should be considered for successful restoration planning.

The information provided below is derived from observations over the large variety of restoration sites described above, conversations with restoration practitioners and technical restoration literature. Data are not analyzed statistically because the main goal was to get native plant cover back on the sites, not to conduct rigorous and extensive monitoring. Therefore, the reader should note that the restoration recommendations provided below in the “Lessons Learned” sections are primarily anecdotal in nature. However, starting in 2013, the ANF, in cooperation with partners, has begun to record statistically valid baseline data prior to site disturbances with the goal of monitoring restoration trends (typically 7-10 years).

Topsoil and Vegetation Salvage and Replacement—

Topsoil and vegetation salvage are the first steps in the restoration site preparation process as they must occur before a site is disturbed. They are only needed, however, if the site will be disturbed by activities that remove or alter the soil horizons and existing native vegetation (e.g. grading, compaction). If these disturbances do not occur, this step in the restoration planning process would be skipped.

Methods—

In an effort to preserve seed banks, native mycorrhizae and soil nutrients, the top 5-16 inches of soil were salvaged in areas scheduled for ground disturbing activities. Heavy equipment (bulldozers, excavators, etc.) was used to carefully scrape off the topsoil and move it in a designated area on site. Topsoil salvage on the ANF has included the O, A, and B soil horizons which were salvaged together given since these horizons are typically thin (less than a few inches). Topsoil was salvaged in a dry state in the late summer to early winter when most plants had finished seeding and seeds in the seedbank were dormant. Every effort was made to avoid salvaging soil in the late winter to spring, as salvaging at this time would have the most deleterious impact on the quantity of seeds in the seedbank. At this time of year seeds germinate and topsoil salvage would likely destroy them, in addition to the offspring they would have produced that season, therefore even further depleting the amount of seed in the seedbank. Since the viability of the beneficial seeds, fungi, bacteria, and nutrients in topsoil decreases with time (Steinfeld et al. 2007) topsoil was for the shortest amount of time possible (still ended up being over 12 months in most cases).

Topsoil stockpiles were protected to maximize recoverability and to minimize the effects of wind and water erosion. This was done by placing topsoil piles in wind-rows around the perimeter of the construction area. Piles were less than five feet in height, in an attempt minimize anaerobic conditions in the soil. Piles were cordoned off from construction activities so they were not disturbed by vehicles or equipment. Attempts were also made to remove all nonnative plants from the stockpiles before they produced seeds that could contaminate the soil. In order to minimize erosion jute or coir netting covered the stockpiles, but plastic covering was used since it sterilizes the seeds and microbes in the topsoil (especially problematic in warm months). Wattles or silt fences were also installed around the stockpiles to act as a barrier and prevent erosion.

Vegetation growing in the salvaged topsoil was either incorporated into it (if it was less than 2-3 feet tall and not robustly woody in nature) or salvaged into its own stockpile (if it was over 3 feet tall or woody) by using heavy equipment before the topsoil was piled.

Once construction, slope recontouring and decompaction (both described in section below) were completed, the topsoil was replaced with a bulldozer and excavator working in tandem. Topsoil was re-spread to a uniform depth comparable to soil site conditions in adjacent undisturbed areas. The topsoil surface was left in an uneven/roughened condition and the bulldozer then track-walked over the soil perpendicular to the site contours, but not to the point of creating soil compaction over 85%. Usually two to three passes with the dozer were sufficient. Track-walking created small depressions in the soil which captured water, soil and seed, thus helping to stabilize the site. In addition, the dirt clod size was kept at 2–3 inches to create a roughened surface that maximized soil/seed contact. Topsoil was not spread with a substantial use of water (only a fine misting was used to keep dust to a minimum), especially during the months of April through August. The goal was to minimize compaction while at the same time preventing seed germination during the driest times of year.

After topsoil was replaced, native woody vegetation not incorporated into the topsoil was placed back over the site either as chipped material, slash, or vertical mulching to discourage unauthorized vehicle traffic.

Lessons Learned—
Well in advance of construction activities (preferably at least 6 months) areas suitable for topsoil and vegetation salvage should be delineated. This will allow for adequate equipment mobilization and storage design.

- Top soils in the San Gabriel and Castaic Mountains tend to be very thin (less than 6-12 inches), thus making separation of the O and A horizons from the B horizon infeasible with heavy equipment. However, every site is different and a site-by-site determination should be made as to whether it is necessary and/or possible to separate the horizons, as salvaging results may be enhanced.
- Do not salvage topsoil that has more than 25% non-native plant cover.
- Do not salvage topsoil that contains a highly noxious weed such as perennial pepperweed (*Lepidium latifolium*) or yellow star-thistle (*Centaurea solstitialis*). When topsoil was salvaged and re-spread in areas with greater than 25% non-native cover or that had a highly noxious weed present the soil was recolonized by non-natives, making it harder to restore the native vegetation.
- Carefully consider if the salvage of topsoil is worth causing a larger footprint of disturbance. On the ANF topsoil was stored on site, as storing topsoil offsite and then bringing it back was cost prohibitive. This meant that the disturbed construction area had to be slightly larger to accommodate the topsoil piles.
- Use smaller sizes of dozers (e.g. D5/D4 CAT) and excavators for topsoil and vegetation salvage work. Larger equipment is not suitable for this detail-oriented work.
- Salvaging topsoil on slopes over 30% is difficult because heavy equipment does not perform well on steeper terrain.
- Ensure that topsoil piles are well flagged and signed so that disturbance is avoided.
- Avoid using jute or coir to cover piles if hand weeding is necessary as it is very difficult and time consuming to remove all of the roots and see invasives through the woven covering.
- The best time of year for topsoil replacement is August through November, because the topsoil seedbank receives the maximum amount of rainfall to stimulate germination. However, replacement in December through February has been moderately successful if a substantial rain event (over 1 inch) occurs post-replacement.
- It is very important to ensure track-walking over the site is enough to hold topsoil/subsoil in place, but not so much that the site is compacted. Track-walking must be done perpendicular to the contours, otherwise the depressions from heavy equipment tracks will channel water/soil down slope leading to rilling or even slope failure.
- Do not add fertilizers to the topsoil. Fertilizers have been shown to increase weed abundance (Newton and Claasen 2003, Steinfeld et al. 2007).
- Non-local mycorrhizae were added to approximately 15 of the restoration sites. Based on field observations the added mycorrhizae did not appear to increase or decrease the amount of plant cover. This may be because 1) the added mycorrhizae were not local and so were not effective or 2) the amount or type of mycorrhizae added was not adequate to increase plant productivity or 3) the mycorrhizae was added to the soil incorrectly.

- To maintain the effectiveness of vertical mulching or slash it may need to be reset or even replaced as it can blow down/away or be crushed by motorists. For this reason it is best to combine mulching/slashing with boulders and/or container plants, which are more effective over the long term.
- Vegetation slashing or mulching may hinder weeding (especially if herbicide is not the eradication method) by making weeds hard to reach and easily camouflaged.

Slope Recontouring and Decompaction—

Slope recontouring and then decompaction are the next steps in the restoration site preparation process (after topsoil/vegetation salvage and replacement). Similar to topsoil and vegetation salvage, slope recontouring is only needed on a restoration site if the original landscape contours have been altered and returning the contours would improve erosion control, hydrologic function and/or scenic integrity. Likewise, soil decompaction is only necessary if the soil horizons have been compacted to the point that plant germination and growth would be inhibited (which is typically only caused by the repeated use of heavy equipment on the site). Decompaction is most often needed when restoring roads, trails or other graded areas.

Methods—

Disturbed areas that had the native surface altered were recontoured using a bulldozer and an excavator as the first step in the on-site restoration process. Recontouring was done by referring to pre-disturbance photos or by comparing the disturbed site to the surrounding terrain. Special care was taken to re-incorporate drainage features.

After recontouring was completed, restoration sites were assessed for soil compaction which would likely inhibit vegetation recolonization. About a third of the restoration sites disturbed areas were tested using a static cone penetrometer, or similar instrument, and compared to several undisturbed, adjacent locations. Decompaction was deemed necessary when the disturbed area range showed more compaction than the undisturbed area.

If restoration areas were compacted a bulldozer, excavator or backhoe was used to loosen the soil to mimic undisturbed conditions. The most common decompaction technique utilized was soil ripping with tines (such as those on a bulldozer) or the teeth on an excavator or backhoe. Ripping occurred prior to topsoil placement, with dry soils, to an average perennial plant rooting depth (~2 ft.), along the contour line of the site. In areas where topsoil was not salvaged but instead mixed with subsoils light track-walking (2-3 passes) with the bulldozer was performed perpendicular to the contours of the site directly after ripping. If the site did not experience any topsoil grading, but decompaction would be beneficial, then great care was taken to not mix the topsoil with lower soil horizons. If there was a potential for soil erosion to occur, erosion control measures such as hydromulching, wattles, wood fibers, or jute netting were installed after recontouring, decompaction and/or topsoil placement.

Lessons Learned—

- Ensure that ripping for decompaction is done in line with the contours of the sites slope. If ripping is done perpendicular to the contours it will most likely lead to soil erosion as water is channeled down the rip lines, creating rills.
- Recontouring and decompaction are best done within a few days of seeding (e.g. hydroseed, imprinting, etc.), which carried out between September to

the middle of December (depending on temperature and precipitation conditions of that season). The soils on the ANF restoration sites tended to develop a thin, hard crust within a few days after recontouring/decompaction (especially after even a minimal amount of precipitation). This crust can make it difficult for seeds to embed in the soil and germinate.
- Erosion control measures must be installed before the first precipitation event to prevent soil erosion.
- Great care should be taken when determining which erosion control measures to utilize and the order in which they are applied. For example a site should not be hydromulched or sprayed with a soil tackifier (e.g. guar) for erosion control and then seeded within the first year. This is because the hydromulch will prevent much of the seed from coming in contact with the soil and greatly decrease germination. Another erosion control material such as wood fibers (applied at less than 40% cover) or wattles should be chosen, as they are compatible with seeding. Similarly, wattles should be placed on a site prior to hydroseeding as the hydroseed material is easily disturbed by foot traffic.
- Work with heavy equipment operators on site to ensure details of recontouring, decompaction, topsoil replacement and trackwalking methods are followed. Do not expect that lengthy restoration plans will be read by those on the ground implementing the restoration.

Chaparral Seeding Techniques

Of the three seeding methods described below, broadcast seeding and hydroseeding can be done without much soil preparation (typically hand tools only). Imprint seeding, however, normally does require soil preparation with the use of heavy equipment (although small areas can be imprinted with a hand imprint tool).

Broadcast Seeding—

***Methods*—**To capitalize on higher soil moisture levels, spreading seed by hand was mostly completed between late October and January 1. Before broadcasting, seeds were mixed with a dispersal agent such as rice hulls or bran to achieve consistent coverage. After evenly broadcasting seeds at the specified rate, they were lightly raked into the soil surface to ensure good soil-seed contact. Seed should not be over-raked as the seed should be buried no further than the length of the longest side of the seed.

Lessons Learned:

- Broadcast seeding has been tried on approximately 40 sites within the past five years. On average, the percent germination and resulting canopy cover of native plants on these sites has been very low (less than 5% cover in year one post seeding), making broadcast seeding the least effective seeding technique across the range of restoration sites. Thus, broadcast seeding is not recommended for reestablishing native cover in a short (under ten years) time frame.
- Two hypotheses as to why broadcast seeding has been ineffective on the ANF is 1) because of high rates of predation by namely birds and rodents and 2) wind dispersal. To mitigate these losses, fall broadcast seeding in 2014/2015 will be covered with wood fibers.

- Broadcast seeding may be the only choice in remote or inaccessible areas where hydroseeding or imprinting is not feasible, or in sites that have minimal impacts such as minor vegetation cutting or trampling.
- If broadcast seeding is the only option, the seeding rates should be 50–75% greater than rates for imprinting or hydroseeding to compensate for losses to predation.

Imprinting—
***Methods*—**Imprint seeding was used in disturbance sites over 50 ft^2 and accessible by equipment (bulldozer and roller). To date roughly 20 sites have been imprinted during the months of November to January. Imprinting was conducted using an imprinting roller with V-shaped angles. The diameter of the roller (20 to 24 inches), the angle length (8 inches) and width (8 to 11 inches), the angle type (straight or bowlegged), and imprinting pressure (15 to 30 pounds per square inch) varied slightly according to site conditions. Imprint size, shape, spacing, and pattern were designed to maximize water concentration and infiltration. Imprinting rollers were towed behind a D4 or D5 CAT sized bulldozer. A drop seeder was attached to the top of the imprinting roller so that seeds were released on the ground before the imprinter rolled over them. Wheat bran, or a similar binder, was mixed with the seed to improve the uniformity of application. Imprinting was always performed before container plants were placed on the site.

Lessons Learned:
- Germination rates and subsequent native plant cover varied from 5-15% across the 20 sites in the first year. Imprinting sites always performed better than areas treated with broadcast seeding. Most likely because imprinting created better soil/seed contact and provided many micro-catchment troughs to collect water.
- To prevent soil compaction, bulldozers or other tracked vehicles pulling the imprinter should not be over the weight of a D4 CAT.
- Compacted or hard soils should be loosened by ripping prior to imprinting to ensure that the imprint troughs are deep enough to retain water. The deeper the imprints the better, since imprints can be washed out by several rain events or one large one.
- Take care to not mix too much bran in the seed mix, as this can clog the drop seeder.
- Do not perform imprinting when soils are more than damp. This can cause soil compaction and/or not allow for deep enough imprint troughs as soil sticks to the imprint roller.
- Perform imprinting well before the first fall rains (September-October) to expose seeds to as much precipitation and other dormancy breaking conditions (freeze, thaw, etc.) as possible.

Hydroseeding—
***Methods*—**Approximately 25 of the restoration sites were seeded using a modified two-stage hydroseed application from late October to late January. The modified method allows for better seed-to-soil contact that is required for good seed germination and reduces the number of trips with the hydroseeding truck (Newton and Claasen 2003, Steinfeld et al. 2007). The application procedure was as follows:

First Application:

- Bran mixed with specified seed mix applied by hand using broadcast seeding method.

Second Application:

- 1,500 lbs./acre of long-strand wood fiber applied by hydroseed truck
- 100 lbs./acre Ecology Control "M" or similar environmentally benign binder applied by the hydroseed truck

All hydromulch mixing was performed in a clean tank. The tank was rinsed a minimum of three times, to remove any undesirable seeds, such as ornamental seeds from a previous hydroseeding job. The hydromulcher was equipped with a built-in continuous agitation and recirculation system of sufficient operating capacity to produce homogeneous slurry and a discharge system that applied slurry to the restoration areas at a continuous and uniform rate. Hydromulch was applied in a sweeping motion and in an arched stream until a uniform coat was achieved, taking care not to create areas of heavy hydromulch accumulation (more than ½ inch thick).

Lessons Learned:

- Hydroseeding helps protect seed from being blown away or eaten, and helps trap moisture in the soil that promotes seed germination. The hydromulch also provides some erosion control by dissipating water over a slope. However, on more than several sites heavily applied hydromulch appeared to accelerate erosion where water did not uniformly penetrate the mulch. This caused water to sheet into openings or at the bottom of sites, which then caused rilling and sediment discharge down the slope.
- Compacted or hard/smooth soils should be loosened and roughened using ripping shanks or similar equipment, prior to hydroseeding to allow for good soil/seed contact and water infiltration.
- The hydromulch slurry should float down from the arched stream, as opposed to being shot directly at the ground to prevent disturbance of the soil and broadcast seeds.
- Preventive measures must be taken to avoid damage to existing native vegetation, container plants, and cuttings (e.g., spraying and covering plants with mulch, breaking stems or branches with hoses). In the event that hydromulch does end up coating existing plants it must be removed the same day to prevent loss of plant growth and productivity.
- Hydroseeding helps protect seed from being blown away by the wind or eaten, and helps trap limited moisture in the soil to promote germination of the seed. The hydromulch can also help provide some erosion control in the areas it is applied by dissipating water over a slope. However, on more than several sites heavily applied hydromulch appeared to accelerate erosion in cases where water was not able to uniformly penetrate the mulch. This caused water to sheet into openings or at the bottom of sites, which then caused rilling and sediment discharge down the slope.
- Based on observations alone it is difficult to say whether imprinting was more effective than hydroseeding because after year one canopy cover was roughly the same (5-15% cover). Both techniques have different advantages. Imprint seeding provides greater soil-seed contact and small troughs for water infiltration that protect and encourage germination. Hydroseeding provides a protective coat on top of the seed that discourages predation and

losses to wind. The best seeding technique, which has yet to be tested on the ANF, is probably a combination of both imprinting and hydroseeding. In other words, after recontouring and decompaction, imprint the site to create pockets in the soil to catch seed and water and then hydromulch over the imprinted area. This would, of course, require the site to be accessible to heavy equipment and reachable by hydromulch hose.

- As with imprinting all hydroseeding effort should be performed before the first fall rains (September-October) so that seeds will be exposed to as much precipitation and other dormancy breaking conditions as possible.

Container Planting Techniques—

After any necessary site preparation is completed container planting may occur. If container plantings are utilized in tandem with a seeding technique it is best to perform the seeding method first, so that the containers are not disturbed.

Methods—

As described in the "Guidelines for Choosing a Plant Palette and Collecting Native Plant Materials" paper of this General Technical R, seeds for container plants should be harvested from areas within 5-10 miles of the restoration site, usually within the same HUC 6 level watershed. Eight of the restoration areas have been planted with container plants, amounting to around five acres total. All planting stock was grown in a southern California nursery for at least nine months and usually closer to one year. Almost all of the container plants used on the restoration sites were in one gallon pots with about a six inch rooting depth. Before transport to the restoration site, container plants were certified free of weeds, disease and invasive insect species.

The container plant quantities and species were specified in the individual restoration site plant palettes. Prior to digging, the locations of the container plants were marked (typically by pin flags) to space the plants in natural-looking patterns. Considerations for the microclimate requirements for each species were also taken into account. On average, plants were installed on 5-7-foot centers for shrubs, with closer spacing for bunch grasses and wider spacing for larger shrubs and trees. All container plants were planted in accordance with the following specifications:

- Chaparral container plants should be planted in the fall/early winter (October1-December 15).
- All planting holes should be augured (although no wheel-mounted augers shall be permitted) or hand dug, have vertical sides with roughened surfaces, and be one and one-half (1.5) times the diameter and twice the depth of the plant's container
- Any roots wrapped around the sides of the containers should be pulled loose from the root balls. Plants should be planted with the roots untangled and laid out in the planting holes to promote good root growth and prevent the plants from becoming root bound
- Roots should be adequately protected at all times from the sun and/or drying winds
- After excavation and before planting, the planting holes should be filled approximately 25% with thoroughly broken up native topsoil and filled with water. Holes should be allowed to drain thoroughly between fillings to reduce settling

- Plants should be set in the thoroughly drained planting holes so that the crowns of the root balls are 0.5 inch above finished grade when backfilled with soil. The soil around the planting should be tamped down sufficiently to eliminate any air pockets in the soil. The root crowns of the plants should not be depressed.
- A watering basin 24 inches in diameter should be constructed around each plant. The basin should be constructed by creating a berm above grade. The soil inside and outside of the basin should be at the same level. The basin should not be a depression in the soil.
- Deep pipes may also be placed next to (within 2-4 inches of the root ball depending on species) the container plant at the time of outplanting. (The deep pipe irrigation method is described below in the "Maintenance" section).
- Each plant should be individually watered at the time of planting with sufficient water to reach the lower roots. Special care must be taken to prevent the soil from washing away from the roots and the root crown from being buried with soil.
- If the plant species is one that is frequently browsed by herbivores (*e.g.* California sagebrush (*Artemisia californica*), California buckwheat (*Eriogonum fasciculatum*), rabbitbrush (*Ericameria nauseousus*), herbivory cages should be placed over the container plant. Herbivory cages should be constructed of chicken wire or similar material that will prevent herbivores from chewing through to the plant. Cages will be a minimum of 2 feet above ground. Cages should be removed before they hinder plant growth.

Lessons Learned—

- Optimal planting periods of chaparral species on the ANF vary by elevation and weather patterns. Most outplantings occurred in early December. This was adequate timing in poor rain years, but not ideal in good rain years because the plants missed several fall rain events, meaning more supplemental watering was needed. Two restoration sites over 5000 ft in elevation that had containers put out in late December and early January were not successful, most likely due to frost kill and a lack of adequate acclimatization prior to planting. In normal or above average rain years it appears to be best to outplant in mid to late October before the first fall rains for all elevations. In poor rain years it may be best to outplant from late October to early November at elevations over 5000 ft., but to wait till mid-November for lower elevations.
- Container plants in the standard nursery "one gallon" pots (~6 inches in depth) are not the ideal planting size for arid restoration. This is because the root system of plants in these short pots is typically not adequate to support the robust above ground foliage grown in the nursery in extremely dry outplanting conditions. In the future the ANF plans to use more deep rooted plants, which should be grown in tree pots (~14 inches tall). The goal will be to have fully developed roots filling the container with a good root-to-shoot ratio (approximately 3:1). Better developed roots with less above ground foliage to support should be more successful and require less watering (which is often exorbitantly expensive).

- Container plants should have had at least a month of exposure to outdoor conditions similar to the climate of the restoration site to ensure that hardening off (or acclimatization) has occurred.
- Repellents to prevent herbivory are not recommended as they have not proven to be an effective deterrent.
- Literature shows that tree/shrub shelters greatly enhance the growth of plants by providing increased moisture and watering ease and protection from wind, and sunlight and heat radiation (Bainbridge 2007; Steinfeld et al. 2007). The ANF has not yet used tree shelters for chaparral species, but they are proposed for upcoming restoration sites, especially those sites receiving chaparral oak and tree plant species.
- Increase container plant numbers by at least 20% to ensure an adequate number of plants will survive.
- Covering the water basin around container plants with mulch or wood fibers increases moisture retention.
- Restoration sites with container plants outperformed all seeded sites, with container plant areas reaching an average of approximately 30-40% cover by the first year (gaining ~10% each year after in normal rain years). This most likely is due to 1) the lack of reliance on inconsistent field germination; 2) the inherently larger canopy cover containers supply and 3) the reduced weed proliferation from the increased competition for light and nutrients that containers provide.
- It is best to use long lived perennials, species that require specialized seed treatments for germination (e.g. heat, smoke, scarification), and/or late seral species in the container planting palette, as these species are often hard to establish by hydroseeding or imprinting and can typically only be reintroduced to a site in a timely manner (less than 10 years) by plantings. Annuals, short lived perennials, species with high seeding germination rates and early seral species are all more efficiently and cost effectively restored in the field by seeding techniques.

Restoration Maintenance Techniques

After the initial seeding or container plant installation maintenance activities are needed to ensure successful restoration. The maintenance techniques of watering and weed control are described below. Watering is only done for container plantings on the ANF, but it can be used for seeded areas if funding is adequate. Weed control is essential on all restoration sites, regardless of the site preparation done or the seeding/planting technique employed.

Watering—

Methods—Supplemental irrigation is necessary to ensure successful establishment and growth of container plants. Seeded areas were never watered in any of the restoration sites. The frequency and amount of watering depends on the type of irrigation utilized, the plant species, site accessibility and funding. The two types of watering techniques utilized are described below:

- Deep pipe watering: Deep pipes are vertical plastic pipes (typically 2 inches in diameter and 14 inches in height) with holes drilled in them at different depths that are inserted next to container plants to allow irrigation water into the deep root zone. This improves water delivery to the plants' roots and minimizes water evaporation and weed growth. A cap or screen is put over the top of the pipe to discourage wildlife and debris from entering. Deep pipes can be filled by a water truck hose, watering can, or a drip emitter with pulsed irrigation from a remote storage tank or water truck hookup. On the ANF only one site has had deep pipes installed and they were watered by a water truck hose. The schedule for the deep pipe watering should be as follows: In the first year after planting, container plants watered (typically 1-2 liters) twice per month every month of the year if no rain event occurs. The rain event should measure at least 1 inch to obviate the need for supplemental water. In the second year after planting, container plants should be watered twice per month, if no rain event occurs, between May and October. In the third year after planting, container plants should be watered once every two months starting in June and ending in October. In the fourth year after planting, container plants should not need to be watered, unless it is an extremely dry year.

- Surface watering: Watering is done using a hose or PVC pipe connected to a water truck or water storage tank. Water is applied using a shower head-type nozzle capable of providing low –pressure application to prevent erosion or damage to the plantings and planting basins. On the ANF seven restoration sites were surface watered by hoses hooked up to water trucks. The schedule for surface watering utilized the same schedule described for deep pipe watering only plants should receive 1-2 gallons per plant during each watering event. More water is needed in surface watering in order to ensure percolation down to the root systems.

Lessons Learned:

- On the one restoration site fitted with deep pipes the watering schedule outlined above was followed for the first five months after installation, but in the middle of summer watering stopped and the plants died in less than a month. However, prior to the cessation of watering, the plants seemed to be faring well. In addition, literature shows that deep pipe watering has been three times more effective at increasing plant survival growth than surface watering (Bainbridge 2007, Steinfeld et al. 2007). In the future the ANF plans to test more deep pipe watering to see it is indeed more effective and cost efficient than surface watering.
- At least one deep pipe should be installed for each container plant; otherwise plants do not receive enough water.
- Do not water when there is a chance of freezing. This is especially harmful to newly planted plants that are not fully acclimatized.
- Take care to not overwater. Most chaparral species are accustomed to low water availability and can die from prolonged overwatering. Of the "workhorse" container plants listed in Table 1 this seems to most affect yucca.
- For long, linear restoration sites consider utilizing the mainline irrigation technique, which involves setting up a PVC main waterline down the length of the restoration site with valves every 50-100 feet to allow for hose or drip access to all container plants. On the ANF this was recently installed over

1000 feet of a restoration site and has more than adequate water pressure when hooked to a water truck.

- The watering schedule described above is a best case scenario. However, even though it has provided the best success rates (close to 70-80% survival) on four of the restoration sites, container plantings on three other sites have received roughly half as much watering and have a 50-60% survival rate. Given that watering installation and maintenance is the biggest cost in container plant restoration, the ANF is currently trying reduced watering schedules on several new sites to see if less water application is economically efficient.

Table 1—Workhorse container plant species for chaparral ecosystems restoration on the Angeles National Forest, California.

Species	Common name
Adenostoma fasciculatum	chamise
Arctostaphylos glandulosa, A. glauca	Eastwood manzanita, bigberry manzanita
Artemisia californica, A. tridentata	California sagebrush, big sagebrush
Atriplex canescens	four-wing saltbrush
Cercocarpus betuloides	birchleaf mountain mahogany
Encelia actonii, E. californica, E. farinosa	encelia
Ericameria nauseousus	rabbitbrush
Eriodictyon crassifolium	yerba santa
Eriogonum fasciculatum	California buckwheat
Hesperoyucca whipplei	chaparral yucca
Malacothamnus fasiculatus, M. fremontii	chaparral mallow
Peritoma arborea	bladderpod
Salvia apiana, S. mellifera, S. leucophylla	white, black and purple sage
Sambucus nigra ssp. *caerulea*	elderberry

Weeding—

***Methods*—**Weeding of non-native plant species was a key maintenance component for all restoration sites. All non-native plant species were targeted for removal, but especially invasive species (see Table 2 for a listing of the most common weed species removed). Weeding was accomplished by one of three methods: hand-pulling, mechanical or herbicide removal. The three methods are described in detail below

- Hand-pulling: Removal was accomplished by taking out the entire plant (roots, stems, flowers, seeds) by hand. Hand removal often included the use of trowels or other digging equipment to remove roots. Hand-pulling of sites was typically done four to five times in a growing season (late February through mid-June, depending on elevation), since some plants resprouted and different species germinated at different times.
- Mechanical: Removal was done by weed whipping or cutting the tops/seed heads off non-natives before they produced viable seed. This was typically done at least three-four times in a growing season as the root system of non-natives is not touched.

- Herbicide: Removal was completed by spraying non-natives with a Forest Service approved herbicide using a foliar, cut stump, or basal bark method (depending on the species, plant stature and time of year). Approved herbicides were glyphosate, triclopyr, chlorsulfuron, imazapyr and clopyralid. Herbicides were only used when weather conditions were conducive to effective uptake of the herbicide by the targeted species (e.g., sunny, dry, and when plants are actively growing) and when wind conditions were such that herbicide drift was nonexistent (at or below the threshold for continuous wind speeds as specified on the product label). Applications of herbicide did not occur 24 hours prior to or following precipitation events. Where riparian areas were present, only water-safe herbicides and surfactants were utilized. All herbicide used was applied in accordance with State and federal regulations and mitigation measures specific to the restoration site. Restoration sites typically were treated two to three times during the growing season to catch newly germinating weeds.

Table 2—Most common non-native plant species removed from restoration sites on the Angeles National Forest, California

Species	Common name
Avena barbata, A. fatua	wild oats
Bromus diandrus	ripgut brome
Bromus hordeaceus	smooth brome
Bromus madritensis ssp *rubens*	red brome
Bromus tectorum	cheat grass
Centaurea solstitialis	yellow star-thistle
Centaurea melitensis	tocalote
Erodium botrys, E. cicutarium	filaree
Festuca myuros	rattail fescue
Hirschfeldia incana	shortpod mustard
Lactuca serriola	prickly lettuce
Medicago polymorpha	bur clover
Melilotus officinalis, M. alba	sweetclover
Nicotiana glauca	tree tobacco
Polypogon monospeliensis	rabbitfoot grass
Salsola tragus	Russian thistle
Schismus barbatus	Mediterranean grass
Sisymbrium altissimum, S. irio	mustard
Spartium junceum	Spanish broom
Stipa miliacea	smilo grass
Festuca myuros	rattail fescue

Lessons Learned:

- It is crucial to conduct weeding several times during the growing season, well before seed is produced, regardless of the weed removal method chosen. Hand pulling and mechanical weeding should be carried out at least 4-5 times and herbicide applications at least 3 times per season to ensure that all non-natives are removed. Timing depends on the precipitation and temperatures of the growing season for a particular year, but in general, the best times appear to be early February, following up in mid-March, then early April and a finishing round in late April/early May. Weeding must be maintained for several years. In areas with weed coverages less than 15-20%, weeding may only be required for three years (if conducted at the correct times so that seed is not produced). However, in areas that had greater than 20% weed cover or that did not receive proper weeding at the correct timing, removal may be necessary for six to ten years.

- Care must be taken with herbicides to not impact nearby natives. This can be difficult, especially when restoration has just begun and newly germinating plants are small and close together. Early herbicide application in February is key as many natives have not yet germinated, but many non-natives are actively growing.
- Hand-pulling and mechanical weeding techniques were the only means of removal on the restoration sites until 2012. This four year period of hand/mechanical weeding provided ample time to show that the efficacy of these techniques was typically low. This was due to 1) the lengthy time and manpower it took to remove the non-natives by hand (typically herbicide removal took a half to a quarter of the time and manpower); 2) the inability to adequately remove roots, which led to resprouting; and 3) the propensity for weed whipping to also harm nearby natives and actually spread weed seed if not done at the right time of the season. Because herbicide is so much quicker to apply, it typically kills the entire plant and there are ways to mitigate harm to nearby natives, it is the most efficient and effective means of weed removal.
- Of all the restoration techniques weeding is by far the most important. Several sites on the ANF have recovered native vegetation in a few years without any seeding or container plants if the non-native cover is kept lower than 5%. When weeding has not been done at the correct time of year or at all native recovery has been stunted (less than 15-20% coverage) or non-existent, no matter how much seeding or container planting has been completed.
- It is important to remove all non-native invasive species. This applies to common weeds like cheatgrass (*Bromus tectorum*), shortpod mustard (*Hirschfeldia incana*) or filaree (*Erodium* spp.), which have the most potential to overrun/outcompete natives, especially in seeded areas.
- In areas that have over 25% cover of non-natives consider only weeding for the first year of restoration, especially if the plan is to seed the area. This is called a “grow/kill cycle”, which allows for the first large flush of weeds to germinate followed by removal over one growing season, thus depleting the amount of weed seed in the seed bank. This will give the native seed applied the following year to have less competition and a better chance of survival. It is also easier to apply herbicide the first year without the concern of herbicide drift to natives. If the non-native cover is particularly high it may be beneficial to go through two “grow/kill cycles.”

Conclusions

The most important factors influencing the success of chaparral restoration are the adequacy of planning, weather patterns, hydrology and soil conditions, pre-project invasive plant infestations, availability of materials and personnel, and, as always, cost. The highest performing restoration sites on the ANF have been those that either had a low presence of non-natives beforehand or had aggressive weed control in the first two/three years of restoration, combined with the use of adequate amounts of genetically appropriate, early seral native plants introduced to the site as container plants at the correct time in the early fall and watered consistently. Hydroseeding and imprinting can be effective restoration techniques as well in sites with less than 25% preexisting weed cover, if adequate amounts of PLS seed (around 30 pounds) are applied before the first fall rain (September-October) and the soil is decompacted and furrowed (has pockets/depressions where water can collect).

In the future, the ANF will continue to test new techniques or combinations of techniques, timing, and maintenance schedules in order to determine the most efficient and effective chaparral restoration methods. However, it must always be recognized that every potential restoration site is different and possesses its own unique microclimate and site attributes, meaning that a restoration technique may perform well in some locations, but not as well in others that may seem very similar. Research to statistically quantify and compare the efficacy of these different restoration techniques under a wide variety of site conditions is needed, so that land managers can rely on more than anecdotal or qualitative data to make the most informed restoration decisions.

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Guidelines for Choosing a Restoration Plant Palette and Collecting Native Plant Materials[1]

Katie VinZant[2]

Abstract

One of the first tasks in restoration planning is determining the appropriate native plant species and associated quantities. Species with the greatest prevalence at the restoration site, collectability, heat, sunlight and disturbance tolerance, germination and reproduction rate, large growth habit, soil retention qualities, nitrogen fixation, wildlife habitat value, and weed competitiveness were the main traits considered and desired. After this analysis is completed, a listing of on average twenty different species is generated, along with seeding or planting rates per acre for each species. Once this palette of species is developed, the locations from which the native plant material will be harvested and guidelines for those collections should be determined.

Keywords: chaparral restoration, Angeles National Forest, seed collection, native plant palette

Introduction

Wildland vegetation restoration, especially arid land restoration, requires multi-year planning to ensure that the correct native species are obtained and effectively utilized. One of the first, and probably most important, steps in beginning restoration planning is to determine the native plant species that will make up the planting palette. The palette is a suite of species that are the most appropriate plants to utilize for revegetation purposes. This intention of this paper is to provide guidance on methods for 1) selecting a successful plant palette; 2) determining the amount of seed or container plants needed for a site; and 3) sustainably harvesting wildland seed and cuttings. A “lessons learned” section also highlights challenges that can arise and techniques for moving forward.

Planting Palettes

The native species present in the vicinity of the proposed restoration areas are used as a reference for proposed seed, cutting and container plant palettes. When determining which plant species to use at a site the species with the greatest prevalence, collectability (*e.g*. not extremely sticky, spiny, or rash inducing), heat, sunlight and disturbance tolerance, germination and reproduction rate, large growth habit, soil retention qualities, nitrogen fixation, wildlife habitat value, and weed

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competitiveness are the main traits considered and desired. To maintain plant species diversity on the site an average of 20 different species are typically listed in the plant palette, with the rationale that probably only 10-15 of these could be collected/grown given logistical and environmental hurdles. After testing a wide variety of species on restoration sites with different aspects, elevation, disturbance levels and vegetation a group of “workhorse” species which have the most consistent success emerged. A sampling of these species is provided in Table A for seeding restoration techniques and Table B for container planting. An example of a plant palette for mixed chaparral is provided in Table C.

In order to determine the amount of seed, cuttings and/or container plants that were needed for a restoration site, the most important factors considered are the projected amount of collectable material, existing non-native plant cover in the vicinity, soil compaction, annual precipitation and potential for unauthorized vehicle traffic. The amount of seed needed is determined in Pure Live Seed (PLS) pounds. PLS represents the percent of the gross seed weight composed of viable seeds. PLS is determined by performing both purity and germination tests. On the ANF the amount of PLS pounds needed on a restoration site varied from around 15 pounds on riparian sites to 35 pounds on desert transition chaparral sites. On average mixed chaparral sites received 30 PLS pounds. Determining quantities of container plants is discussed in the paper “Restoration in type converted and heavily disturbed chaparral: Lessons learned” of this General Technical Report.

Lessons Learned-

- It is important to create seeding palettes using PLS pounds, not bulk seed pounds, as much of the bulk poundage is not viable and will not provide any canopy cover for the restoration site. PLS rates vary widely by species and by season, so it is best to have the PLS percentage tested for each year’s collection. The gross average PLS percentage for these restoration sites was around 30%, meaning the bulk rate of seed applied to the restoration site needed to be roughly three times greater than the PLS pounds required for the site (1 pound PLS=3.3 pounds bulk).
- If seeds which require some sort of pre-treatment (which will not naturally occur on the site) for germination must be used, ensure they have indeed received that treatment. Standard methods for overcoming seed coat dormancy include scarification, stratification, soaking in hot/cold water, chilling, heating, exposure to different kinds and durations of light, chemical treatment, and combinations of these. Examples of common species that require pre-treatment are chamise, manzanita, ceanothus and yerba santa.
- In the plant palette be sure to include species that act as nitrogen fixers, such as species in the pea (Fabaceae) family.
- Do not include fire following species in the planting palette unless money and time are not an issue. These plants are typically very difficult to propagate, especially under wildland restoration conditions.
- Have large plant palette species lists (in the 20’s), knowing some will drop out due to environmental conditions or logistical hurdles.
- Over-collect by around 20% of the needed PLS rate. This will provide contingency for species that end up having low germination rates and build seed stores for remedial plantings.

Table—A. Workhorse plant species for restoration by seeding of chaparral ecosystems on the Angeles National Forest, California.

Species	Common name.
Acmispon glaber	deerweed
Ambrosia acanthacarpa	annual bur-sage
Artemisia californica, A. tridentata	California sagebrush, big sagebrush
Astragalus filipes	threadstalk milkvetch
Atriplex canescens	four-wing saltbrush
Bromus carinatus	California brome grass
Elymus elymoides	squirreltail grass
Eulobus californicus	California suncups
Encelia actonii, E. californica, E. farinosa	encelia
Ericameria nauseousus	rabbitbrush
Eriogonum fasciculatum	California buckwheat
Helianthus gracilentus	slender sunflower
Peritoma arborea	bladderpod
Poa secunda	perennial bluegrass
Salvia apiana, S. mellifera, S. leucophylla	white, black and purple sage
Salvia columbariae	chia sage
Stipa coronatum, S. lepida, S. pulchra, S. speciosa	needle and thread grass
Vulpia microstachys	Pacific fescue

Table B—Workhorse container plant species for chaparral ecosystems restoration on the Angeles National Forest, California.

Species	Common name
Adenostoma fasciculatum	chamise
Arctostaphylos glandulosa, A. glauca	Eastwood manzanita, bigberry manzanita
Artemisia californica, A. tridentata	California sagebrush, big sagebrush
Atriplex canescens	four-wing saltbrush
Cercocarpus betuloides	birchleaf mountain mahogany
Encelia actonii, E. californica, E. farinosa	encelia
Ericameria nauseousus	rabbitbrush
Eriodictyon crassifolium	yerba santa
Eriogonum fasciculatum	California buckwheat
Hesperoyucca whipplei	chaparral yucca
Malacothamnus fasiculatus, M. fremontii	chaparral mallow
Peritoma arborea	bladderpod
Salvia apiana, S. mellifera, S. leucophylla	white, black and purple sage
Sambucus nigra ssp. *caerulea*	elderberry

Table C—Example of plant palette for a mixed chaparral restoration site (includes seed and container plant rates) on the Angeles National Forest, California.

Species	Common name	Seed or container	Pure live seed pounds/ac. containers/0.5 ac.
Achnatherum coronatum	giant needlegrass	seed	1
Acmispon glaber	deerweed	seed	2
Adenostoma fasciculatum	chamise	seed	1.2
Adenostoma fasciculatum	chamise	container	100
Ambrosia acanthicarpa	annual bur-sage	seed	0.5
Arctostaphylos glandulosa ssp. *glaucomollis*	Eastwood's manzanita	container	50
Ceanothus leucodermis	chaparral whitethorn	container	50
Cercocarpus betuloides	mountain mahogany	container	40
Corethrogyne filaginifolia	common sand aster	seed	0.2
Cryptantha intermedia/C. muricata	popcorn flower	seed	0.2
Eriastrum densifolium	shrubby eriastrum	seed	0.3
Eriodictyon crassifolium	yerba santa	container	100
Erigeron foliosus	leafy daisy	seed	0.3
Eriogonum elongatum	long-stemmed buckwheat	seed	0.5
Eriogonum fasciculatum	California buckwheat	seed	4.5
Eriogonum fasciculatum	California buckwheat	container	100
Eriophyllum confertiflorum	long-stemmed golden yarrow	seed	1
Hazardia squarrosa	saw-toothed goldenbush	seed	0.3
Hesperoyucca whipplei	chaparral yucca	seed	3
Hesperoyucca whipplei	chaparral yucca	container	50
Heterotheca grandiflora/H. sessiliflora	telegraph weed/golden aster	seed	1.3
Lotus scoparius	deerweed	seed	2
Malacothrix saxitalis	slender-leaved malacothrix	seed	0.2
Malocothamnus fasiculatus/M. fremontii	lax-flowered bushflower	seed	0.5
Melica imperfecta	coast range melica	seed	1
Phacelia cicutaria	caterpillar phacelia	seed	0.5
Poa secunda	perennial bluegrass	seed	2.5
Prunus illicifolia	holly-leaved cherry	container	40
Salvia apiana	white sage	seed	2
Salvia columbariae	chia	seed	1
Salvia mellifera	black sage	seed	3
Salvia mellifera	black sage	container	75
Sambucus nigra ssp. *caerulea*	elderberry	container	20
Vulpia microstachys	Pacific fescue	seed	3
Total			30 lbs, 625 containers

Plant Material Collection Guidelines

All cuttings and seeds should be collected from sources in the local area. The use of local plant materials, which are adapted to local conditions, increases the likelihood that the cuttings and seedlings are successful and at the same time maintains the genetic integrity of the local ecosystem (Erickson et al. 2003, Rogers & Montalvo 2004). Local in this case means within a five mile radius of the same HUC6 watershed of the restoration site. Care should also be taken to ensure that plant materials are from within 500-1,000 vertical feet of the elevation of the site. On the ANF in a few circumstances when sources were limited, this radius was extended to ten miles for more widespread herbaceous species that were likely to be genetically homogeneous. Following is a list of the native plant material collection protocols that were developed:

- For each species in a permitted seed mix, seed should be obtained from several (more than two) populations within the permitted collection area. A different population is defined here as two populations that have a very low chance of exchanging genetic material with each other. In other words they are separated by distance or a geographical barrier. To make this determination it is often necessary to have some knowledge of the species' main pollinators and their movement capabilities.
- Seeds must be collected from at least 35 different, well dispersed individuals in a population. A record should be kept of the estimated number of individuals sampled.
- No more than 25% of the total seeds of an individual can be taken, with the exception of areas that have all vegetation removed by project activities.
- Significant damage (such as cut limbs or crushing) to the parent plant should not be incurred during seed collection.
- Seed heads may be cut off just below the inflorescence; however, cutting shears must be sterilized between project areas to prevent disease spread.
- Seeds must be mature at time of collection. This is highly dependent on the plant species and weather, but in general runs from April through late December.
- Dry seeds should be collected into bags or plastic bins and then transferred into breathable (paper, cloth or poly) bags. Fleshy seeds should be collected directly into plastic bags.
- Seeds should be stored temporarily in a cool, dry (best to use desiccants) state in breathable (not plastic) containers and labeled with the species name, date collected, location collected (latitude/longitude, UTMS), name of collector, average elevation, and Project name.
- To the extent feasible, harvest cuttings from plants outside the migratory bird nesting season (February 1–August 15). If activities must be performed during the breeding season, the crew must take precautions to avoid nests in the work area. If active nests are observed, the supporting vegetation must be excluded from any type of

collection or cutting. A minimum 10 foot exclusionary buffer is recommended.

- Take cuttings only from healthy, vigorous plants that are in a dormant state (typically late November-late February depending on elevation).
- Collect the cuttings within 24 hours of anticipated planting or propagation. All cuttings should be placed (the entire cutting) in water until planting time. Cuttings that are allowed to dry out shall not be used.
- Do not collect from more than 25% of the plants in a given area and do not remove more than 25% of any individual plant.
- Cuttings shall be done with sharp, sterilized tools (to minimize spread of disease) and approximately 6–24 inches in length and will range from 0.5 to 1 inch in diameter.
- Cut the top of each cutting square at the nodes, above a leaf bud and the base below a leaf bud at an angle of approximately 45 degrees in order to be able to tell the correct orientation of the cuttings for planting (angled cut placed down).
- Trim leaves and branches from the cuttings flush with the stem to encourage rooting success.
- Cuttings should be labeled with the species name, date collected, location collected (latitude/longitude, UTMS), name of collector, average elevation, and Project name.
- Whole plants or bulbs can only be removed from a site for later transplanting if they are likely to be destroyed by site disturbances.
- Avoid weedy areas in order to prevent seed contamination and the spread of non-natives. Areas showing signs of vegetation type conversion to non-natives (i.e. greater than 100 ft^2 areas estimated to have over 25-30% non-native canopy cover) shall not have seeds/cuttings collected.
- Off road vehicle driving is not allowed for collection. Vehicles must stay on designated, unvegetated roadbeds.
- Before entering the ANF vehicles must be washed if they have driven off of paved roads. Vehicle washing should include the wheels, undercarriages, bumpers and grill portions of vehicles. Tools/equipment that have been utilized off ANF lands or in weed infested areas shall also be cleaned prior to moving to a different site.

Lessons Learned-

- Start seed/cutting collection at least two years (preferably 3-4 if the restoration site is over a few acres) in advance of when the plant material is needed at the restoration site. In light of the current California drought conditions it is best not to rely on only one season of collection, as plants are stressed and seed production is typically lower than normal.
- A determination of what is “local” plant material is often subjective, but preserving genetic integrity and using plants best adapted to the restoration site must be considered a priority.
- Wildland plant material collections are often very difficult given terrain and questionable availability (especially for native grasses in

chaparral habitats). Bulking seeds by growing it out as a crop in a farm-like fashion is often a preferred alternative so some environmental variables such as lack of water can be controlled. Protocols for bulking should be followed: 1) only seed collected from the wild should be utilized to grow the first crop; 2) species with the potential to hybridize or the same species from different geographic locations should be separated far enough to prevent cross-pollination; and 3) seeds produced from bulking should not be utilized to grow the next crop so that genetic traits specific to the restoration area can be preserved.

- Adequate long term seed storage is of great importance if seeds are going to be kept more than three years. In this case seeds should be stored in cool temperatures (around 40F) and low humidity (6-7 % moisture), but seed specific requirements should be followed. It is also important to protect seeds from rodents, insects and mold/fungi for both long and short term storage.
- If seed/cutting collecting can be done prior to a site being disturbed consider collecting more than 25% of the seed or the plant (for cuttings) in the actual disturbance footprint, as these plants will likely be destroyed by construction activities.
- Transplanting bulbs and whole plants has been tried four times over the past five years on the ANF. Transplanting of whole plants was only tried if the plant was a perennial, under 2-3 years of age. The reasoning for only transplanting young plants is that older plants typically have longer, more developed root systems, which are more easily damaged. Overall both transplanting of bulbs and whole plants was only marginally successful (around 10% survival when outplanted). This was most likely due to human error in transplanting (*e.g*. cutting roots, over/under watering, incorrect soil medium), but also may not be feasible given environmental requirements that are not understood or difficult to mimic.

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