Reevaluating Desert Upland Habitat Restoration Sites

Developing effective restoration practices for habitats across Clark County

Shown in 2024, view of a 2013-2014 restoration treatment including soil recontouring and planting on a decommissioned road, Boulder City Conservation Easement, Clark County. At this location, restoration succeeded in visually concealing the disturbance, repairing soil structure, and re-establishing perennial cover plants reported as used by desert tortoises. This is plot BCCE33, one of 363 monitoring plots measured in this project.

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Through a

INTERLOCAL AGREEMENT

REEVALUATING DESERT UPLAND HABITAT RESTORATION SITES

between

CLARK COUNTY, NEVADA

DEPARTMENT OF AIR QUALITY AND ENVIRONMENTAL MANAGEMENT

DESERT CONSERVATION PROGRAM

and

BOARD of REGENTS, NSHE, obo UNIVERSITY OF NEVADA, LAS VEGAS

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EXECUTIVE SUMMARY

For the benefit of MSHCP-covered species and their Mojave Desert habitats, the Clark County Desert Conservation Program (DCP) funded a broad-scale field assessment of the current status of 15 habitat restoration projects in upland deserts throughout Clark County. In this project, we sampled full plant communities (annuals-perennials, native and non-native) and soil conditions (e.g., stability rating, biocrust cover) on 363 plots. Plots were in disturbed/restored, disturbed/unrestored, and undisturbed reference habitat settings. Restoration projects sampled ranged in age from 2-25 years, spanned a broad gradient of environmental conditions (e.g., soil parent material types), and included a diversity of treatment types. Restoration treatment types ranged from seeding and outplanting to ripping to de-compact soils and applying topsoil as well as many others. We have produced thousands of comparative treatment field images, a database containing raw vegetation and soil data for the 363 plots, and metrics of vegetative conditions such as the cover of food plants known to be favored by desert tortoises. This report provides an overall synthesis of findings across the 15 restoration projects as well as summaries of the findings of each of the 15 projects individually. As an Appendix, we provide a case study example (Bonnie Springs Fire seeding project) of a value-added leveraging of long-term data (back to 2009 in this case) available for some of the project sites.

A major conclusion from this DCP project is that we consider 9 of the 15 (60%) of the restoration projects to be successful or at least partially successful based on various effectiveness metrics and comparison with unrestored or reference habitat. Given the degree of difficulty of desert habitat restoration and the fact that dry conditions preceded and occurred during the 2024 field measurements, we consider this success rate to be encouraging while also highlighting areas where further work is necessary to improve future restoration effectiveness. There were examples of every major restoration treatment type (e.g., seeding, outplanting, ripping) succeeding in at least one project. This is significant as it indicates a full suite of treatment types has potential for success in Clark County, while also indicating that careful consideration of synergistic treatment combinations may be important. In particular, a major recommendation based on this project is to apply multiple treatment types in combination to hedge against failure of any one component and to leverage synergies between treatments. We also note that there were some broad-scale project failures, including after priority disturbances such as wildfires and roads on unique soil types where restoration was needed but failed. In these cases, we recommend that experimental trials occur in small areas to identify effective treatments so that they are available when needed or before attempting to upscale to the landscape.

Project Objectives

Objectives for this project included:

- Determine soil (e.g., stability) and vegetative habitat condition (e.g., proportion of native plant species, shrub density as cover plants for wildlife such as desert tortoises) of restoration sites up to 10-20+ years after restoration treatments were implemented as long-term indicators of restoration success;

- Compare the effectiveness of different restoration approaches including practices such as soil amendments, seeding, and outplanting, and consider the potential influences of site conditions or climate (e.g., drought) that could influence restoration effectiveness; and

- Modeling variation within and among restoration project sites across Clark County to identify potential factors associated with what works where and when to aid enhancing the ecological and financial cost effectiveness of future habitat restoration approaches.

By working with Clark County staff and resource managers in southern Nevada to ensure direct applicability of the project results to land management, we were able to identify and sample 15 different restoration projects implemented throughout Clark County in the last 25 years to meet the project objectives listed above. We sampled 363 plots, comprehensively analyzed a variety of soil and vegetation habitat quality metrics, and conducted assessments of restoration treatment effectiveness both within and among project sites.

Significant Results

- Of the 15 restoration projects, we classified 3 as fully successful, 6 as partially successful, and 6 as likely failed at achieving objectives.

- We consider the 60% project success or partial success rate to be encouraging given the degree of difficulty in restoring severely disturbed desert habitats and the severe dry period in the last four years leading up to the field assessments.

- The success or failure of individual restoration treatment types appeared highly contingent on their variations. For example, seeding could work or not work depending on whether the seed was pelleted.

- Every major restoration treatment provided at least some benefits in at least one project. For example, some treatments increased the abundance of cacti, a priority species group. Other treatments or combinations of treatments increased cover of foundational shrubs or forage plants known to be favored by desert tortoises.

- Some restoration projects succeeded despite the failure of several or nearly all of their components but were driven to success by the one or few components that did succeed. For example, some outplantings had most of their species die, but the few species that were successful enabled meeting restoration objectives.

Conservation and Management Applications

- Multiple restoration treatment types applied in combination appear most successful, as different treatment types can have positive synergies with each other while also buffering a project from total failure if only one treatment is applied that does not work. A good example of synergy was a long-term project along Northshore Road where the combination of topsoil salvage + planting was most effective and was even resistant/resilient to the recent severe drought.

- Desert habitat restoration has often been viewed as a one-time treatment implementation. This is different than in other restoration settings, such as keeping temperate prairies or savannas open that are subject to woody plant encroachment, where repeated treatments are assumed. It is possible that the restoration of desert habitats is similar, where multiple restoration inputs in phases or a replenishment (e.g., providing new plants to replace dead ones or injecting further abiotic resources) is required over time. Although conducting restoration in multiple phases may not always be logistically feasible, applying and re-applying different restoration resources over time could be important for effective and sustainable restoration. For example, a low-density outplanting on the Goodsprings Fire seems to have provided benefits, but following up with an additional outplanting (or abiotic treatments) may tip the habitat toward more rapid and complete recovery.

- Active revegetation (seeding, outplanting, transplanting) was effective in some projects but failed in others. We recommend that native plant species continue to be screened for their amenability to restoration as well as to their needs for being successful in restoration. For example, some species appear amenable to seed pelleting while others do not. Some species may require different herbivory protection or irrigation to become established. A major conclusion from this project is that these details can matter greatly in influencing whether restoration is successful.

- Although overall the restoration project results were encouraging, some of the failures including expensive projects that were implemented across broad areas provide the valuable insight that small, experimental trials may be critical to identify effective treatments before attempting to upscale. Although it may seem like there is an urgent need for action after some disturbances, it should be remembered that applying what ends up being an ineffective treatment across thousands of acres just means upscaling failure. A major conclusion from this DCP project is that the results highlight an important role of small adaptive management trials or experiments to identify the factors limiting to restoration success, identifying which treatments or species are likely to be successful, and working out details on treatment variations needed for success. This disciplined approach may be the most cost effective and capable of restoring the most strategic acres in the long term.

- The dry climate overall since the 2000s in southern Nevada and specifically the four years preceding and encompassing our 2024 field measurements underscore that development of restoration treatments needs to be dynamic as conditions change. What worked in wetter periods may not work currently or if projected increasing drought frequency and severity is realized. We recommend increased use and exploration of abiotic treatments (e.g., vertical mulch) that do not require the survival of emplaced seeds or live plants which can be highly contingent on rainfall. Some of the most successful among the 15 projects heavily relied on abiotic treatments. More fully developing abiotic treatments that can stimulate natural plant recovery or that can be deployed in combination with active revegetation may become increasingly important if ongoing drought reduces the number of years in which active revegetation can succeed.

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INTRODUCTION

Description of the Project

Ecological restoration techniques are needed in southern Nevada to improve habitat quality for conservation-priority species and ecosystem services for humans (e.g., soil retention to limit dust and maintain air quality). A variety of disturbances, such as past land clearing, soil compaction, loss of native plants, and non-native plant invasions have reduced habitat quality. This degradation can compromise habitat quality for desert tortoises and other conservation-priority species. Degraded habitats can impact humans by generating dust that can produce particulate air pollution, exacerbate effects of ongoing regional drought of recent decades, compromise aesthetic appeal and recreational opportunities (e.g., native plant displays, wildlife viewing), and in some cases increase chances for environmental hazards such as flooding and wildfires. While ecological restoration is not a panacea always capable of reestablishing pristine conditions, prior research and synthesis funded by Clark County has demonstrated numerous examples of at least partially successful restoration in southern Nevada deserts (Abella et al. 2023). Continuing to improve restoration approaches could reduce financial costs while enhancing ecological effectiveness when restoration is applied to small, targeted areas and to broader landscapes.

With extensive areas of disturbance already existing in southern Nevada and projections for continued human population growth, new temporary and permanent disturbances, and potentially continued non-native species invasions and drought conditions, habitat restoration is anticipated to remain an important or increasingly important species conservation tool in the future. Our project, funded by the Clark County Desert Conservation Program, is a unique effort to efficiently learn from prior restoration projects what has worked and where (or has not worked) throughout Clark County. The overarching objective is to determine current habitat conditions at sites where a variety of restoration treatments were applied at different times in the past throughout Clark County. This information is then used to develop management applications for improving the chance for success (for both habitat enhancement effectiveness and financial/logistical implementation efficiency) of future habitat restoration efforts in Clark County. These applications can be useful to identify further management activities that could enhance habitat quality at existing sites where restoration was less effective and to plan new restoration projects.

Background

Similar to elsewhere in the Mojave Desert and drylands in the southwestern U.S. and beyond, furthering habitat restoration techniques in southern Nevada is important to ameliorate degradation and improve ecological functions and services provided to wildlife and humans (e.g., reducing dust generation to improve air quality). Degradation can compromise habitat quality for desert tortoises and other conservation-priority species, such as by reducing important cover and forage plants while increasing non-native grasses producing hazardous fuels. Habitats denuded of native vegetation reduce native floral resources available to pollinators. Degraded habitats are dust sources that lower ecosystem carbon storage capacity, compromise aesthetic appeal and recreational opportunities, and could increase environmental hazards. Ecological restoration techniques mitigate some of these impacts through reintroducing native plants, abiotic structures, or landscape features that contribute to ecosystem structure and functioning. However, most research or monitoring studies of restoration are conducted at only one or a few sites and are short term, with monitoring often extending only a few years after restoration. By including a wide diversity of restoration sites in Clark County and evaluation encompassing up to several decades, this project is intended to help with: choosing optimal restoration techniques appropriate to the diversity of site, climate, and restoration goal situations that managers can encounter; identifying the most cost-effective treatments appropriate to small or large areas; and understanding the long-term effectiveness of restoration approaches including in contemporary drought conditions in the southwestern U.S.

Included in the Desert Conservation Program – Multiple Species Habitat Conservation Plan (MSHCP) are restoration and enhancement measures for public lands that support MSHCP-covered species and their habitats. Restoring plant productivity on disturbed areas includes identification and evaluation of existing habitat with significant potential for enhancement and restoration, and includes a range of rehabilitation, reclamation, and revegetation techniques to improve habitat for reestablishing native flora and fauna. Because of the emphasis on restoring damaged habitats for the benefit of MSHCP-covered species and their habitats, identifying best restoration practices in the county is important for furthering this goal. This project also supports activities further enhancing restoration practices on public lands including in Lake Mead National Recreation Area, Tule Springs Fossil Beds National Monument, Boulder City Conservation Easement, and Bureau of Land Management-administered lands (e.g., Red Rock Canyon National Conservation Area) in Clark County.

One of the underlying premises of ecological restoration is that active intervention by humans is required for ecosystems to recover after disturbance or to recover on a timeframe meeting habitat management objectives. While in some cases after disturbance a “do nothing” approach may be the most ecologically and cost-effective (e.g., if most top-killed native perennials do re-sprout rapidly), much research in and around Clark County in the Mojave and Sonoran deserts has shown that plant and soil recovery after disturbance is slow or essentially non-existent without restoration. In a synthesis of 47 published studies in the Mojave and Sonoran deserts, the time estimated for the full reestablishment of perennial plant cover after disturbances (clearing such as for pipeline right of ways, wildfires, and road disturbances) averaged 76 years (Abella 2010). Recovery rate of perennial species richness (number of species per unit area) was highly variable among studies, ranging from full recovery within 3 years to projections of 152 years to full recovery. Recovery of species composition (species present and their relative abundance) to that typical of nearby undisturbed areas was estimated to require an average of at least 215 years, assuming that recovery would continue along a linear trajectory. Recovery of native annual plant species composition may similarly require decades to centuries and be contingent on recovery of native perennials providing nutrient-enriched fertile islands conducive to recruitment of many annual species (Berry et al. 2016).

A recent, separate synthesis of recovery after 32 wildfires dating back to 1980 across a large Mojave Desert landscape centered in Clark County reported that perennial plant cover required decades to recover while species composition was projected to require centuries at many sites

(Abella et al. 2021). Species composition of mature creosote bush (*Larrea tridentata*) communities required an estimated 82 years to full recovery. Blackbrush (*Coleogyne ramosissima*) shrublands required even longer: 550 years to full recovery. In both communities, the time to recovery actually lengthened between measurements made in 2007 and 2016, implying that instead of recovery, species composition of the burned sites was diverging from that of nearby undisturbed communities. This implied that species composition on burned sites, at least when unassisted by active restoration, may not be capable of recovering to that resembling unburned sites in ambient environmental conditions.

These long recovery times are significant to MSHCP conservation efforts (e.g., for desert tortoises and other conservation-priority species) for several reasons. First, these types of disturbances (e.g., wildfire) already cover vast areas of habitat and they are continuing to increase (Brooks et al. 2018). Second, even if some plant recovery occurs (e.g., cover increasing over time), the species may not necessarily be those that were originally present, so their benefit to priority species may be comparatively lower. A good example is that early colonizing perennial plants tended to be smaller-statured species, too small to cover an adult tortoise, rather than the larger shrubs tortoises favor for cover and burrow-construction locations (Abella et al. 2021). Severe disturbances can become even less hospitable to tortoises if some of the early colonizing perennial forbs that serve as food plants decline after their initial colonization (Drake et al. 2015). Third, the long recovery times imply a cumulatively increasing “recovery debt,” whereby increasing portions of the landscape spatially and temporally are not in a mature habitat condition. These observations underscore the potential for restoration to accelerate ecological recovery on existing disturbances while also contributing to efforts to curtail new anthropogenic disturbances.

Another main premise of ecological restoration is that it seeks to identify and ameliorate the factor(s) limiting natural recovery. For example, a disturbance that removes vegetation and compacts soil could result in at least two major limiting factors: lack of plant propagules and soil properties negatively affecting water infiltration and plant growth. In such a case, restoration may seek to de-compact soils, to enable them to support plant growth, while actively reintroducing plant propagules through seeding or planting intended to accelerate plant colonization. Our project includes examples of a comprehensive array of approaches intended to ameliorate limiting factors such as by including soil amendments and revegetation treatments.

In addition to the limiting factors imposed by the disturbance itself upon which restoration is conducted, drylands pose numerous other challenges to restoration for promoting recovery. This is underscored by the fact that undisturbed, natural desert habitats are generally sparsely vegetated where opportunities for natural plant establishment are infrequent. The themes common across the desert regions that often challenge restoration practitioners and require overcoming include limited availability of native plant materials for restoration; low and erratic precipitation and hot, desiccating summers; often intensive levels of herbivory at restoration project sites given limited natural forage (e.g., herbivores targeting planted species enriched in nutrients from propagation processes in greenhouses); infertile or shallow soils after disturbance; and competition or fire hazards stemming from non-native plants (Bainbridge 2007). Compared to moister regions, these issues are heightened challenges in drylands where plant regeneration even under natural conditions is relatively infrequent, precisely a reason why plant cover in undisturbed drylands is low.

Restoration of disturbed sites across desert regions generally includes one or a combination of the following: reintroducing plant propagules (e.g., seeds or seedlings), supplemental treatments or care of the propagules to encourage plant establishment (e.g., protecting seedlings from herbivory), restoring abiotic habitat structure or surrogate structures partly replacing functions of live plants (e.g., using dead plant materials to conceal disturbances), stabilizing eroding soil and reestablishing biocrust where appropriate, and ameliorating inhospitable site conditions otherwise deterring natural recovery (e.g., augmenting soil nutrient availability or enhancing soil seed banks). These broad categories of techniques are briefly summarized below as context for the specific descriptions of habitat restoration techniques covered later in this report.

Seeding Seeding first entails collecting or obtaining seed, which is challenging as seed of few native perennials is commercially available and often mandated only local genetics can be used in restoration projects. Additionally, seed can be unobtainable during dry years with little or no viable seed production, potentially delaying restoration projects reliant on seeding. Owing to its contingency on suitable climate conditions and vulnerability to failure from seed predation by fauna, seeding is not considered fully reliable as a revegetation method across the desert regions. Seeding can be effective, however, if climate conditions are suitable, seed quality is high, and in some cases if supplemental treatments are applied such as coating seeds with protective substances (e.g., Grantz et al. 1998, Abella et al. 2015b). Due to its uncertainty and propensity to fail, seeding can be paired with other treatments as a “bet-hedging” restoration strategy to try and ensure that at least one restoration activity is successful.

Outplanting Outplanting involves growing seedlings from seed and caring for the seedlings in nurseries or greenhouses before seedlings are planted at field sites. Many desert perennials reside in nurseries for up to a year to develop root systems before being outplanted. Seedlings are grown in standard 1-gallon pots or in other container types (e.g., biodegradable) to facilitate transport to the field. For many species at many sites, supplemental care of outplants at field sites is required to enhance survival. For example, irrigation (where possible) or protective structures (e.g., small cages to deter herbivory) can enhance survival. The added cost and complexity of these treatments and their potential benefits should be balanced with the alternative of simply planting more plants and accepting lower survival. With proper planting stock and plant care, outplanting has been shown to reliably restore perennial plants in dozens of studies in North American deserts (e.g., Bean et al. 2004, Abella et al. 2012). It should be recognized, however, that outplanting success may be hindered during extreme drought regardless of plant care. Additionally, owing to the challenges of outplanting large areas, outplanting is typically a strategy suited for small disturbances or for restoring vegetated patches to stimulate recovery within large disturbances. Similar to the nursery propagation of seedlings for outplanting, salvaging plants and either transplanting them directly to a restoration site or caring for them in a nursery for a period of time can also be a useful restoration technique.

Soil amendments or stabilization These types of treatments seek to replace lost topsoil or organic matter, rake out tracks such as for visual blending of soil disturbance, restore biocrust, and otherwise repair soils and enable conditions for habitat recovery. Stabilizing and improving conditions of soil at disturbed sites can stimulate recovery of desert habitats and vegetation.

Abiotic structural restoration Given the cost and difficulty with restoring live desert plants directly, surrogate, abiotic structures can sometimes be used to partly provide the functions of live plants (e.g., trap seeds and slow soil erosion). Abiotic structures can also potentially serve as "nurse objects" that can provide shade, ameliorated temperatures, and protection from herbivores, thus facilitating recruitment of seedlings. An example is vertical mulch, consisting of placing dead plant material upright in soil to simulate a live plant (Rader et al. 2022). These abiotic structures can partially provide some of the functions of live plants while saving the cost and effort of collecting seed, propagating plants, and requiring survival of reintroduced propagules to become mature plants as needed for successful seeding or outplanting.

Topographic restoration Many disturbances alter the configuration of the land, such as by flattening and compacting the land surface. In addition to negatively impacting visual appeal of the habitat for humans, these disturbances negatively affect habitats such as by removing natural topographic heterogeneity including depressions as water catchments or areas where seeds can lodge and recruit. Restoration approaches to reestablish natural topographic patterns can include roughening the land surface, recontouring sites to mimic pre-disturbance or nearby undisturbed landforms as closely as possible, or replacing rocks to reestablish surface land structure.

In addition to incorporating examples of all of these types of treatments, our project incorporates three further features of restoration uncertainties that can complicate the application and predictability of restoration outcomes: non-native plants, variation in inherent site conditions (e.g., soil parent materials), and lack of clarity as to whether short-term results portend longer-term trends. By seeking to improve site conditions for plant growth, restoration can inadvertently improve conditions for undesired, non-native plant species. This can be the case, for example, when restoring fertile islands (locations of nutrient-enriched, shaded and protected soil below some perennial plant species) that are often important to reestablish native desert ecosystem structure. Rather than native plants, non-native plants such as red brome (*Bromus rubens*) can disproportionately benefit from these fertile islands. Restoration may need to deal with non-native plants for projects to be successful, and we have included several metrics of non-native plant abundance in our evaluation of projects subsequently described in this report.

A variety of sites varying greatly in topography, elevation, and soil parent material (e.g., soil derived from limestone, gypsum, or a variety of other materials) can incur disturbances and require restoration. One of the major uncertainties in ecological restoration generally and in deserts specifically is how this inherent variation in site conditions can affect the implementation and performance of restoration. The same restoration treatment on the same type of disturbance may perform differently among different soil types or across elevation gradients. These differences can stem from a variety of factors, such as differential responses of different soil types to amendments or different plant compositions responding to treatments differently. In our project, we measured and obtained a variety of ancillary variables to provide site context to help potentially explain variation in restoration results. For example, we augmented site-specific soil descriptions we made in the field with use of the Clark County Soil Survey to obtain soil taxonomic classifications of restoration sites (Lato et al. 2006).

Performance of most restoration projects is monitored for at most a few years after restoration activities (if monitoring is conducted at all), often making it unclear whether the early outcomes portend longer-term performance. Projects that could appear unsuccessful initially could become successful over the long term, or vice versa. For example, of 16 published studies including outplanting in the Mojave and western Sonoran Desert, the maximum time that outplant survival was monitored was five years (Abella et al. 2023). In many cases, it was only 1-2 years. If an outplant is already confirmed dead, then longer-term monitoring may not be necessary. However, in many cases it may be important to monitor survival through at least several years of differing weather conditions (i.e. including droughts as well as performance in wetter years), exposure to natural stressors at the sites such as soil movement or herbivory, and to plant reproductive maturity to determine if the outplants (even past their death) can have resulted in sustaining populations of the species.

The long-term effectiveness of restoration is a crucial knowledge gap that can affect numerous facets of treatment implementation and costs. For example, it may be more effective in the long-term to implement cheaper treatments that do not necessarily provide strong immediate results but that can result in sustainable enhancements of habitat conditions for decades. On the other hand, it is possible that one-time treatment applications, whether intensive or less-intensive, are insufficient for long-term success as instead multiple treatment applications in phases may be required. Our project incorporates restoration sites varying widely in age from 2 to 25 years.

Goals and Objectives of the Project

The specific goals of this project included:

i. Determine plant community condition, based on metrics such as native and non-native species cover, at sites throughout Clark County that had received a variety of restoration treatments and that span a diversity of soil, topographic, and vegetative contexts.

ii. Assess variation in habitat quality and functional metrics, such as in soil stability (which can be linked to soil erodibility and air quality) and cover and forage plants utilized by federally listed desert tortoises, across the diversity of sites receiving different restoration treatments across Clark County.

iii. Use this project’s dataset, which included many factors (e.g., time since restoration, disturbance type) co-varying with restoration treatment type (e.g., seeding, topsoil salvage), in combination with previous published studies to examine the relative effectiveness of restoration treatment types in different situations to help inform future restoration efforts.

iv. Identify potential factors – both controllable and uncontrollable by resource managers – most likely associated with the degree of restoration project effectiveness warranting consideration to improve the ecological and cost-effectiveness of future restoration efforts.

To meet our goals, the objectives for this project included:

i. Field sampling of vegetation and soil conditions at 15 restoration projects throughout Clark County for which we worked with land managers to obtain records of the restoration activities or for which we had conducted previous monitoring. This resulted in two types of projects with respect to prior monitoring: 1) projects with longer-term, existing monitoring data which this current project greatly extended into the realm of long-term (multi-decade) effectiveness assessment; and 2) projects assessed for the first time. At the 15 restoration projects, we sampled 363 monitoring plots for this project.

ii. Compilation of ancillary information which can be quite important to restoration success and relating this information to project outcomes. For example, we compiled soil taxonomic classifications from the Clark County Soil Survey and specific descriptions where available of nuances of restoration treatments (e.g., genetic origins of seeds).

iii. Completion of data analyses at two complementary scales: separately within each of the projects to understand restoration effectiveness at each of the 15 project sites, and in aggregate across all 363 plots at the 15 project sites to identify broader patterns and variation in how restoration is influencing habitat conditions across Clark County.

iv. Assembly of this comprehensive report together with value-added preparation for case study scientific manuscripts (covering the individual projects for which prior monitoring data are available and a comprehensive manuscript) to distribute results more broadly to restoration practitioners in the Mojave Desert and other drylands working on similar challenges of improving restoration techniques for drylands. Clark County and the funding sponsors are and will be fully acknowledged using standard text provided by the DCP in these manuscripts for providing positive reinforcement of the value and effective use of DCP funding.

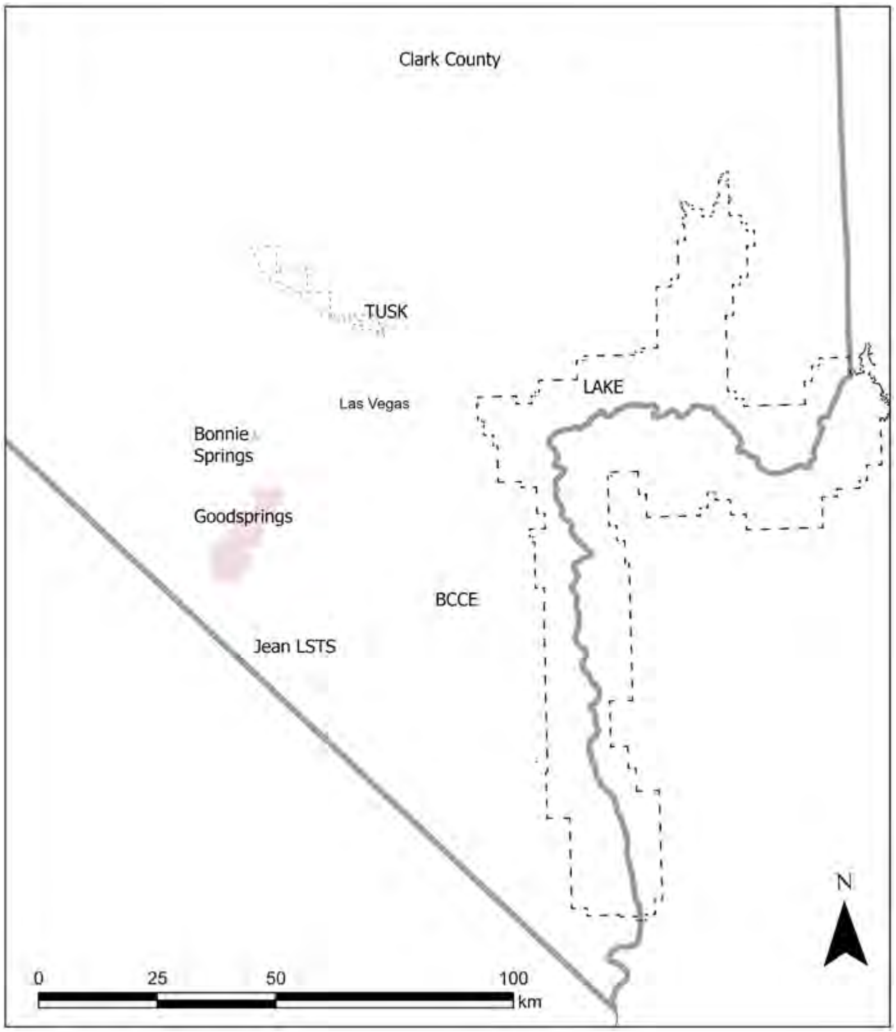
METHODS AND MATERIALS

In 2023 to identify potential restoration sites suitable for sampling, we queried land managers at all four federal agencies (National Park Service, Bureau of Land Management, U.S. Forest Service, and Fish and Wildlife Service) administering lands in Clark County and project managers with Clark County. In addition, our query was submitted to the interagency Southern Nevada Restoration Team email listserve that includes state-level and other resource managers. We also pursued leads with private contractors who had worked on restoration projects on public lands, such as for the Kern River Pipeline. In all of our queries, we sought to identify restoration projects suitable for sampling that were in Clark County and met the following criteria:

- Be an intentional restoration activity conducted on one or more discrete disturbances (e.g., wildfire, decommissioned road) in desert uplands in Clark County. The restoration activities could include, but were not limited to, outplanting, seeding, soil enhancements, topographic enhancements, vertical and horizontal mulch, topsoil salvage, and others.
- Have records on specifically when (the year) restoration was performed and fundamental details on the activity, such as the species planted. The location of the restoration activity must be known or identifiable via a field verification visit.
- Be accessible for sampling, on public lands and where monitoring was permitted.

We were also able to include prior restoration projects that we had implemented ourselves (e.g., Goodsprings Fire outplanting, Lake Mead shoreline outplanting) or for which we were involved and had records (e.g., Fish Hatchery site including soil amendments and outplanting).

We identified and sampled 15 restoration projects that met the criteria described above. Each restoration project was defined as occurring in a particular geographic region (which could include multiple, geographically separated sites) and occurred on the same disturbance type and was implemented in the same time period. For example, the Bonnie Springs Fire restoration project occurred on different sections of a 2007 wildfire and involved seeding implemented in 2008. The Northshore restoration project occurred at different sites disturbed by road maintenance activities and distributed along Northshore Road. Restoration activities including reapplying salvaged topsoil and planting salvaged plants were completed in 2010 at the distributed disturbed sites in that project. Sampling locations among these 15 projects spanned a broad gradient across Clark County and are shown below.



Location of 15 restoration projects in which we sampled 363 plots on public lands in Clark County to determine habitat conditions in a variety of restoration treatments implemented between 2 and 25 years before sampling in 2024.

Project locations included Lake Mead National Recreation Area (LAKE), Tule Springs Fossil Beds National Monument (TUSK), Bonnie Springs and Goodsprings Fires, the Large-Scale Translocation Site for desert tortoises located near Jean, Nevada (Jean LSTS), and the Boulder City Conservation easement (BCCE). LAKE and BCCE have several projects. Projects in LAKE include SNWA Endcap planting, the Las Vegas Bay Landfill planting and Callville Bay Landfill planting, the Shoreline plantings, the Lake Mead Lodge planting, the Fish Hatchery restoration, the Road108 road ripping, and the Northshore Rd realignment project and restoration. In the BCCE, projects included the 2013-2014 and 2020-2022 restorations and a 2020 experimental seeding.

The 15 restoration projects included several disturbance types, treatments, and time since treatment completion as of the 2024 sampling (summarized in the table below). For some projects, the treatments listed in the table were implemented separately in different areas, enabling us to compare treatment types within the projects. For other projects, the treatments were applied in combination. On the clearing disturbance in the Fish Hatchery project, for example, outplanting was combined with rocks being re-spread on the soil surface and the coloring agent Permeon applied to simulate the surface color of desert pavement and rock varnish found on nearby undisturbed areas. In cases such as at Fish Hatchery, we were not able to partition individual treatment effects but were able to compare the combined restoration treatment with disturbed, untreated and nearby reference (undisturbed habitat).

Summary of the disturbance types, ages, and restoration treatments applied in 15 restoration projects we sampled throughout Clark County. Details on each restoration treatment are in the project database and descriptions in this report.

Project	Disturbance	Age (yrs)	Planting	Seeding	Soil ripping/ recontouring	Vertical mulch	Topsoil	Rock mulch	Imprinting	Fencing	Permeon	Horizontal mulch
BCCE 2013/2014 Planting	Road	10	x		x		x	x				
BCCE 2020 Seeding	Road	4		x	x				x			
BCCE 2020/2021/2022 Planting	Road	2-4	x		x	x		x				
Bonnie Springs	Wildfire	16		x								
Callville Bay Landfill	Clearing	12	x									x
Fish Hatchery	Clearing	25	x					x			x	
Goodsprings	Wildfire	16	x									
Jean LSTS	Clearing	12		x						x		
Lake Mead Lodge	Clearing	2	x		x			x				x
Las Vegas Bay Landfill	Clearing	12	x									x
Northshore Road	Road	14	x				x					
Road 108	Road	22			x							
Shoreline Planting	Clearing	13	x									
SNWA Endcaps	Road	4	x		x			x				
Tule Eglington	Road	13	x	x	x	x	x	x				x
Total - 15 projects			**11**	**4**	**7**	**2**	**3**	**6**	**1**	**1**	**1**	**4**

Among these 15 restoration projects, five had existing, standardized full plant community plots (Bonnie Springs Fire, Fish Hatchery, Jean LSTS Tortoise Translocation, Northshore, and Road 108). Some additional projects, such as the outplanting on the Goodsprings Fire, had existing information, such as initial outplant survival, but not full plant community inventory data around the restoration treatments. Our focus in the present project was on overall habitat condition at the restoration sites, rather than attempting to quantitatively determine the effects of individual actions. For example, our goal did not include attempting to determine the percentage of seeds sown in seeding projects that resulted in plants. Instead, we sought to determine complete vascular plant community composition at seeded sites which could include both seeded and non-seeded species.

For the projects with existing plant community plots, we re-sampled those plots. For projects without existing community plots, we randomly established plots in the project areas. The number of plots in restoration areas ranged from just one in small restoration areas (e.g., the small < 0.05-ha old landfill sites in Lake Mead National Recreation Area) to at least 2-3 plots in larger or distributed restoration areas. Where possible, we also randomly located and sampled equal numbers of plots in disturbed areas not receiving restoration treatments and in nearby undisturbed (and untreated) reference habitat of similar soil and topography as the disturbed areas. For most projects, this enabled comparing habitat condition in restoration areas with disturbed controls and undisturbed reference habitat.

Restoration projects with existing plots had been sampled with 100-m^2 plots. To keep plot sizes consistent for comparison across projects, we also established and sampled new 100-m^2 plots in projects not containing existing plots. The plot dimensions were generally 10 m × 10 m, but for some linear disturbances like roads, dimensions were required to be 4 m × 25 m or 3 m × 33.3 m to fit within the disturbance but in all cases totaled 100 m^2. In theory, by encompassing a longer environmental gradient, rectangular plots may be expected to encounter more plant species than square plots of the same size. We do not consider this in the present project because in practice, many of the linear disturbances (roads) were the most uniform topographically and our intention is to compare plant communities within a standardized area (100 m^2) across all disturbances. In total, we sampled 363 plots in restoration, disturbed control, and reference habitats. We sampled plots in spring from March to early May 2024.

Standardized datasheets for each of the 363 plots are provided in the deliverables for this project. In summary, we visually recorded the aerial percent cover by species for all vascular plant species on each plot using cover classes: 0.01, 0.1, 0.25, 0.5, 0.75, 1, 1.5, and 2%, 1% increments to 10%, then 5% increments to the maximum 100% aerial cover that an individual species could have on a plot. For shrubs, we recorded cover separately for seedlings (< 15 cm tall, cotyledon may still be present, or few true leaves present and minimally branched) and adults (≥ 15 cm tall and branching present). We also counted the number of individuals of shrub and cactus species separately for seedlings and adults to calculate plant density per plot. Cover and counts of live and dead individuals were performed separately for each perennial species. When counting shrubs and cacti, we categorized average canopy dieback among individuals for each species following criteria based on Bowers and Turner (2001): 1 is no dieback (100% of the leaves are green or alive all branches have live leaves); and classes 2 (0.01-0.25), 3 (0.26-0.50), 4 (0.51-0.75), and 5 (0.76-0.99) categorize proportions of the canopy branches that are leafless or have brown leaves. If dieback is 1.0 (100% of the canopy branches are leafless or have brown leaves), then the individual was categorized as dead that year, at least aboveground. This does not exclude the possibility that certain species (e.g., the drought-deciduous *Ambrosia dumosa*) could leaf out or re-sprout in subsequent years. Dead perennials can provide important structure in sparsely vegetated desert habitats and thus we included them in vegetation inventories. Nomenclature and classification of species by growth form (e.g., annual forb) and nativity to the U.S. follow the PLANTS Database (NRCS 2025).

In addition to recording location (Universal Transverse Mercator, North American Datum 1983) and elevation, we used a clinometer to measure slope gradient (%) and a compass for slope aspect (degrees, linearized following Beers et al. [1966]). Using the same cover classes as for plants, we recorded the aerial cover on each plot of the following substrate types: biocrust (which could include lichens, mosses, or cyanobacteria), litter (unattached organic material including woody debris), gravel (diameters ≥ 2 < 76 mm), cobbles (≥ 76 < 250 mm), stones (≥ 250 < 600 mm), boulders (≥ 600 mm), or exposed bedrock (Soil Science Division Staff 2017). The sum of all of these substrates on a plot could not exceed 100%.

At three evenly spaced locations across the center of the plot, we recorded several field-measured properties of the 0-5 cm surface soil, which was exclusively or nearly exclusively was mineral soil as O-horizons were sparse at these desert sites. We categorized texture by feel, which a recent study found more closely matched laboratory analyses than many previous studies (Salley et al. 2018). We evaluated precision of texture by feel in our study by having 40 of the samples assessed by two different members of our research staff. The two observers determined texture by feel to within one textural class (out of 12 textural classes) of each other for 93% of the samples (identical for 33%), with texture differing by two classes for 7% of samples. As an indicator of presence of free carbonate (CO_3), we categorized the effervescence reaction of soil to 1N HCl on an ordinal scale from 1 to 5 ranging from non-effervescent to violently effervescent (Soil Science Division Staff 2017). Using Munsell color books, we recorded the dry soil hue, value, and chroma. We measured soil compaction in Kg/cm^2 using a pocket penetrometer (AMS, Inc., American Falls, ID). Following Herrick et al. (2001), we measured soil aggregate stability. The three measurements each for compaction and stability were averaged on a plot basis. Lastly, we categorized as present or absent whether a plot was a desert pavement surface (Wood et al. 2005). Supplementing these field measurements, we obtained the dominant soil taxonomic unit and parent material for each plot from the county soil survey (Lato et al. 2006).

We prepared a Microsoft Office Excel database, included as a deliverable with this report, containing all of the descriptive information for each of the 363 plots, the full plant community data (species cover and densities), and metrics of habitat quality that we derived. For example, as detailed in the database, we calculated habitat quality metrics such as relative abundance of plant species associated with disturbance or with mature communities based on prior Mojave Desert published research, the cover and species richness of plants reported as preferred forage by desert tortoises, the cover and density of cacti as this is a conservation-priority group, and many other metrics describing the plant communities across the restoration, disturbed/unrestored, and reference habitat plots. In total, we calculated 92 univariate vegetation metrics, in addition to multivariate community composition also contained in the database. Although some of the metrics might be more relevant than others, these 92 metrics offer a variety of ways for assessing the communities. We used the statistical software SAS 9.4 to compute means and standard errors of means for each metric for restoration treatments (and controls and references) for each project. These statistics are included in full in the attached database.

Overall Synthesis of Restoration Treatments across All 15 Projects

Project Summaries

The 363 plots sampled across the 15 restoration projects in this DCP project span a remarkable diversity of disturbance types, restoration techniques, time since restoration including the climate context in which restoration occurred, and edaphic factors such as elevation and soil parent materials. Accordingly, project outcomes, as of the 2024 assessment, are highly variable. Moreover, the perception of effectiveness can vary depending on which effectiveness metric (e.g., native perennial cover, annual desert tortoise food plants, cacti abundance, soil stability) is used and in which type of comparison (restored:disturbed control or restored:undisturbed reference).

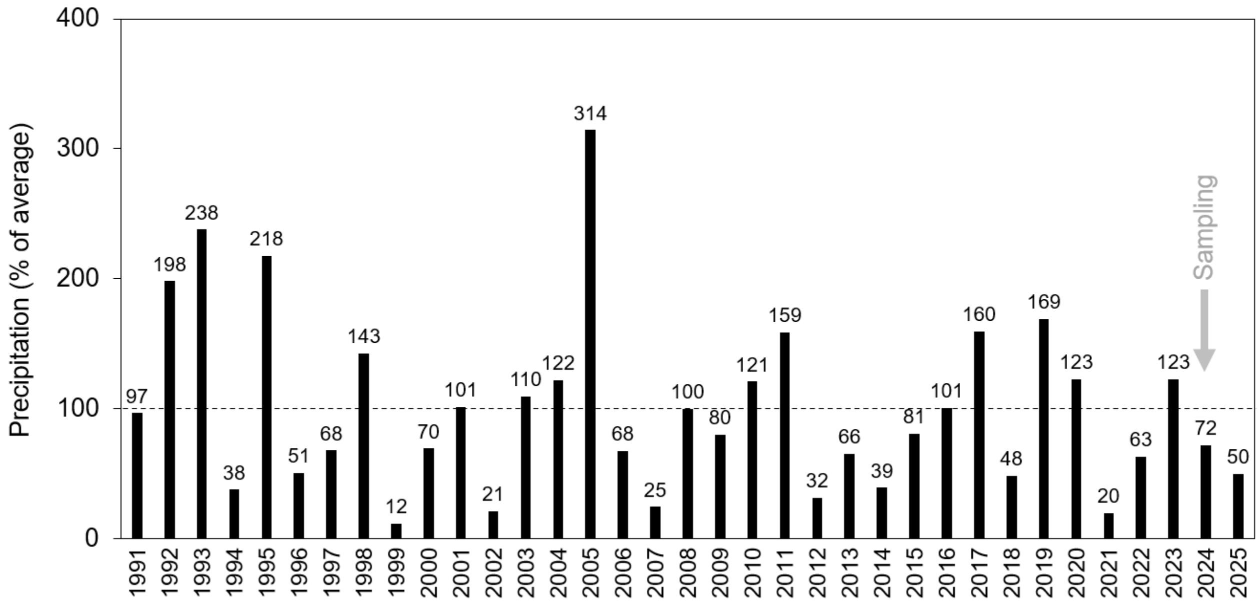
Before proceeding to summarize project effectiveness, the backdrop of dry climatic conditions leading up to the 2024 sampling for this DCP project warrants consideration. As shown in the graph below, three of the four years through the 2024 spring sampling had below-average rainfall. Notably, 2021-2022 represented severe drought with consecutive years of minimal precipitation. After above-average rainfall in 2023, the hydrological year (preceding November 2023 through April 2024) for the 2024 growing season was also dry. Based on ongoing work in the southern Nevada and southeastern California Mojave Desert, dieback or outright mortality of foundational shrubs such as creosote bush and white bursage was occurring even in many undisturbed reference habitat sites from 2021-2024. Furthermore, annual plant abundance was relatively low in undisturbed habitat in 2024 relative to wetter years. Given this context, we suggest several interpretations of the relationship of this dry climatic period with the 2024 restoration assessment:

It is not known whether plant communities on disturbances at restoration sites are affected differently by drought than are undisturbed reference communities. What seems likely, though, is that cover or density of perennial and annual plants was depressed on all plot types (restored, control, reference) in 2024.

Given above, 2024 may have been an excellent representative time period in which to assess restoration effectiveness. This dry period may offer a rigorous test as to the resistance and resilience of restored plant communities, which need to be sustainable through droughts to maintain restoration gains or further restoration inputs may be required. Furthermore, this dry period may be representative of the conditions in which restoration must operate in the future if climatic projections for increased aridity in the Mojave Desert in the coming decades are realized.

Although dry, 2024 still received sufficient rainfall for annual plants to appear, thus it was possible to include the full perennial-annual plant suite in evaluation of restoration effectiveness. This is not trivial, because if monitoring had occurred in say 2021, sites had few to no annuals, which would have made it difficult to assess restoration effects on annuals. The study design of including disturbed/restored, disturbed/unrestored, and undisturbed reference together at most project sites also enables direct comparison of plant communities at these locations that all experienced the same climatic conditions in the 2024 sampling. This represented a major advantage by sampling all 363 plots in the same year.

We also add that recent dry climatic conditions underscore the value of a long-term approach to assessing restoration effectiveness. This DCP project leverages a consistent, broad-scale sampling in 2024 across Clark County restoration projects in tandem with five projects (Bonnie Springs seeding, Fish Hatchery planting, Jean seeding, Northshore Road planting, Road 108 ripping) that have long-term data going back decades. We have already completed one value-added case study (Bonnie Springs Fire seeding) as an Appendix to this report that analyzes long-term fluctuations in plant community conditions at the restoration sites that in that case go back to 2009 through extreme climatic variation. As discussed later in this report, we have communicated with DCP staff plans for developing value-added case studies to this project for the other four projects that include long-term fluctuations. A strength of this DCP project is that it combines spatial variation across Clark County in restoration outcomes with long-term temporal variation at a subset of restoration sites.



Winter growing-season precipitation (November through April) expressed as a percent of the long-term average measured at Red Rock Canyon, near two of the restoration projects (Bonnie Springs Fire seeding and Goodsprings Fire planting) in this report. This climate graph is representative of the dry conditions that occurred since 2020 leading up to the 2024 sampling. The most severe annual drought in the last 20+ years occurred in 2021, followed by a second consecutive dry year in 2022. After higher-than-average rainfall in 2023, the sampling year of 2024 was also dry. One interpretation of these dry conditions leading up to 2024 assessment is that they provided rigorous assessment conditions for restoration effectiveness and may be representative of future climatic conditions if projections for increased aridity in future decades are realized.

The table below summarizes overall outcomes and notable findings based on different plant community effectiveness metrics for the 15 restoration projects. Note that in many cases, the restoration included combining treatments (e.g., ripping combined with planting), the effects of which were not intended to be separable. Although some of the projects were set up as experiments to differentiate effects of different treatments (e.g., pellet compared with bare seed in BCCE 2020 Seeding and Jean LSTS Seeding), many of the projects were operational projects not intending to distinguish aggregated treatments. Thus, we consider overall outcomes of these aggregated treatments in the table below.

Summary of outcomes of 15 restoration projects throughout Clark County assessed using 363 plots in 2024. In the table, VM is vertical mulch; + notes increase. All projects included planting or seeding, aside from the Road 108 ripping project. Projects are thus noted as planting or seeding, and many also included soil manipulations or mulch introduction as summarized previously in the project treatments table.

Project	Summary outcome
BCCE 2013/2014 Planting	Ripping and transplanting created VM; perennial cover minimal, slight + tortoise food plants
BCCE 2020 Seeding	No to minimal + in seeded species; slight + in seeded globemallow
BCCE 2020-2022 Planting	Ripping, rock mulch, transplanting + shrub density, structure early on
Bonnie Springs Fire Seeding	Despite using best practices, no evidence of seeding effectiveness 2009-2024
Callville Bay Landfill Planting	No outplants alive 12 years later; natural recovery of creosote bush on small disturbance
Fish Hatchery Planting	Soil amendment, planting recovered creosote community and tortoise food plants after 25 yrs
Goodsprings Fire Planting	Of planted species, 40% persist 16 yr later, subtle cover + with the low-density planting
Jean LSTS Seeding	Seeding perennials failed, but pellet seeding with fencing + desert tortoise food plants
Lake Mead Lodge Planting	Early (2 yr) establishment of foundational creosote-bursage evident; other planted species fail
Las Vegas Bay Landfill Planting	All 4 planted species persist 12 yr later; natural recovery of creosote bush on small site
Northshore Road Planting	Topsoil application and planting + perennial cover, cacti, and tortoise food plants
Road 108 Ripping	Ripping + perennial cover, including foundational shrubs, at 22 yrs post-tmt
Shoreline Planting	Early outplant survival occurred but by 14 yrs, little evidence outplanting increased natives
SNWA Endcaps Planting	Ripping, rock mulch, planting little discernable effect at 4 yrs; ocurred during drought
Tule Eglington Planting	Despite extensive plantings, minimal native cover on still bare roads after 12 yrs

In summarizing the overall restoration outcomes across the 15 projects, the timeframe of project success could be viewed two ways. First, early results the first few years after restoration may be critical in project success, such as in cases where wildfires occur and restoration to stabilize sites is considered an emergency and the goal. Second, whether a project succeeds in the long-term (decades) for meeting a goal of restoration of jump-starting recovery may be most important in a long-term restoration perspective. This could apply to either the gradual accumulation of ecological components or "stops and starts" whereby conditions at restoration sites may fluctuate through time including losses and gains in ecosystem components.

From these different perspectives and in no way critical of implementers (who in some cases were the authors of this report!) of the projects, as many factors (e.g., climate) beyond the control of implementers can affect project success, we categorize overall project effectiveness in terms of achieving at least some restoration benefits (e.g., increased tortoise food) as the following:

Fail: 6 projects: BCCE 2020 Seeding, Bonnie Springs Fire Seeding, Callville Bay Landfill Planting, Shoreline Planting, SNWA Endcaps Planting, Tule Eglington Planting

Highly mixed success: 6 projects: BCCE 2013/2014 Planting, BCCE 2020-2022 Planting, Goodsprings Fire Planting, Jean LSTS Seeding, Lake Mead Lodge Planting, Las Vegas Bay Landfill Planting

Successful: 3 projects: Fish Hatchery Planting, Northshore Road Planting and Topsoil, Road 108 Ripping

Note that, in our opinion, the classification above trends toward the overly rigorous spectrum of project evaluation. First, some of the six "failed" projects still may have achieved some goals, via natural recovery, despite the apparent failure of the restoration action itself. A good example is the Callville Bay Landfill Planting, which did not have evidence of persisting outplanted species but did have extensive natural recovery of the non-planted creosote bush which dominated surrounding reference habitat. While we declared the project a failure on the basis of no outplanted species persisting, we cannot rule out the possibility that the site preparation activity or early presence of outplants somehow facilitated recovery. Moreover, as described further in the individual project section, the restoration site regardless of anything else has creosote bushes naturally establishing which is consistent for recovery goals for the site. Also note that the SNWA Endcaps Planting, which we tentatively classified as "fail," is still early (completed 2020) and its entire post-restoration period has been in overall drought. Continued assessment of effectiveness could be important, and we emphasize that the above categorization is as of 2024.

Second, it is possible that some "mixed success" projects could move into the successful category over time, or that could be supplemented with further restoration activities to enable them to do so. For example, about half of the outplanted species persisted 16 years after outplanting on the Goodsprings Fire. We believe this is significant because the outplanting occurred at such a low initial density (< 1 outplant/100 m^2 overall). It is possible that a more intensive planting, or even supplementing current likely survivors with additional outplants, could further project success. Also notable in this "mixed success" category is that while some parts of a project may have failed, other parts, potentially even the most important parts, did succeed. A good example is the Jean LSTS Seeding which had dual goals of increasing tortoise perennial cover plants and annual forage plants. While seeding did not increase cover plants, the more paramount goal of enhancing tortoise forage availability did succeed via establishment of desert plantain (*Plantago ovata*).

In our opinion, the two projects in which the outcomes were most disappointing to us were the Bonnie Springs Fire Seeding (a project the report authors were involved with) and the Tule Eglington Planting. These projects, and potential reasons for the failures, are discussed further in the individual project sections. We note that for Bonnie Springs, even with following best-practices for seeding and with the project being expensive over a broad area, there was no indication of effectiveness during the entire 16-year available monitoring period from 2009 to 2024. For Tule, we emphasize that given the extreme difficulty of the soils at the sites where natural plant recruitment is so limited, a comprehensive program based on experimentation may be needed to avoid costly restoration failures. Conducting greenhouse and small field experiments may greatly aid the identification of effective techniques, before attempting to upscale. This type of work can focus on identifying limiting factors causing the failures and how to overcome them.

The three projects we consider highly successful did include a combination of multiple intensive treatments (Fish Hatchery involving soil manipulations as well as planting, which worked very well despite just one of the planted species persisting, and Northshore involving topsoil reapplication and planting) or a long time period (22 years) in the case of the Road 108 ripping. In all of these projects, at least some metrics have already recovered to reference levels. Given the degree of difficulty of desert restoration even under good conditions, nonetheless the recent severe drought, we consider the fact that 9 of 15 (60%) of projects being successfully or partially so to be quite encouraging. This illustrates that despite severe drought, restoration practices can still achieve persistent benefits. Although the six projects we thus far consider "fail" in terms of restoration outcomes, there is still much that can potentially be learned from them to improve future restoration. The 6 failed projects, combined with partial success in another 6, do illustrate though that much more work remains to be done to develop restoration techniques that are reliable when and where needed. This was underscored by the urgent need for restoration after the Bonnie Springs Fire, but the seeding was ineffective. This highlights that targeted development of effective restoration strategies is still needed for many disturbance types and scenarios that managers may encounter. Our DCP project has highlighted some of the treatments with potential or that need more work.

Restoration Treatment Type Effectiveness

As mentioned previously, this DCP project examined operational restoration projects. This has the disadvantage where many were not established as experiments to distinguish individual treatment types (instead often applying multiple treatments in aggregate). The advantage, however, is that treatments can be assessed in a realistic, operational context. The table below summarizes the overall influences of different restoration treatments. We note that numerous variations of each of the treatments can influence their effectiveness. For example, outplanting was effective in half of the projects. Although we cannot necessarily definitively pinpoint causes of the failures, some of the outplant failures occurred in projects where outplants were not protected from herbivory. In another example, pelletized seeding was highly effective (at least for some species) in one project but minimally effective in another. This underscores that continuing to screen which species benefit from pelleting, which types of materials of pellets, is likely to be important. This variation can interact with the numerous other contingencies associated with seeding, such as when to time seeding and to what extent site preparation is needed for success.

Summary of the influences of individual restoration treatments across 15 restoration projects in Clark County. In the table, VM is vertical mulch; + notes increase.

Treatment	Summary effectiveness
Transplanting	+ shrub cover in 2/3 projects; dead transplants created VM; + tortoise annual food plants in 2/3
Outplanting	Partial outplant species persistence in 4/8 projects; variations like cages with outplants important
Seeding	Materially effective in 1/3 projects (+ tortoise forage in the 1); failure even with best practices
Ripping	+ native perennials in 1/1 projects; combined with other tmts in others, likely mixed effects
Topsoil application	+ native plant cover, tortoise food in 1/1 projects, synergistic with planting
Rock mulch/varnish	Applied with other tmts so cannot distinguish, but associated with + native cover in 3/4 projects
Vertical mulch	Confounded with other tmts so cannot distinguish, but created structure for visual blending
Fencing	+ tortoise annual food plant in 1/1 projects, did not increase seeded perennials
Specialized sub-tmts	
Irrigate plantings	DriWater did not forestall some planting failures, but may have + survival in others
Protect plantings	Cages/shelters did not forestall some planting failures, but may have + survival in others
Mulching plantings	Did not forestall planting failure in 1 project, though outplants persist in 2 other projects
Pelleting seed	Aided seeding success for 1 species in 1 project; appeared ineffective in 1 project
Permeon varnish	Ripping, rock mulch, planting little discernable effect at 4 yrs; ocurred during drought
Recontouring	Included with other tmts so effects not isolatable, but did roughen surface where applied

Overall, there were instances of each of the major treatment types being effective, where seeding, outplanting, transplanting, and various abiotic treatments like mulching having benefits. This is significant because it highlights that a full toolbox is available to restoration practitioners while underscoring that more work is likely needed to determine which combination (or variation) of treatments can be synergistic with each other. Each added treatment to a restoration project can increase cost and complexity. However, aside from the Road 108 ripping project, a major message from this DCP project is that often multiple restoration treatments applied in combination synergistically can achieve the most success. This is exemplified in several projects, such as the synergistic positive effects of topsoil salvage + planting in the Northshore Project, pelleting + seeding (as well as protection) in the Jean Seeding project, early results in the Goodsprings Fire Planting project where shelters were required with outplanting, and in many other examples.

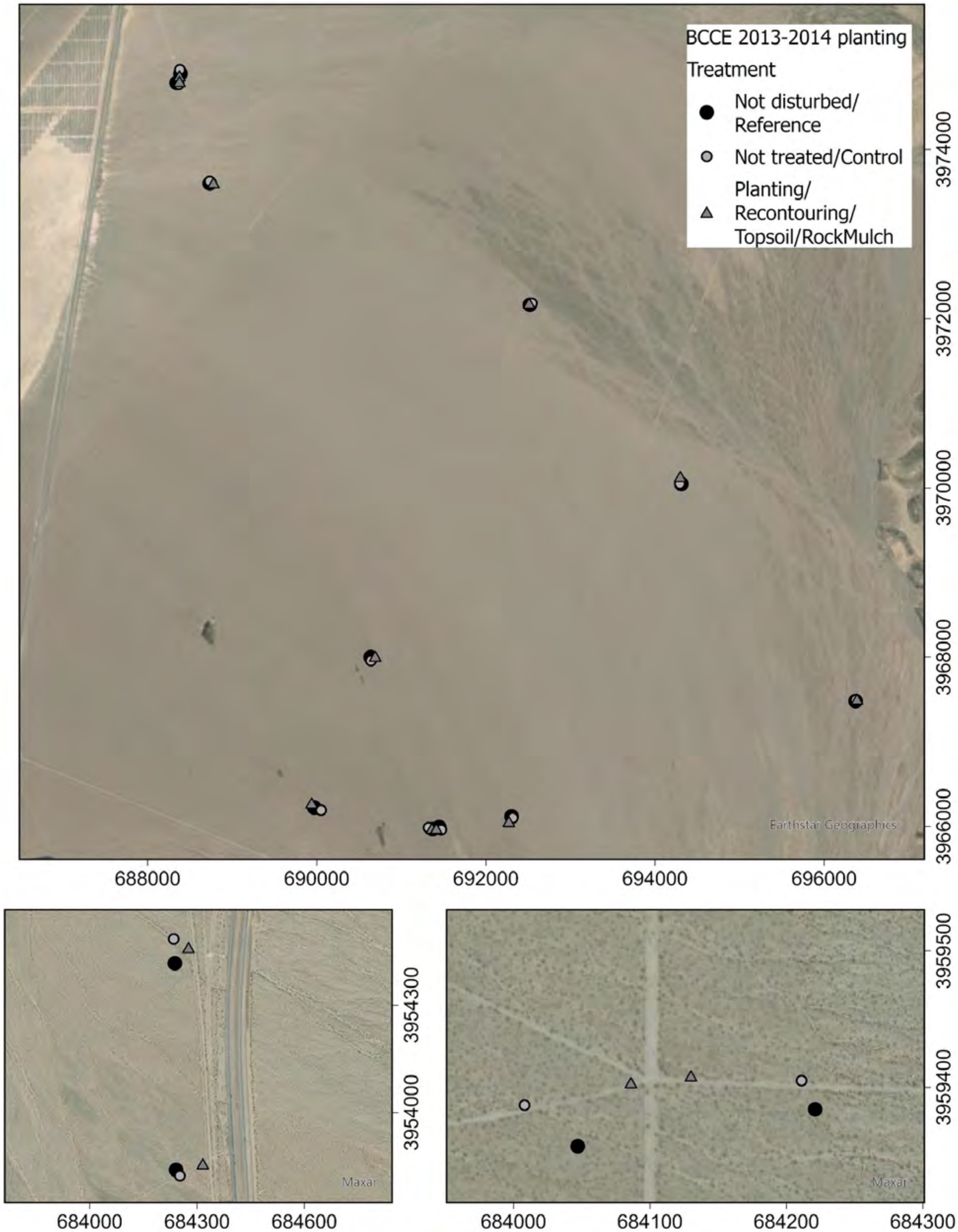
INDIVIDUAL PROJECT DESCRIPTIONS AND OUTCOMES

For the 15 individual restoration projects, this section of the report provides a description of each project's restoration treatments and summarizes their outcomes and interpretation. For at least five of the projects (ones with pre-existing, long-term data), we plan on developing case study scientific manuscripts reporting long-term data (including data collected in years preceding) this DCP project and the 2024 DCP project data collection. One of these manuscripts (Bonnie Springs Fire seeding) is already complete and is included as an appendix to this report as an example. At the end of this report, we also discuss plans for the other four case studies which can be a value-added contribution to this DCP project. In all of the project maps, coordinates are Universal Transverse Mercator (UTM, m), zone 11, North American Datum 1983.

Boulder City Conservation Easement 2013-2014 Planting, Soil Amendment

This project occurred within the 35,316-ha Boulder City Conservation Easement (35°44'37.67"N, 115° 0'15.58"W), a protected area established in 1995 and administered by the Clark County Desert Conservation Program. The easement is southwest of Boulder City in southern Nevada, eastern Mojave Desert. The project sought to reestablish native plant cover on decommissioned, dirt roads (~ 4 m wide) and to visually blend the road disturbances into the surrounding creosote bush-white bursage (*Larrea tridentata-Ambrosia dumosa*) shrubland landscape. Using a backhoe, the soil surface of the decommissioned roads was ripped to a depth of 15-30 cm, which also resulted in bringing rocks back to the surface. Next, mature creosote bush and white bursage plants were salvaged from a donor site in Henderson, Nevada (within ~ 16-20 km of the restoration sites) and transplanted the same day to the ripped road sites. The transplants were placed in patterns mimicking the surrounding reference habitat at a density ~ 15-75 transplants/100 m^2. A layer of salvaged topsoil from the donor site was also spread (depth not known) beneath the canopy of each transplant. Immediately after planting, the soil below the canopy of each transplant was watered to field capacity. It is believed that no further watering occurred, as transplant survival was considered desirable but death and conversion to vertical mulch was also considered acceptable. Restoration activities were performed between November 2012 and January 2013 and in February 2014.

In the 2024 inventory, transplanted shrubs, frequently appearing dead, were visible at the sites and shown in photos such as below. The data were consistent with this observation, as restoration plots contained an average of 2800 dead shrubs/100 m^2, compared with 1440/100 m^2 on disturbed, control plots. These dead plants can serve as vertical mulch, as shown on the example site photos. Total native perennial and annual cover was similar between restoration and control plots. Food plants reported in the literature as favored by desert tortoises averaged 2.9% cover in restoration, 2.2% in control, and 2.6% cover in undisturbed reference plots. Some of these tortoise food species common on the restoration plots included the native annual forbs *Amsinckia tessellata*, *Chaenactis fremontii*, *Chorizanthe brevicornu*, *Chorizanthe rigida*. There were also more isolated occurrences of *Malacothrix glabrata*. Although survival of the transplanted perennials may have been low, we note that once dead, the plants became vertical mulch and provided structure. We propose that microhabitats below the vertical mulch may have aided the recruitment of annual plants, which in this case, included tortoise forage plants without a noteworthy increase in non-native plants. The cover of non-native plants was lower on average in restoration (0.20%) than in disturbed/control (0.25%) and undisturbed (0.30%) plots. For this planting project, using vertical mulch may have provided similar benefits as the transplants (which largely died) more inexpensively and with less effort.


BCCE 2013-2014 planting
Treatment
Not disturbed/ Reference
Not treated/Control
Planting/ Recontouring/ Topsoil/RockMulch
Earthstar Geographics
688000
690000
692000
694000
696000
3966000
3968000
3970000
3972000
3974000
Maxar
684000
684300
684600
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684100
684200
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3959400
3959500

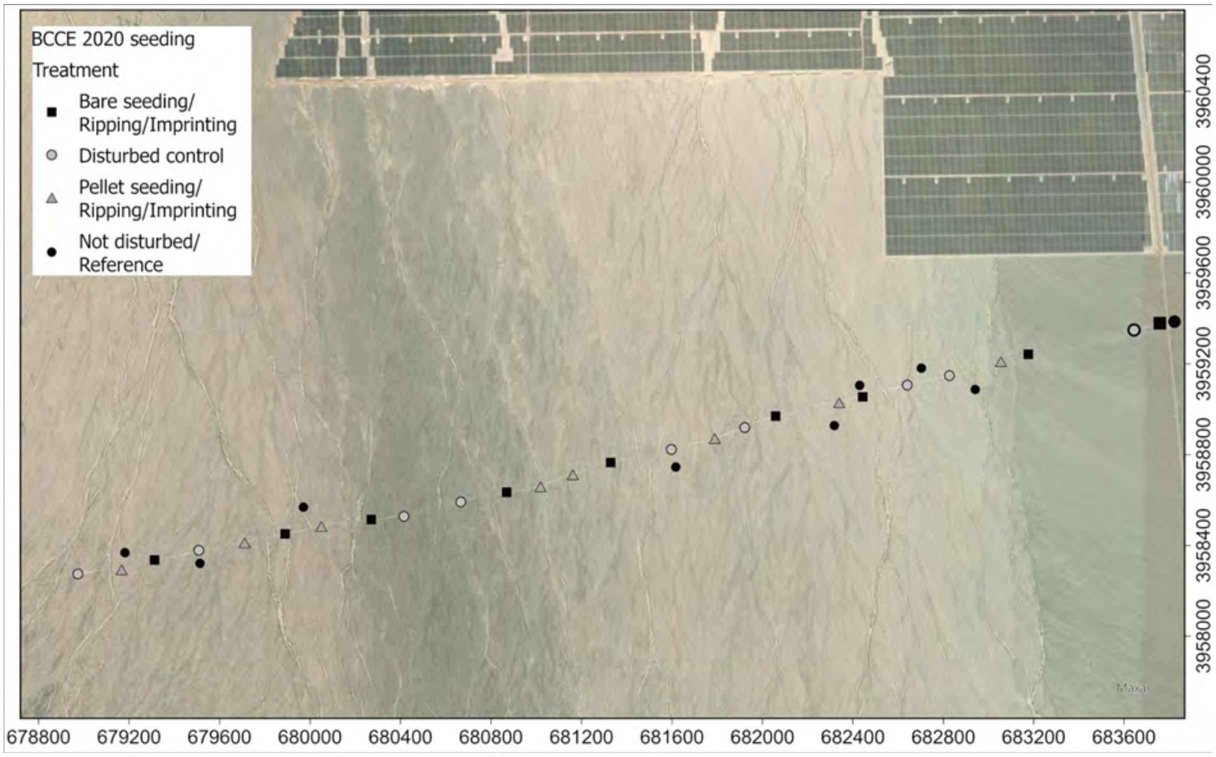
Example restoration plot that was ripped and outplanted with creosote bush and white bursage (left) and control (right), Boulder City Conservation Easement 2013-2014 Planting project. Photos taken in March 2024 by UNLV staff (BCCE01P,C).

Based on the minimal increase in native perennial plant cover observed in 2024, 10 years after transplanting, a question is why or when the transplant mortality occurred and whether further treatments could have increased survival. While we do not know the timing of when mortality occurred, this underscores the value of monitoring planting survival and restoration success for many years after treatment. It is possible that mortality occurred before the onset of the post-2019 dry period southern Nevada experienced, or the mortality could have occurred during the drought. Our on-site soil assessment showed that some restoration plots on the decommissioned road had high soil compaction ratings, and most of the soils were gravelly and sandy. On the road surfaces with these soil conditions, hydrology may not have been favorable for transplant survival, particularly during drought. Water infiltration may have been limited on the surfaces, coupled with high runoff from the road surface away from the transplants and to the sides of the decommissioned roads. Assuming that the transplants did survive the transplanting operation and were locally adapted, it is possible that further soil manipulations or irrigation may have been needed to increase transplant survival on the road surfaces either in the short- and longer-term.

Boulder City Conservation Easement 2020 Seeding, Soil Amendment

This project occurred within the 35,316-ha Boulder City Conservation Easement (35°44'37.67"N, 115° 0'15.58"W), a protected area established in 1995 and administered by the Clark County Desert Conservation Program. The easement is southwest of Boulder City in southern Nevada, eastern Mojave Desert. Intending to reestablish cover of native plants and visually blend the disturbance into the surrounding creosote bush-white bursage (*Larrea tridentata-Ambrosia dumosa*) shrubland landscape, this project occurred on a decommissioned, dirt road approximately 4 m wide. Behind a barrier closing the road, the road was divided into 24 segments, each 185 m long and separated from each other by 10 m. Three seeding treatments (none, bare seed, and pelletized seed, described below) were implemented in alternating segments of the road, with 8 segments for each treatment. Before seeding, the soil surface on the road was ripped to a depth ~ 15 cm and imprinted (creating a series of V-shaped indentations, ~ 10 cm deep intended to roughen the surface and catch seeds, organic material, and water) using heavy equipment. A broadcast seeding at a rate of 4.5 Kg/ha of pure live seed was then applied using a seed spreader pulled by a tractor. The treatment included pelletized (commercial pine mulch and tackifier) and non-pelletized seed of four native perennial species. The four species and percentage by pure live seed weight in the seed mix included: the perennial grasses Indian ricegrass (*Achnatherum hymenoides*; 38% of the mix) and sand dropseed (*Sporobolus cryptandrus*; 5%), the perennial forb desert globemallow (*Sphaeralcea ambigua*; 38%), and the annual forb desert bluebells (*Phacelia campanularia*; 19%). Seed was purchased from Granite Seed (Tempe, AZ) and was non-locally sourced from Washington (Indian ricegrass), Colorado (sand dropseed), Arizona (desert globemallow), and New Mexico (desert bluebells). These species were selected based on potential ability to colonize disturbances and were not necessarily major components of undisturbed reference habitat adjoining the decommissioned road. The ripping and seed were performed in December 2020.

Regarding results, three of the seeded species (*Phacelia campanularia*, *Achnatherum hymenoides*, and *Sporobolus cryptandrus*) were not recorded on any plot in 2024. The fourth seeded species, *Sphaeralcea ambigua*, occurred on about half (7 of 16) of the seeded plots. Although this species does occur in the study area generally, it did not occur in any of the disturbed, non-seeded plots nor in the undisturbed reference habitat containing our plots. Thus, it seems possible that the occurrences on the seeded plots originated from the seeding. There was little difference in other vegetation metrics between seeded and non-seeded plots. This observation is not trivial, as the act of introducing seed, even if seedling establishment is minimal, could affect the recipient community in various ways such as through the behavior of seed-eating fauna. As shown in the phots on the next page and in the data where native cover averaged only 0.3% on seeded plots, the seeding did not appear to substantially increase native cover, even with the minimal *Sphaeralcea ambigua* establishment. Many factors could have affected the outcome, such as the origin of the seed, the road surfaces minimally trapping seed or providing few favorable microsites (with favorable hydrology) for plant establishment, very dry conditions after seeding, granivory, or other factors. It is possible (but not known) that merging abiotic treatments, such as vertical mulch, with seeding could increase chance for success. Supported by Clark County, there is currently an ongoing project within the BCCE to evaluate whether abiotic treatments (vertical mulch, topographic depressions, litter with seed transfer) can result in an ecologically reliable and cost-effective strategy for habitat restoration on roads.

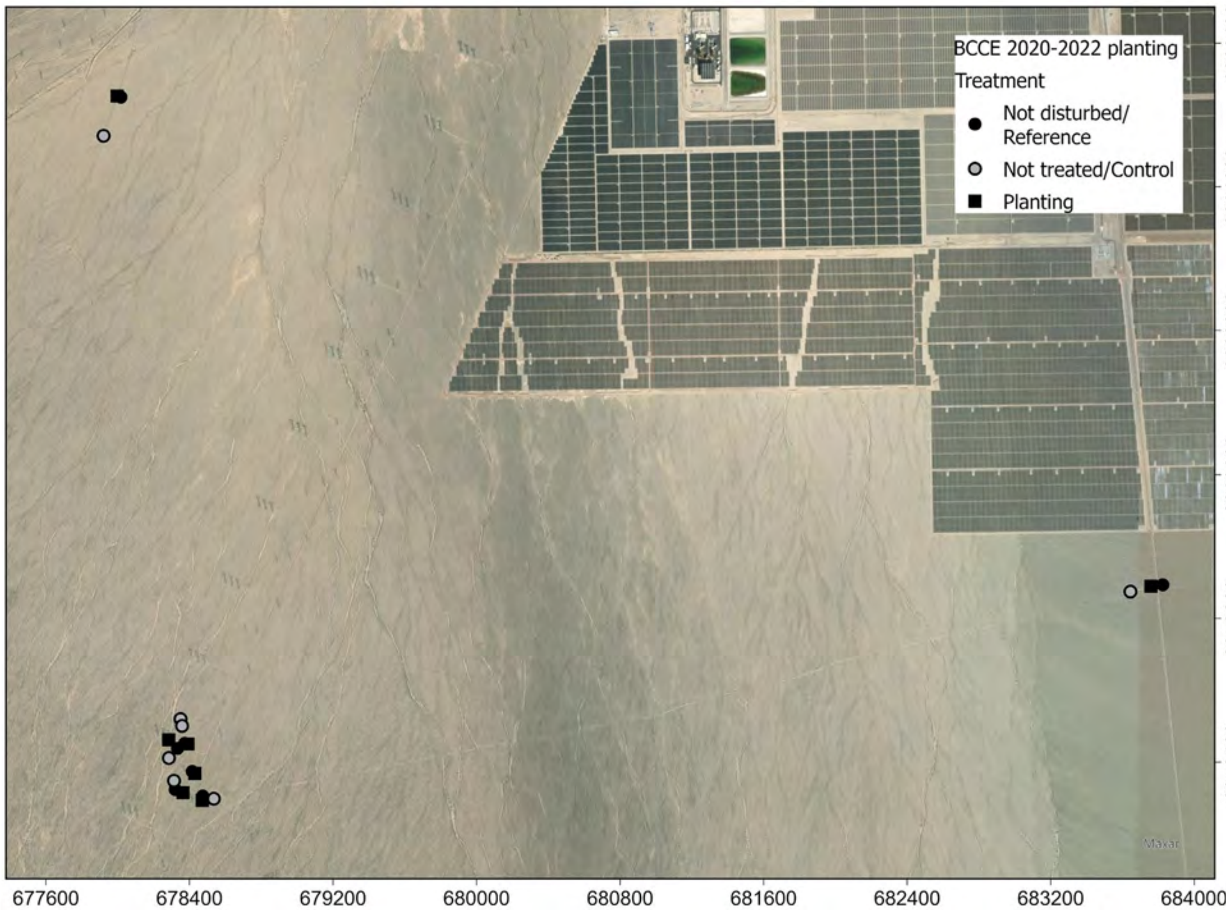


Example restoration plot that was ripped, imprinted, and seeded (left) and control plot (right), Boulder City Conservation Easement 2020 Seeding project. Photos taken in March 2024 by UNLV staff (plots K2-05, K2-04).

Boulder City Conservation Easement 2020-2022 Planting, Soil Amendment, Mulch

This project occurred within the 35,316-ha Boulder City Conservation Easement (35°44'37.67"N, 115° 0'15.58"W), a protected area established in 1995 and administered by the Clark County Desert Conservation Program. The easement is southwest of Boulder City in southern Nevada, eastern Mojave Desert. Intending to reestablish native plant cover, visually blend the disturbances into the surrounding creosote bush-white bursage (*Larrea tridentata-Ambrosia dumosa*) shrubland landscape, and deter future unauthorized vehicle incursions, this project occurred along five decommissioned dirt road areas behind their closure points. The dirt roads were approximately 4 m wide and generally without side berms. The closure points had boulders or dirt barriers (1+ m high) placed across the roads. In an approximately 0.1-ha area behind the closure points, the soil surface of each decommissioned road was ripped to a depth of 15-30 cm. Rock mulch, consisting of cobbles (64-256 mm in diameter) and boulders (> 256 mm) sourced from around the roads, was then applied to the ripped surface at a density ~ 1-5 cobbles or boulders/m^2 (higher density for cobbles and lower for boulders). Vertical mulch, consisting of 2-6 dead branches (primarily of creosote bush) placed vertically into the soil (reaching a height of ~ 1-1.5 m above the soil) to simulate a dead shrub, was sourced from adjacent areas and emplaced at a density ~ 10 structures/100 m^2. Finally, mature individuals of creosote bush and white bursage were salvaged from a site ~30 km away and were transplanted to the restoration sites. The transplants were placed at a density ~ 10 transplants/100 m^2. Soil below the plant canopies was watered to field capacity at the time of planting. Subsequently, the soil below the transplant canopies was watered to field capacity monthly for six months after planting.

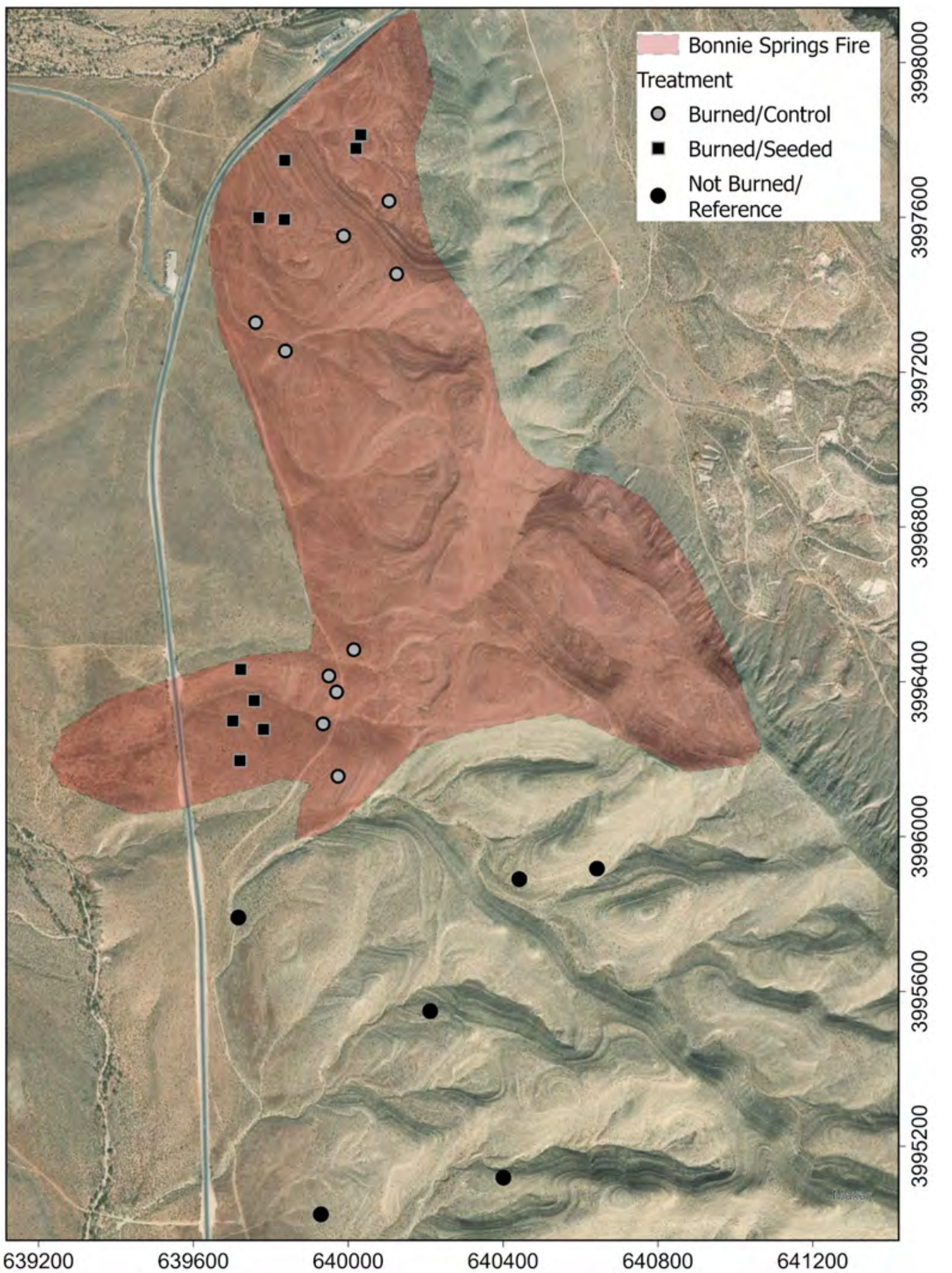
Although as of 2024 and still early in the restoration process (just 2-3 years after treatment), the project thus far has resulted in much higher average density (1400/100 m^2) of live adult shrubs (though still small size) on restoration plots compared with disturbed, untreated control plots (57 adult shrubs/100 m^2). In fact, the density on restoration plots almost approached the 1686/100 m^2 density on undisturbed reference plots. The planted species *Ambrosa dumosa* and *Larrea tridentata* comprised much of the cover on restoration plots. Non-native annual cover was minimal (~ 0.01% on average) across all plots, indicating that restoration did not promote non-native plants. However, native annual cover, including food plants for desert tortoises, also had low cover (<< 0.25%) on restoration plots. Potentially the integration of multiple abiotic treatments (ripping, addition of rock mulch and vertical mulch) coupled with transplanting resulted in this project thus far producing benefits in terms of increasing abundance of foundational desert shrubs, including creosote bush which is a shrub that desert tortoises use as a cover plant.



Example restoration plot that was ripped and received rock mulch, vertical mulch, and salvaged transplants (left) and control (right), Boulder City Conservation Easement 2020-2022 Planting project. Photos taken in March 2024 by UNLV staff (plots I1P, C).

Bonnie Springs Fire Seeding

The study site was the 158-ha Bonnie Springs Fire (36°6'7"N, 115°26'29"W), within the 80,000-ha Red Rock Canyon National Conservation Area, administered by the U.S. Bureau of Land Management (BLM), 15 km west of Las Vegas, Nevada. Ignited on July 2, 2007, the wildfire was contained by July 4, 2007. The fire was primarily fueled by a mixture of senesced non-native annual plants and native shrubs in a mid-elevation, mixed-shrub community including *Coleogyne ramosissima* (blackbrush). A weather station 5 km away recorded an average of 18 cm/year of precipitation (1991 through 2024), 67% falling from October through March.



A study of 32 wildfires since 1980 in the eastern Mojave Desert surrounding our present study site indicated that cover of native perennial plants (though not necessarily comprised of pre-fire species) in burned *C. ramosissima* communities required an average of 40-50 years to fully recover to levels found on unburned areas (Abella et al. 2021). With a goal of increasing and accelerating recovery of native plant cover, the BLM organized a restoration seeding after the Bonnie Springs Fire using native species from the local flora that prior research showed are capable of being early post-disturbance colonizers (Abella 2010, Vamstad and Rotenberry 2010). The BLM purchased 300 kg of pure live seed ($63,000 in 2008 USD) in total of five native species collected from within Clark County, Nevada (the county containing our study site). The seed vendors were Comstock Seed (Gardnerville, NV), Native Seed (Park City, UT), and Granite Seed (Lehi, UT). The species included a perennial grass (Aristida purpurea [purple threeawn]), two perennial forbs (*Baileya multiradiata* [desert marigold] and *Sphaeralcea ambigua* [desert globemallow]), an annual forb (*Salvia columbariae* [chia]), and a shrub (*Hymenoclea salsola* [cheesebush]). In 40 ha of the wildfire BLM prioritized for restoration, the seeding was implemented in November 2008 with field personnel using hand-held seed spreaders to broadcast the seed mix of the five species at pure live seeding rates listed below. The November timing was chosen to enable potential exposure of the seeds to autumn and winter-spring rains for emergence and growth during the 2009 and subsequent growing seasons. We obtained a random sample of 120 seeds each for four of the five seeded species on which to conduct a laboratory germination test. The test revealed that 20-100% of the pure live seed was readily germinable under laboratory conditions when seeding occurred. The table below is excerpted from a case study manuscript presented in the Appendix of this report.

Table 1. Summary of five native plant species seeded on the Bonnie Springs Fire, Mojave Desert, southern Nevada.

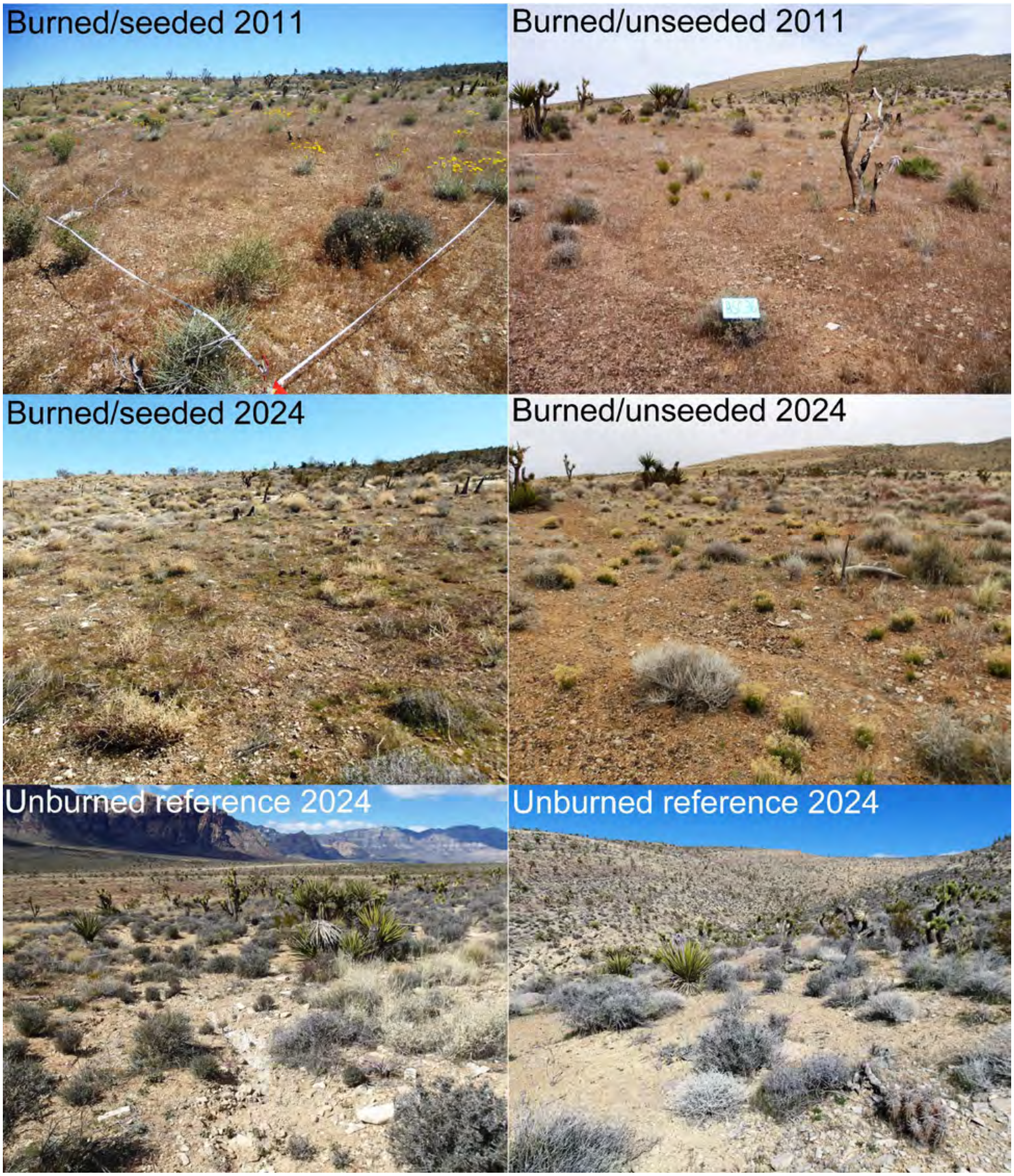
Feature	*Aristida purpurea*	*Baileya multiradiata*	*Hymenoclea salsola*	*Salvia columbariae*	*Sphaeralcea ambigua*
Growth form	Perennial grass	Perennial forb	Shrub	Annual forb	Perennial forb
Pure live seeds/100 m^2	4400	13,600	2200	16,600	12,300
Seed weight (mg)[a]	0.7	0.5	3.9	–[c]	1.0
Germination (%)[b]	100	70	20	–	30
	Average plant density/100 m^2 ± SEM (% frequency[d])				
Seeded 2009	2 ± 2 (10)	51 ± 26 (100)	2 ± 2 (30)	0 ± 0 (0)	92 ± 35 (100)
Seeded 2010	1 ± 1 (20)	33 ± 20 (90)	1 ± 1 (20)	20 ± 20 (20)	43 ± 10 (90)
Seeded 2011	1 ± 1 (20)	34 ± 17 (100)	1 ± 1 (10)	0 ± 0 (0)	28 ± 5 (100)
Seeded 2024	14 ± 6 (60)	79 ± 27 (100)	1 ± 1 (20)	0 ± 0 (0)	55 ± 7 (100)
Unseeded 2009	10 ± 7 (20)	29 ± 11 (80)	1 ± 1 (20)	2 ± 2 (10)	182 ± 67 (90)
Unseeded 2010	1 ± 1 (20)	10 ± 4 (100)	1 ± 1 (20)	0 ± 0 (0)	76 ± 22 (100)
Unseeded 2011	1 ± 1 (20)	11 ± 4 (90)	2 ± 1 (40)	8 ± 8 (10)	40 ± 9 (100)
Unseeded 2024	14 ± 9 (70)	39 ± 12 (100)	3 ± 1 (50)	0 ± 0 (0)	61 ± 12 (100)
Reference 2024	0 ± 0 (0)	8 ± 5 (83)	3 ± 3 (17)	0 ± 0 (0)	37 ± 13 (100)

[a]Average weight of air-dry, individual seeds we measured from samples of seed lots used for the post-wildfire seeding.

[b]Using 120 seeds/species randomly selected from seed lots, we assessed germination (radicle protrusion) by placing seeds on moistened paper towels in Petri dishes at room temperature for six weeks in a laboratory lighted from outside windows.

[c]Not available.

[d]Frequency is out of 10 plots each for seeded and unseeded areas on the burn and for six plots in unburned reference habitat. Plant densities are provided by measurement year (2009-2011, 2024). For the species (*B. multiradiata*, *S. ambigua*) with sufficient detection, average density did not vary with seeding ($p > 0.05$), but *S. ambigua* varied across years ($F_{3,72} = 14.6$, $p < 0.001$).



Example photos from the Bonnie Springs Fire seeding, one of the projects for which long-term data are available (since 2009) for plots which were re-sampled in 2024 during the DCP project. The upper left photo pair taken from the same location show a seeded plot, while the upper right photo pair show an unseeded plot. Unburned, unseeded reference habitat examples are shown in the bottom photo. The seeding had no apparent effect.

A full discussion of the results is presented in the case study manuscript for the Bonnie Springs seeding project in the Appendix of this report. A brief summary is presented here.

There was little evidence in any of our analyses that post-wildfire seeding increased abundance of seeded species during our study period including the first three (2009-2011) and 16 years (2024) after seeding. Individual seeded species either had low density and occurrence among burned plots with no trend for greater abundance with seeding (*A. purpurea*, *H. salsola*, and *S. columbariae*) or were common but had no significant differences between seeded and unseeded plots (*B. multiradiata* and *S. ambigua*). Totaled for the five seeded species, average density and cover of seeded species did not vary ($p > 0.05$) with seeding, instead varying only with year. In 2024, density of seeded species in both the seeded and unseeded burned areas were higher than in unburned (and unseeded) reference habitat. Likewise, using cover in indicator species analysis, no seeded species in 2024 was associated with seeding. The four seeded species still present in 2024 (*S. columbariae* was absent) were primarily associated with burned habitat, independent of seeding, when compared with unburned habitat.

We detected only one of the five seeded species in soil seed bank samples collected in both 2009 and 2010 (2-3 years post-fire and 1-2 years post-seeding). The only seeded species we detected was *S. columbariae*, occurring in seed banks of only one of the 10 seeded plots.

Including *S. columbariae* as the only seeded species detected, native species dominated seed banks in terms of the number of species (16 of 20 total species, 80%). However, non-native species dominated in terms of seed density, comprising 87-93% of all detected seeds. The non-native annual grass *B. rubens* and the annual forb *E. cicutarium* in sum had densities averaging 1213-5436 seeds/m^2. Although with sharply lower densities always averaging < 106 seeds/m^2 for individual species, the predominant native species we detected in seed banks were annual forbs such as *Lepidium lasiocarpum* (shaggyfruit pepperweed), *Phacelia fremontii* (Fremont's phacelia), *Plantago patagonica* (woolly plantain), *Plantago ovata* (desert plantain), *Cryptantha* spp., and *Amsinckia tessellata* (bristly fiddleneck).

In comparing variation in seeded and unseeded burned areas across years (2009-2011 and 2024 spanning 1-3 and 16 years post-seeding), most (10 of 13, 77%) of the univariate vegetation variables differed only through time and not with seeding nor its interaction with year. Among the other three variables, cover of desert tortoise food plants did not vary significantly with seeding or time. Non-native annual forb cover (dominated by *E. cicutarium*) varied with a seeding × year interaction overall, though none of the seeded-unseeded comparisons differed within years. Lastly, native perennial species richness was the only variable to differ with the seeding main effect, but perennial richness was lower (12 ± 1 species/100 m^2, mean ± standard error of mean) in seeded compared with unseeded areas (14 ± 1 species/100 m^2).

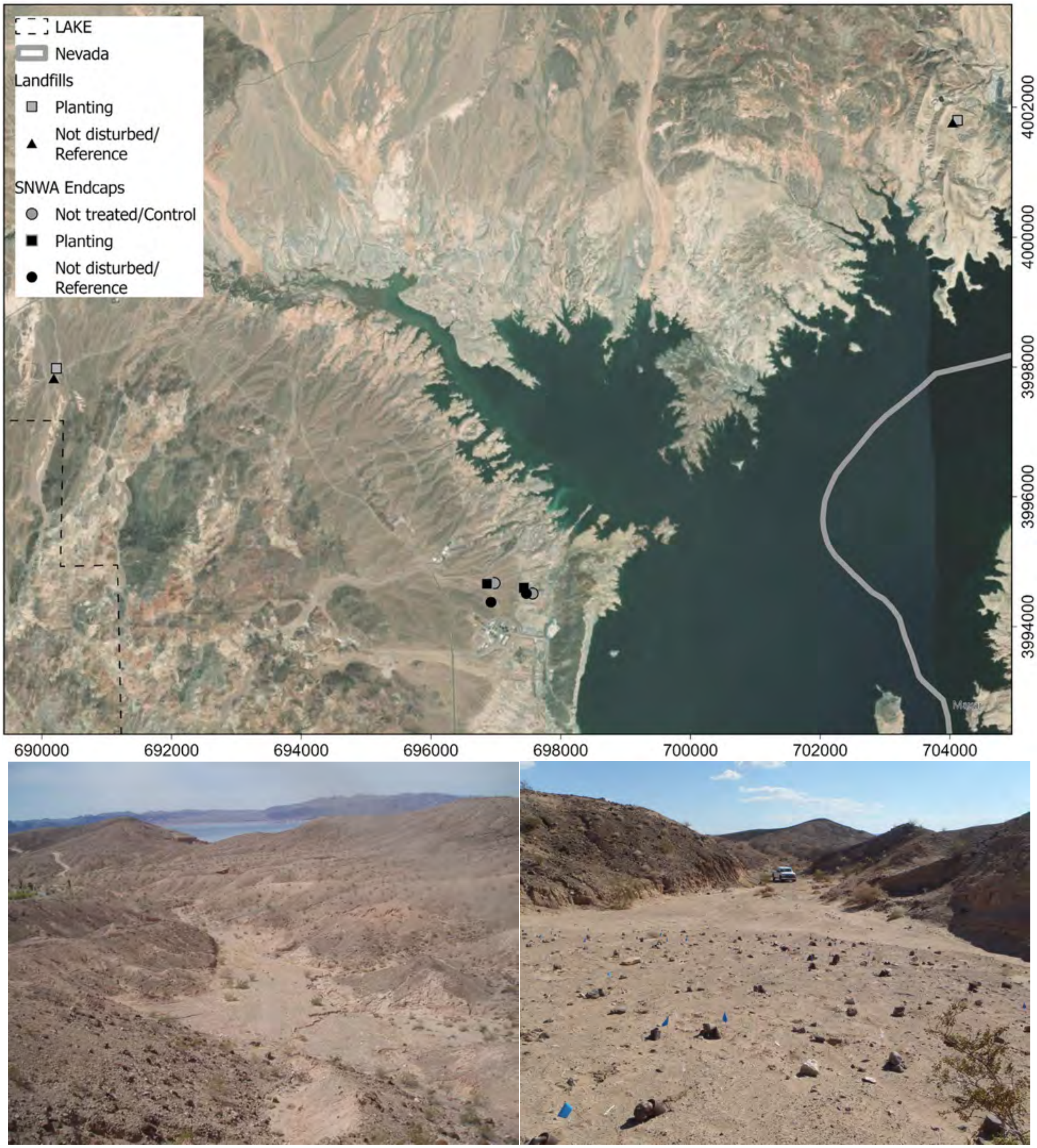
In 2024 comparisons of seeded and unseeded areas on the burn with unburned reference habitat, the 13 univariate vegetation variables either did not differ across the habitats or generally just differed between the burn (irrespective of seeding) and unburned reference habitat. An exception was non-native species variables (annual forb and grass cover and *B. rubens* biomass) were higher only in the burned, seeded areas than in the unburned reference habitat.

In multivariate analyses, burned seeded and unseeded areas never differed ($p > 0.05$) in community composition in any study year. Both seeded and unseeded burned areas, however, differed from unburned reference habitat in 2024. In follow-up indicator species analysis, no seeded or non-seeded species were associated with seeding in 2024 comparisons between seeded and unseeded burned areas. When compared with unburned reference habitat, several species were associated with burned areas irrespective of seeding, such as the native perennial forb *S. ambigua* (a seeded species), perennial grass *Dasyochloa pulchella* (low woollygrass), annual forb *Eriogonum deflexum* (flatcrown buckwheat), and the non-native annual grass *B. rubens*. Other natives, such as the shrubs *C. ramosissima* and *Y. schidigera* and the perennial forb *Astragalus nuttallianus* (small-flowered milkvetch), were associated with unburned habitat.

The attached full manuscript in the Appendix of this report provides interpretation as to potential reasons why the seeding was not effective and management considerations for future seeding projects. We suggest that despite following best practices for seeding (e.g., using locally collected seed known to be viable and appropriate species), this seeding did not enhance native plant establishment for restoration after wildfire. We suggest that this underscores that further work is necessary to understand the germination requirements, seed fates, and detailed factors associated with seeding success (or lack thereof). Experiments in the lab and small field trials on small plots seem prudent to figure out these details before trying to skip forward to landscape-scale restoration. It is important to recognize that landscape-scale treatments do not represent “scaling up” of successful restoration application if the landscape treatments are not successful. Figuring out what may be successful in small areas may be a prudent strategy first, then working on scaling up the treatments that have succeeded in small areas.

Callville Bay Landfill Planting

This project occurred on a 0.5-ha landfill (36° 8'21.27"N, 114°43'56.24"W, 33 km east of the City of Las Vegas) within Lake Mead National Recreation Area, administered by the National Park Service, in southern Nevada in the eastern Mojave Desert. The landfill was in a depression surrounded by low hills, 50-100 m higher in elevation than the landfill, in creosote bush-white bursage (*Larrea tridentata-Ambrosia dumosa*) shrubland. The landfill contained scrap metal, glass, and assorted waste deposited on the soil surface as well as an access road through the center. The site was graded, with mostly flat soil and minimal plant cover. After removing the waste material, in March 2012, the National Park Service implemented outplanting intending to reestablish native plant cover on the disturbance. Using seed collected from within the recreation area and propagated at the park’s nursery (Song Dog Native Plant Nursery, Boulder City, NV), the following native shrub species and numbers of individuals were outplanted: *Ambrosia dumosa* (white bursage; 152 plants), *Bebbia juncea* (sweetbush; 276), *Encelia farinosa* (brittlebush; 303), and *Hymenoclea salsola* (cheesebush; 274). These species all occurred in nearby reference habitat on the low-lying areas and hillslopes surrounding the disturbance. At the time of outplanting, each outplant was provided with one slow-release irrigation gel (DRiWATER Inc, Santa Rosa, CA). Application of the gel followed manufacturer recommendations by placing an 8-cm diameter plastic tube into the ground, angled toward plant roots and with the top (covered with a plastic cap) near the soil surface. A cylindrical DRiWATER gel was then inserted into the buried tube. The amount of water delivered by the gel is variable, dependent on how much water that roots extract (DRiWATER Inc, Santa Rosa, California). One new irrigation gel was provided to each live outplant in July and August 2012. Additionally at the time of outplanting, half of the outplants received a mulch treatment (the other half did not).



Callville Bay landfill restoration site, shown in 2008 before restoration (left) and in September 2012 after restoration (right). The photo on the right shows the slow-release irrigation gel tubes installed around outplanted seedlings. The plots for the Callville Bay site are in the northeastern corner of the map above.

None of the four outplanted species, all of which are capable of colonizing disturbances, were recorded on our plot in this small site in 2024. Instead, as shown in the photo example below, the non-planted *Larrea tridentata* (1% cover) was the main perennial on the plot. *Larrea* had 4% cover on the plot in adjacent reference habitat and may have been able to readily spread to the disturbed plot. The small size of this disturbance, coupled with the landfill being in a depression that may have collected water, potentially aided colonization by the normally slow-colonizing *Larrea*. The other natural colonizer of the restoration plot was the native annual forb *Plantago ovata* (1.5% cover), which is a desert tortoise food plant. Although total native cover on the restoration plot was low at 6%, the reference plot also had only 6% total native cover.

Although we cannot rule out the possibility that the planting in 2012 resulted in transient establishment of the planted species that somehow facilitated recruitment of the non-planted species, we suspect that the present plant cover on the restoration plot arose from natural colonization that may have occurred anyway in the absence of restoration. Interestingly, the outplanting species are known to be early colonizers (or versatile in the case of *Ambrosia dumosa*) yet did not become established through restoration (at least about a decade later), while the usually later-colonizing *Larrea* did become established naturally. It is possible that outplanting *Larrea* directly, instead of the early colonizers, could have hastened *Larrea* establishment and produced Larrea cover already similar to that found on nearby reference habitat. Another conclusion from this project is that installation of the slow-release irrigation gel is expensive and labor intensive, but apparently did not result in any outplants surviving. We conclude that potentially further work to identify whether these types of small disturbances – close to natural seed sources nearby – and disturbances such as this landfill that receive supplemental moisture will require restoration could aid future restoration planning.

Example restoration plot that was outplanted (left) and compared with a reference plot (right), Callville Bay Landfill Planting project. Photos taken in March 2024 by UNLV staff (plots CVBP, R).

Fish Hatchery Planting and Soil Amendment

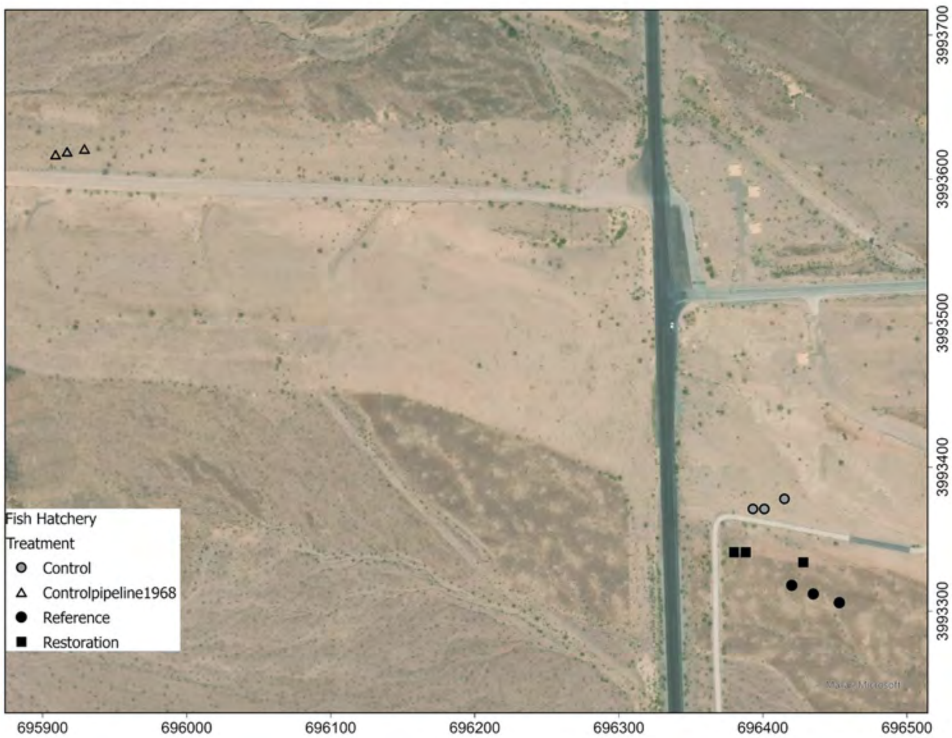
This site was in Lake Mead National Recreation Area, Clark County, Nevada, 30 km east of the City of Las Vegas at an elevation of 400 m (36° 3'57.36"N, 114°49'9.14"W). The study area consisted of a 0.21-ha area in each of four adjacent locations: a 1998 corridor receiving no restoration treatments (hereafter untreated 1998 corridor), an adjacent section of the same corridor that received restoration treatments (hereafter treated 1998 corridor), an adjacent *Larrea tridentata* community off the corridor that served as a control, and a corridor cleared in 1968 adjacent to the 1998 corridor. This study of a pre-existing disturbance and an operational management activity is limited by a lack of replication; however, the study area comprises one landform (an alluvial fan) and one soil association (Carrizo-Carrizo-Riverbend, primarily consisting of Typic Torriorthents; Lato et al. 2006). This supports an assumption that potential differences among the four areas primarily result from their successional age or the restoration treatments, rather than from pre-existing environmental differences. Both the treated and the untreated 1998 corridor were cleared by blading with heavy equipment, with the upper 20 cm of soil stockpiled and reapplied after construction. The 20-cm depth may have varied slightly depending on rockiness or other factors. The 1968 corridor also was cleared by mechanical blading, but topsoil was not replaced (David Connally, Southern Nevada Water Authority, personal communication).

Restoration treatments applied by the National Park Service in January-February 1999 to the 1998 corridor included hand-raking the soil surface after soil replacement to re-spread rocks, applying artificial desert varnish (Permeon™, Soil-Tech, Inc., Las Vegas, NV) evenly to the soil surface for color restoration, and planting *Larrea tridentata* (96 plants), *Ambrosia dumosa* (12 plants), *Opuntia basilaris* (9 plants), and *Acacia greggii* (2 plants). Desert varnish is a brown-black coat given its color by iron and manganese oxides. This varnish commonly forms on stable surfaces on volcanic rock, a process that is hastened by the chemical application of artificial desert varnish. The planting treatment is detailed in Newton (2001). By 2001, no planted *A. greggii* or *A. dumosa* were alive, but survival was 92% for *L. tridentata* and 100% for *O. basilaris* (Newton 2001).

Based on the 2024 re-inventory, 25 years after restoration, one of the conclusions from the Fish Hatchery site restoration project is that the different metrics used to assess restoration effectiveness can result in contrasting conclusions. The second control area, which was disturbed in 1968 and did not receive restoration, continues to have a higher density and species richness of native perennials than the restoration area. However, these species remain primarily early colonizing species. Meanwhile, although with a lower density and richness of plants, the restoration area is resembling the undisturbed reference habitat in terms of containing a widely spaced *Larrea* community with a well-developed annual community. In fact, in terms of desert tortoise annual food plants, the restoration plots contained an average of 5.2% cover of annual tortoise food plants compared with 4.2% in the reference. This was much higher than the average 0.4% cover of tortoise annual food plants in the unrestored control for the 1998 disturbance. *Plantago ovata* comprised much of the annual tortoise food plants, with 3.5% cover in the restoration area compared to only 0.1% cover in the control.

As shown in the photo below, the artificial desert varnish applied to rocks seems to still be persisting 25 years later. Our perception is that the rocks in the restoration area appear darker and more varnished than the lightish rocks in the control. The raking and soil surface treatment, potentially coupled with maturation of the annual plant community and creosote bushes, may be associated with the greater average soil stability in the restoration compared with the control.

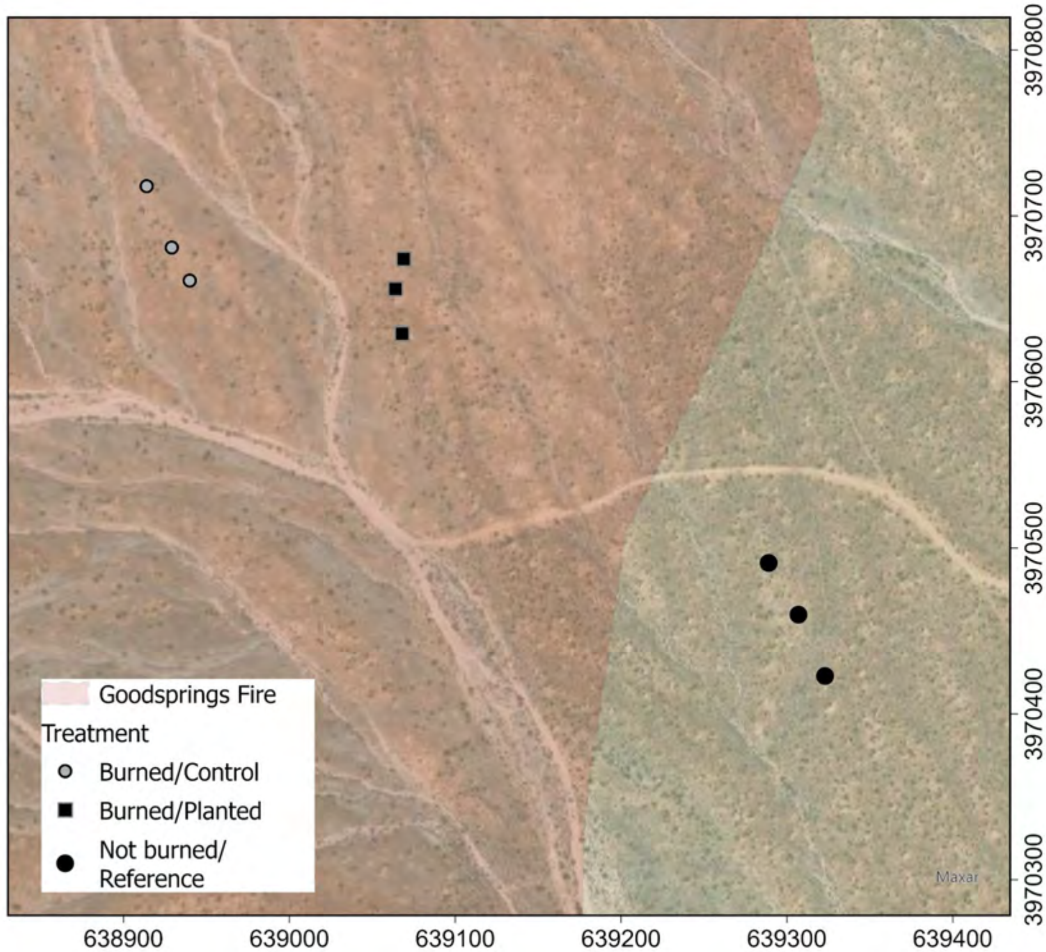
Overall, we conclude that without restoration, the 1968 and 1998 disturbances still contain a mixture of early colonizing perennials which do provide floral resources. It is possible that the timing of both disturbances, followed by a notably wet decade in the 1970s and a notable wet period in the 1990s, enabled more rapid and complete colonization of these disturbances by early colonizers despite the low elevation and low average rainfall of the area. However, these disturbances in the absence of restoration sharply deviate from undisturbed references. The implementation of restoration appears to have enabled the restoration area to converge in many metrics with the undisturbed reference by 25 years post-restoration. One of these important benefits is that restoration has created conditions for the natural recruitment of desert tortoise forage plants, which were orders of magnitude greater in cover on restoration than control plots.



Example restoration plot that received salvaged topsoil, artificial desert varnish on the rocks, and outplanting (left) and an untreated control plot (right), Fish Hatchery Planting and Soil Amendment project. Photos taken in March 2024 by UNLV staff (plots FH01, FH04).

Goodsprings Fire Planting

The experimental site (35º52'08"N, 115º27'31"W, 25 km southwest of the City of Las Vegas) was on Bureau of Land Management land within the 2005 Goodsprings Fire. This fire was ignited by a June 22 lighting strike and burned 13,585 ha of desert shrubland with native fuels augmented by exotic annuals such as *Bromus rubens*. At an elevation of 1,250 m, the site was on a generally flat plain containing burned woody skeletons indicative of a mixed-shrubland community dominated by white bursage (*Ambrosia dumosa*), creosote bush (*Larrea tridentata*), and blackbrush (*Coleogyne ramosissima*). When we initiated the study three years after the fire, the burned area was typical of initial post-fire environments in the Mojave Desert: the mature, late-successional shrub community was lost, annual plants remained sparse, and colonization by early successional perennials such as desert globemallow (*Sphaeralcea ambigua*) was sporadic (Steers and Allen 2010). Soils were mapped as loamy, mixed, superactive, thermic, shallow Typic Petrocalcids of the Irongold series (Lato et al. 2006).



Seeds for the outplanting experiments were collected locally in Clark County and included a range of native perennial forb, grass, and shrub species that regional land managers are interested in working with in revegetation projects (listed in the table below). We divided shrubs into two categories: 1) early successional species that are initial colonizers of disturbance and 2) early-late succesional species that can either be initial colonizers and components of mature communities or late-successional species (Abella 2010). To prepare outplants, seedlings were grown for a year in 4-L (15 cm in diameter and 20 cm tall) plastic pots filled with 2:1 sand:organic potting soil in local greenhouses. We kept plants in the containers, rather than creating bareroot stock, because our field site was accessible (so added weight of containers was not a concern) and we wished to retain the full root mass and potting soil during outplanting (Bainbridge 2007).

The outplanting experiment was a randomized complete block design, containing species (10 native perennial species), irrigation (present or absent), and shelter (present or absent) factorially applied as 40 combinations each appearing once in each of the 10 blocks. This resulted in 40 total outplants per species and 400 overall. Blocks (replicates) were 80-m-long transects containing treatments assigned at random distances (1-m increments) along transects. Transects were separated from each other by 3 m. Plants were 1-year-old when outplanted on 12 February 2008. After digging holes with shovels, we carefully ensured that the full root mass and soil from pots was transferred to the ground. We immediately provided each outplant with 1 L (0.25 gal) of water. Shelters (10 cm diameter × 60 cm tall [4 × 24 in] green plastic cylinders, Tubex® tree shelter brand, South Wales, United Kingdom) were installed over appropriate plants and fastened to the ground with bamboo sticks (Figure 1). The irrigation treatment consisted of installing DriWater™ gel packs (DriWater, Inc., Santa Rosa, California) in plastic tubes open in the soil adjacent to the root mass following manufacturer-recommended procedures. DriWater was recharged after 3, 5, 7, 15, and 20 months.

Species outplanted and their survival percentage three years after outplanting on the Goodsprings Fire. There were 40 outplants for each of the 10 species, totaling 400 outplants.

Species	Survival (%)
Forb	
Baileya multiradiata	0
Penstemon bicolor	3
Sphaeralcea ambigua	55
Grass	
Aristida purpurea	0
Muhlenbergia porteri	5
Sporobolus airoides	3
Early successional shrub	
Encelia farinosa	0
Early-late shrub	
Ambrosia dumosa	23
Eriogonum fasciculatum	28
Larrea tridentata	23

This project had pre-existing results (Abella et al. 2012). The table below summarizes the early results of outplant survival, which was 14% (55 of 400 total outplants) in May 2011 (three years after outplanting). However, outplant survival was highly variable among species, ranging from 0% to 55% for *Sphaeralcea ambigua*. Although overall outplant survival was low, a high percentage (60%, or 33 outplants) of the 55 surviving outplants were flowering in May 2011. In addition to providing potential floral resources for pollinators and other invertebrates, flowering by outplants suggests the potential they could produce seed and establish self-sustaining populations.

In 2024 regarding the status of species that had been outplanted, the early results of outplant survival and flowering through 2011 (three years after outplanting) appeared to be predictive of which species would be abundant in 2024. Five of the six species with < 23% survival in 2011 were absent in 2024, with the sixth species (*Bailey multiradiata*) present only at trace amounts. In contrast, the four species with higher survival in 2011 had higher cover in 2024 on plots where they had been outplanted or at least persisted there. In 2024, the outplanted *Ambrosia dumosa* had 2.2% cover in restoration plots and only 0.3% in disturbed, unrestored plots. *Eriogonum fasciculatum* only occurred in plots where it had been outplanted and was absent from disturbed, unrestored plots. Although *Larrea tridentata* was detected in all plots where it had been outplanted, outplanting did not appear to increase its cover relative to the few individuals that resprouted or recolonized on unrestored plots on the burn. This was also the case for *Sphaeralcea ambigua*. Thus, with regard to the 10 outplanted species, outplanting was correlated with either increased cover or at least persistence on outplanted plots 16 years later.

Although directly linking non-native cover with the outplanting action is difficult, restoration plots did have slightly lower cover (31%) of non-native annuals compared with unrestored controls (39%). Unburned reference habitat, although still heavily invaded by non-natives at this mid-elevation site, had the lowest non-native cover with 19%. It is not known whether competition from outplants, which would not necessarily have fully developed a fertile island (which can promote annuals in some cases), reduced non-native cover but may be possible.

There were few other differences in vegetation metrics between the burned restoration and burned unrestored plots. We conclude that 16 years after the outplanting, it may have retained a subtle effect on the species composition of the burned areas. Based on the 2011 results (3 years after outplanting), we also know that the outplanting did provide cover and floral resources at least for a period of time when few other plants occurred on the burned area. For a low-density outplanting, we conclude the project was at least partially a success. Using these same techniques but with a greater density of planting may have further accelerated post-fire recovery. We note that at an elevation of 1,250 m, this site was at a higher elevation and received more precipitation than some of the very low elevation sites. This, along with the practices of providing irrigation and protection (shelters) early on, may have enabled the outplanting to produce plants that became persistent components of the community.

Example restoration plot that received low-density outplanting (left photo) and a burned, untreated plot (right), Goodsprings Fire Planting project. Photos taken in March 2024 by UNLV staff (plots GSP3, GSC1).

Jean Large-Scale Translocation Site Seeding, Fencing

This existing seeding and fencing experiment occurred within a 10,600-ha site, managed by the Bureau of Land Management (Southern Nevada District), where release of translocated desert tortoises is authorized. The site is in Clark County, 35 km south of Las Vegas, and 10 km southwest of Jean, Nevada. The area is in the eastern Mojave Desert, a hot desert receiving most of its rainfall in winter. At an elevation of 800 m, our experimental site occupied a valley between low mountain ranges and typifies the creosote bush-white bursage (*Larrea tridentata-Ambrosia dumosa*) valleys with high desert tortoise habitat potential. Average density of mature perennial plants (> 10 cm tall) included 1,700/ha for creosote bush; 5,900/ha for white bursage; and scattered occurrences of cheesebush (*Hymenoclea salsola*), winterfat (*Krascheninnikovia lanata*), desert globemallow (*Sphaeralcea ambigua), littleleaf ratany (Krameria erecta), and big galleta grass (Pleuraphis rigida).* Major annual plants included the non-native grass Arabian schismus (*Schismus arabicus*) and the native forbs desert plantain, shaggy fruit pepperweed (*Lepidium lasiocarpum*), pebble pincushion (*Chaenactis carphoclinia*), devil's spineflower (*Chorizanthe rigida*), and broadfruit combseed (*Pectocarya platycarpa*). Soils had parent material derived from limestone, dolomite, and sandstone and are classified as Haplocalcids and Petrocalcids within the Weiser-Oldspan-Wechech association (Lato et al. 2006). A weather station, 5 km away, reported an average of 11 cm of rainfall/year during the 2008 through 2013.

The study area has a long history of grazing by livestock and feral animals, perhaps dating back to the 1500s when Spanish expeditions passed through. Commercial livestock operations and expansion of feral horse (*Equus caballus*) and burro (*Equus asinus*) populations followed, peaking in the late 1800s and early 1900s for livestock and through the mid-1900s for horses and burros. Commercial livestock grazing does not presently occur in the study area, but some feral horses and burros inhabit the area. Effects of potentially centuries of grazing and trampling of plant communities by livestock and feral animals are poorly understood. We do know that on contemporary landscapes, livestock and feral animals eat many of the same plant species favored by desert tortoises. Some of these plants include the native annual forbs desert plantain and desert dandelion (*Malacothrix glabrata*) and herbaceous perennials such as desert globemallow. Our study was performed to enhance present vegetation in ways anticipated as favorable to desert tortoises, through increasing favored forage plants and perennial plant cover above levels in existing vegetation.

We selected our experimental site (14 ha) because it was bisected by a little-used dirt road (~ 1 vehicle/day) that enabled access for implementing treatments, was typical of valley habitat in the translocation area, and was near a weather station. We located sampling units at least 5 m from the road to reduce potential roadside influences. The experimental design was a split-split plot, including the whole-plot factor of watering (present or absent), the subplot factor of seeding (none, bare seed, or pelletized seed), and the sub-subplot factor of fencing (present or absent). Each of the 12 treatment combinations was randomly assigned and replicated five times, totaling 60 sampling units.


Jean LSTS
Treatment
Bare seed/Fenced
Bare seed/Not fenced
No seed/Fenced
No seed/Not fenced
Pelleted/Fenced
Pelleted/Not fenced
646800
647200
647600
3950000
3950400
3950800
3951200
3951600
3952000
3952400
3952800

At the finest level of the experimental design (sub-subplot level), we constructed 30 fenced areas (each 10 m × 10 m) nested within the center of seeded and watered areas. Each fenced area had a paired unfenced area of equivalent size 4 m away. The wire fencing was 1 m tall, affixed to metal poles at the four corners, bent parallel to the ground at the base to deter burrowing animals, and had openings of 3 cm in the wire. We constructed the fence to deter large- and medium-sized herbivores (e.g., feral burros, jackrabbits [*Lepus californicus*]).

At the subplot level of the experimental design, seeding was conducted in areas of 24 m × 106 m, overlapping the fenced and unfenced areas. Either bare or pelletized seed of four native species was broadcast by hand on the umanipulated soil surface in January 2013. The seeded species included three native perennials, designed to augment cover (the shrubs cheesebush and winterfat) and forage (the herbaceous forb desert globemallow). We seeded the species at approximately the following densities: 5,000 seeds/m^2 for cheesebush, 1,700 seeds/m^2 for winterfat, and 13,000 seeds/m^2 for desert globemallow. The annual forage species – desert plantain – was seeded at 5,300 seeds/m^2. These seeding densities reflected the maximum density feasible from the seed we had available for each species. The seed lot was obtained by the Bureau of Land Management from an eastern Mojave Desert seed zone of Clark and southern Nye County, Nevada, and eastern San Bernardino County, California. For the pelletized seed treatment, seed was coated using Gro-Coat® (Seed Dynamics, Inc., Salinas, CA). The Gro-Coat® substance included a coating of mineral and organic material and binding (e.g., clay, starches, sugars) to hold the coating together. We assayed germinability of the seed lot, with methods and results presented in the source publication (Abella et al. 2015b). At least 10% of seed among species and seed types, and up to 98%, was readily germinable in a greenhouse.

At the whole-plot level of the experiment, a watering treatment was applied beginning the day after seeding (January 31, 2013) and on February 28, March 15 and 28, and April 12 and 24, 2013. Watering was performed using a sprayer affixed to a tanker truck, filled with Las Vegas municipal water. The truck was driven along the road and water was sprayed evenly across the five blocks receiving watering. On each of the first five watering dates, the treatment delivered 20,000 liters of water, or 0.5 mm of water over the soil surface. With the intention of maintaining forage as green as possible, the last event (April 24, 2013) delivered 2.5 mm of water. In sum, the watering treatment delivered 0.5 cm of water, doubling the amount of natural rainfall (0.4 cm) that fell during February through April 2013.

Within a total of 60, 8 m × 8 m sampling units (centrally located in fenced and paired unfenced areas) corresponding to each of the 5 replicates of the 12 treatment combinations, we measured the complete vascular plant community including seeded species during the first year after treatment in 2013. Note that for sampling in 2024 for the DCP project, we measured the original 8 m × 8 m plots and an expanded area to total 10 m × 10 m for each plot to standard plot size across the entirety of the DCP project. We performed measurements during the spring growing season (April, 3 months after seeding) and during fall (November, 10 months after seeding). In entire 64-m^2 sampling units, we visually categorized cover (to the nearest 1%) by species of all vascular plants. Cover was defined as the area of ground covered by live foliage and stems rooted in plots. A ground coverage of 0.5 m^2 in the 64-m^2 plots corresponded with 1% cover. We further counted the number of perennial plants (including seedlings). To estimate densities of annual species, we counted annual plants within 5, 1 m × 1 m quadrats evenly spaced within each sampling unit. During all measurements, we included both live and dead annual plants because desert tortoises eat senesced plants (Esque et al. 2014).

The first-year, short-term results after treatments were of greatest interest for testing reliability of augmenting forage in the context of a desert tortoise translocation occurring that year. But from a standpoint of sustainability of the one-time seeding and for perennial plant establishment, longer-term results were of interest. Thus, we re-visited plots 20 months after seeding on September 24, 2014. Annual plant cover was sparse then, so we focused on categorizing cover (in 0.01% increments) of the seeded desert plantain in the five quadrats per sampling unit. Additionally, we counted perennial plant seedlings in sampling units. These results were then further updated by data collection in spring 2024 for the DCP project.

As reported in the original publication (Abella et al. 2015b), early results for the first two years of seeding through 2014 were disappointing for the seeded perennial species but were encouraging for the seeded annual forage species, desert plantain. In summarizing the early results, seedling establishment of the three seeded perennial species was minimal during the experiment through 2014. Winterfat was not observed during any inventory. Live seedlings of cheesebush and desert globemallow were first observed during the fall 2013 inventory (10 months after treatments). At that time, cheesebush seedlings inhabited 3 of 20 (15%) pelletized-seeded plots and 5 of 20 (20%) bare-seeded plots. The species never had more than 5 seedlings/m^2 and was not analyzed statistically. Seedlings of desert globemallow were more common: they inhabited 13 of 20 pelletized (65%) and 9 of 20 (45%) bare-seeded plots in fall 2013. In the statistical analysis of fall 2013 globemallow seedling densities, seeding was the only significant term, as only seeded plots contained seedlings. Both cheesebush and globemallow were then absent in the fall 2014 inventory (20 months after treatments).

The seeded native annual forage species, desert plantain, displayed greater establishment than did the perennials. In spring 2013 (3 months post-treatment), pelletized seeding resulted in greater density of desert plantain than did bare seeding. The only interaction in the experiment for any 2013 variable occurred for desert plantain in fall 2013 (10 months after treatments). Seeding with fencing produced the highest density of 39 ± 18 plants/m^2 (mean ± standard error of mean, n = 10), compared to all other treatment combinations averaging 6 ± 1 plants/m^2 (n = 50). Within the factor of seeding, pelletized seeding resulted in significantly greater desert plantain density than bare or no seeding. Fencing also increased desert plantain, tripling its density.

In the fall 2014 inventory (20 months after treatments), the positive effect of fencing on desert plantain persisted. For desert plantain frequency, a fencing × irrigation interaction occurred, where unfenced, non-watered plots had lower frequency (27%) than did other treatment combinations (43-49%). With triple the cover and almost double the frequency on pelletized-, compared to unseeded or bare-seeded plots, the positive effect of pelletizing also tended to persist but was not statistically significant.

By 2024 (11 years post-seeding), a main finding is that consistent with the early results, the tortoise food plant desert plantain attained its higher cover in fenced plots receiving pelletized seed. The following table shows the 2024 percent cover of desert plantain. Demonstrating statistical significance is difficult, however, as occurrence of plantain was sporadic throughout the site. Nevertheless, this observation may underscore the biological significance of the finding, as tortoise food plants were limited in 2024 and thus occurrences of plantain on fenced, pelletized seeded-plots could have been important food resources.

Univariate statistics

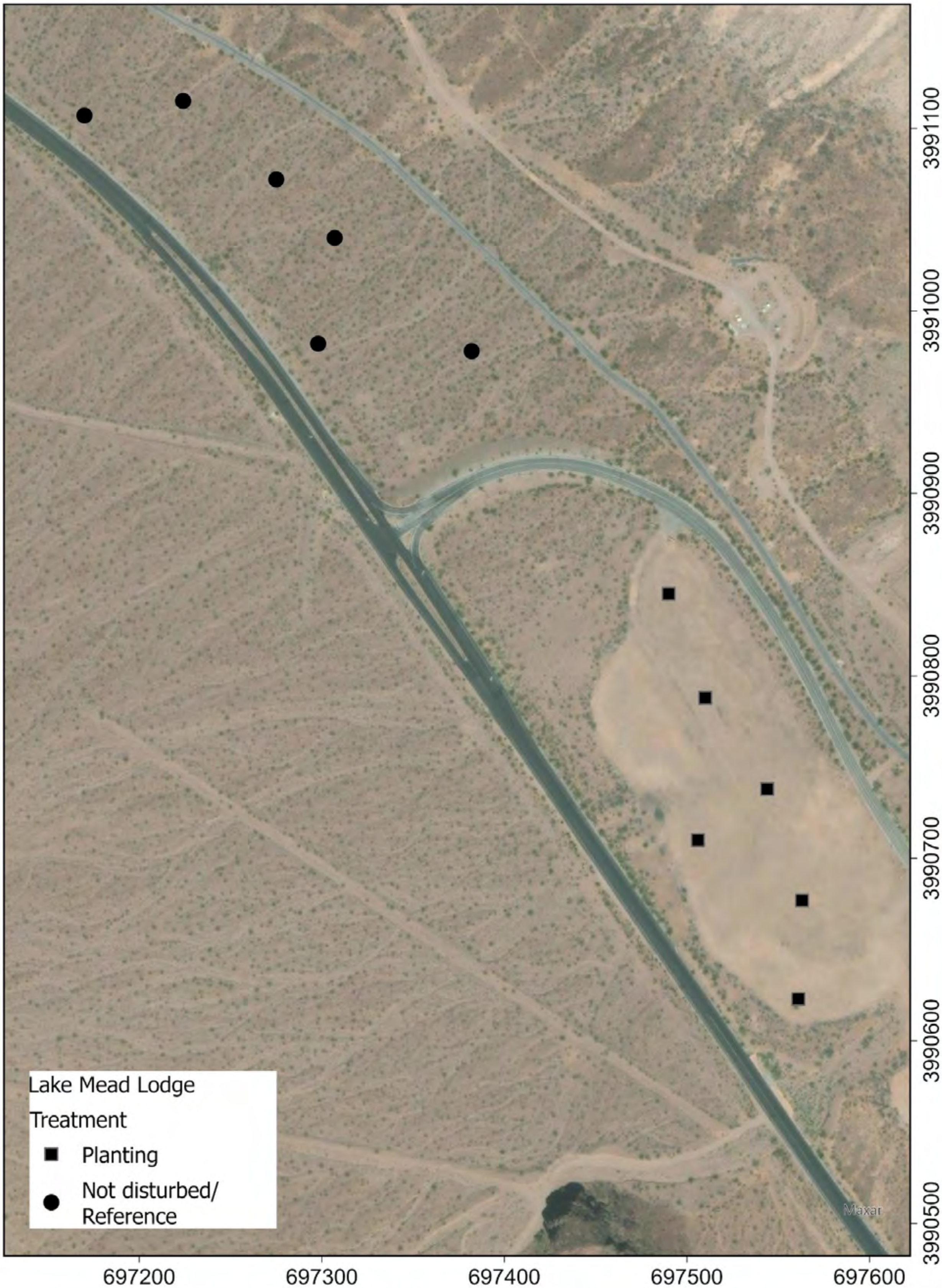
	NoSeedFenced	NoSeedUnfencec	PelletedFenced	PelletedUnfencec	BareSeedFenced	BareSeedUnfenc
N	10	10	10	10	10	10
Min	0	0	0	0	0	0
Max	0.25	0.5	0.5	0.1	0.25	0.25
Sum	0.48	0.73	1.41	0.43	0.65	0.47
Mean	0.048	0.073	0.141	0.043	0.065	0.047
Std. error	0.0257682	0.0491042	0.06158012	0.01556706	0.02587362	0.02595081

The major conclusion from this seeding and fencing project is that it appears that the combination of using fencing and pelletized seed can successfully boost availability of desert tortoise food plants for varying lengths of time. While a diverse menu may be important for desert tortoise nutrition rather than just one species, the annual forb desert plantain appears to have potential as being a core part of species mixtures when augmenting tortoise food plants is a goal. Seeding cheesebush and winterfat to augment cover plants for tortoises did not appear to be effective, though creosote bush may already occur at sufficient density to avoid having availability of cover plants as a limiting factor for tortoises. The seeded perennial forage species, desert globemallow, occurred at least at low cover on many plots. While the variation makes it difficult to ascribe occurrences to seeding, globemallow as a seeded species was available as a perennial forage plant for tortoises in 2024. Although pelleting does not work with all species, we recommend screening additional species for their amenability to pelleting as it did sharply increase the establishment of desert plantain in this project.

Example seeded, fenced plot (left photo) and unseeded, unfenced plot (right) plot, Jean Large-Scale Translocation Site (LSTS) Seeding project. Photos taken in March 2024 by UNLV staff (plots 6PF, 6CU).

Lake Mead Lodge Planting and Soil Amendment

This project occurred on a 3-ha disturbance (36° 2'26.96"N, 114°48'27.71"W; 383 m in elevation) where buildings (residential housing units), surrounded by creosote bush-white bursage (*Larrea tridentata-Ambrosia dumosa*) shrubland, were removed in 2021. The site was within Lake Mead National Recreation Area, administered by the National Park Service, in southern Nevada in the eastern Mojave Desert. Intending to reestablish native plant cover and blend the disturbance into the surrounding landscape, restoration included surface recontouring and soil ripping using heavy equipment followed by outplanting. Using seed collected from within the recreation area and including species inhabiting undisturbed reference areas adjacent to the restoration site, outplants were propagated in 4-L containers at the park's nursery (Song Dog Native Plant Nursery, Boulder City, NV). Outplanting occurred in two phases in 2022 in spring (March) and autumn (October). In the spring planting which occurred on about half the disturbance, outplants were placed at a total density of 104/ha and included four native shrub species in the following mixture: white bursage (*Ambrosia dumosa*; 69%), sweetbush (*Bebbia juncea*; 4%), brittlebush (*Encelia farinosa*; 8%), and creosote bush (*Larrea tridentata*; 19%). On the half of the disturbance for the autumn outplanting at a total density of 142/ha, six native species in the following proportions were outplanted: white bursage (*Ambrosia dumosa*; 33%), sweetbush (*Bebbia juncea*; 4%), brittlebush (*Encelia farinosa*; 7%), cheesebush (*Hymenoclea salsola*; 13%), creosote bush (*Larrea tridentata*; 36%), and desert globemallow (*Sphaeralcea ambigua*; 7%). All of these species were shrubs other than the forb-subshrub desert globemallow, and all inhabited nearby undisturbed reference areas. For both spring and autumn outplanting, planting holes received 8 L of water at the time of planting and another 8 L of water after seedlings were emplaced. Thereafter, outplants were given 8 L of water by hand weekly through October 2022. The autumn outplants were not watered following the initial 16 L of water at the time of planting.


Lake Mead Lodge
Treatment
Planting
Not disturbed/
Reference
697200
697300
697400
697500
697600
3991100
3991000
3990900
3990800
3990700
3990600
3990500
Maxar

Lake Mead lodge restoration site, where residential buildings were removed, the soil ripped and recontoured, and native perennials outplanted. The National Park Service provided these images, showing the outplanting operation in March 2022 and one year later (April 2023) when some live outplants are shown where planted next to the 4-L containers in the 2022 photo (photo point #5).

Lake Mead Lodge restoration site in March 2024 (two years after soil ripping and recontouring and outplanting), showing creosote bush (an outplanted species) present on the disturbance and dominant in the background undisturbed reference habitat.

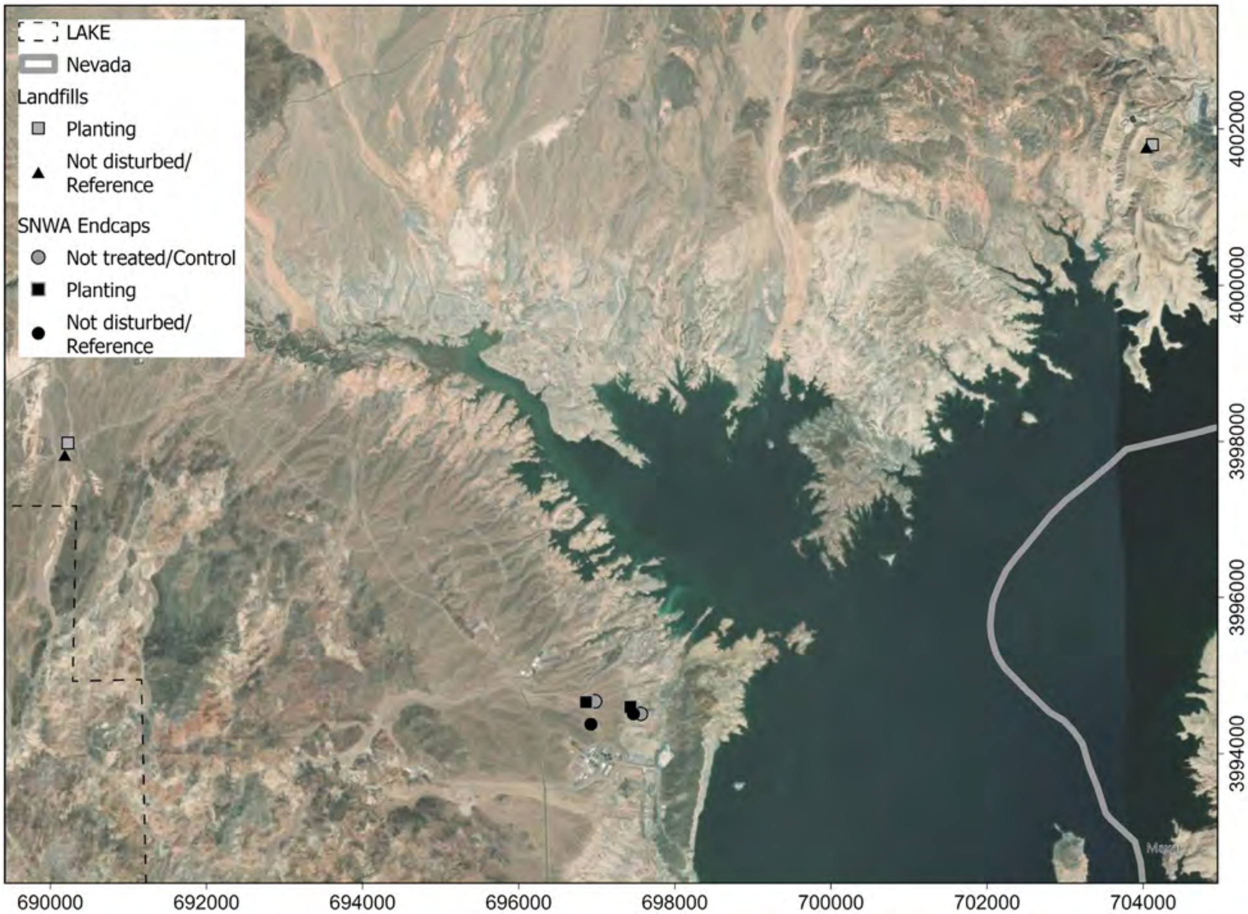
The 2024 assessment revealed that of the diverse mixture of outplanted species, the most successful were white bursage and creosote bush, which at least occurred on all restoration plots. The other species exhibited minimal establishment, typically occurring on just 1 or 2 plots if they occurred on plots at all. Although native cover remains much lower on average on restoration plots (3%) compared with undisturbed references (21%; note that no disturbed, unrestored area was available for sampling), just two years into the restoration there are establishing creosote-bursage which are the major components of reference habitat. Moreover, although the shrubs are small in the restoration area as may be expected, the restoration plots already contained 2/3 of the shrub density found in reference habitat. It is possible that regular watering, at least after the spring 2022 planting, enabled outplants to survive the dry weather occurring after outplanting.

Native annuals, including desert tortoise forage plants, averaged only 1% cover in restoration plots compared with 13% cover in reference habitat. This project site could provide an opportunity to determine whether the native annual community co-develops with maturity of surviving, outplanted shrubs potentially serving as nurse plants.

If the trajectory of outplant survival and growth persists, this project seems likely to be successful over the longer term. Another key conclusion from this project outcome is that restoration can still be successful (based on the early results) even when only a small proportion of the outplanted species survive. In this project, only a third (2 of 6) of the outplanted species became established in appreciable numbers. We would also note that like in the Callville Bay Landfill restoration, species with capability for colonizing disturbances naturally were not necessarily the most amenable to restoration. The four early colonizing species were the least successful in outplanting in this project, while the normally late-colonizing creosote bush and versatile white bursage were the most amenable to outplanting.

Las Vegas Bay Landfill Planting

This project occurred on a 0.25-ha landfill (36° 6'28.29"N, 114°53'11.97"W, 25 km east of the City of Las Vegas) within Lake Mead National Recreation Area, administered by the National Park Service, in southern Nevada in the eastern Mojave Desert. The landfill was in a depression 10 m lower than adjacent hills, which contained creosote bush-white bursage (*Larrea tridentata-Ambrosia dumosa*) shrubland. In March 2012, the National Park Service implemented outplanting intending to reestablish native plant cover on the disturbance. Using seed collected from within the recreation area and propagated at the park's nursery (Song Dog Native Plant Nursery, Boulder City, NV), 312 seedlings each of the following native shrub species were outplanted: *Ambrosia dumosa* (white bursage), *Bebbia juncea* (sweetbush), *Encelia farinosa* (brittlebush), and *Hymenoclea salsola* (cheesebush). These species occurred on surrounding hillslopes or drainages comprising undisturbed reference habitat. At the time of outplanting, each outplant was provided with one slow-release irrigation gel (DRiWATER Inc, Santa Rosa, CA). Application of the gel followed manufacturer recommendations by placing an 8-cm diameter plastic tube into the ground, angled toward plant roots and with the top (covered with a plastic cap) near the soil surface. A cylindrical DRiWATER gel was then inserted into the buried tube. The amount of water delivered by the gel is variable, dependent on how much water that roots extract (DRiWATER Inc, Santa Rosa, CA). One new irrigation gel was provided to each live outplant in July and August 2012. Additionally at the time of outplanting, half of the outplants received a mulch treatment (the other half did not).



The Las Vegas Bay Landfill site is in the western part of the map above.

Las Vegas Bay landfill, shown in 2012 when restoration commenced. Photo #P9170001 provided by the National Park Service.

Las Vegas Bay Landfill restoration plot shown in 2024. While some outplanted species like brittlebush are visible, the non-planted creosote bush is a dominant. Photo taken in March 2024 by UNLV staff (plot LVBP). There were no disturbed, unrestored control areas.

Based on the 2024 results, the restoration plot at the small site of the Las Vegas Landfill restoration had higher total native cover than the nearby reference plot on surrounding hillslopes. Total native cover averaged 9% on the restoration plot and 6% in the undisturbed reference plot. Adult shrub density was 3× higher on the restoration plot, as also illustrated in the photo above.

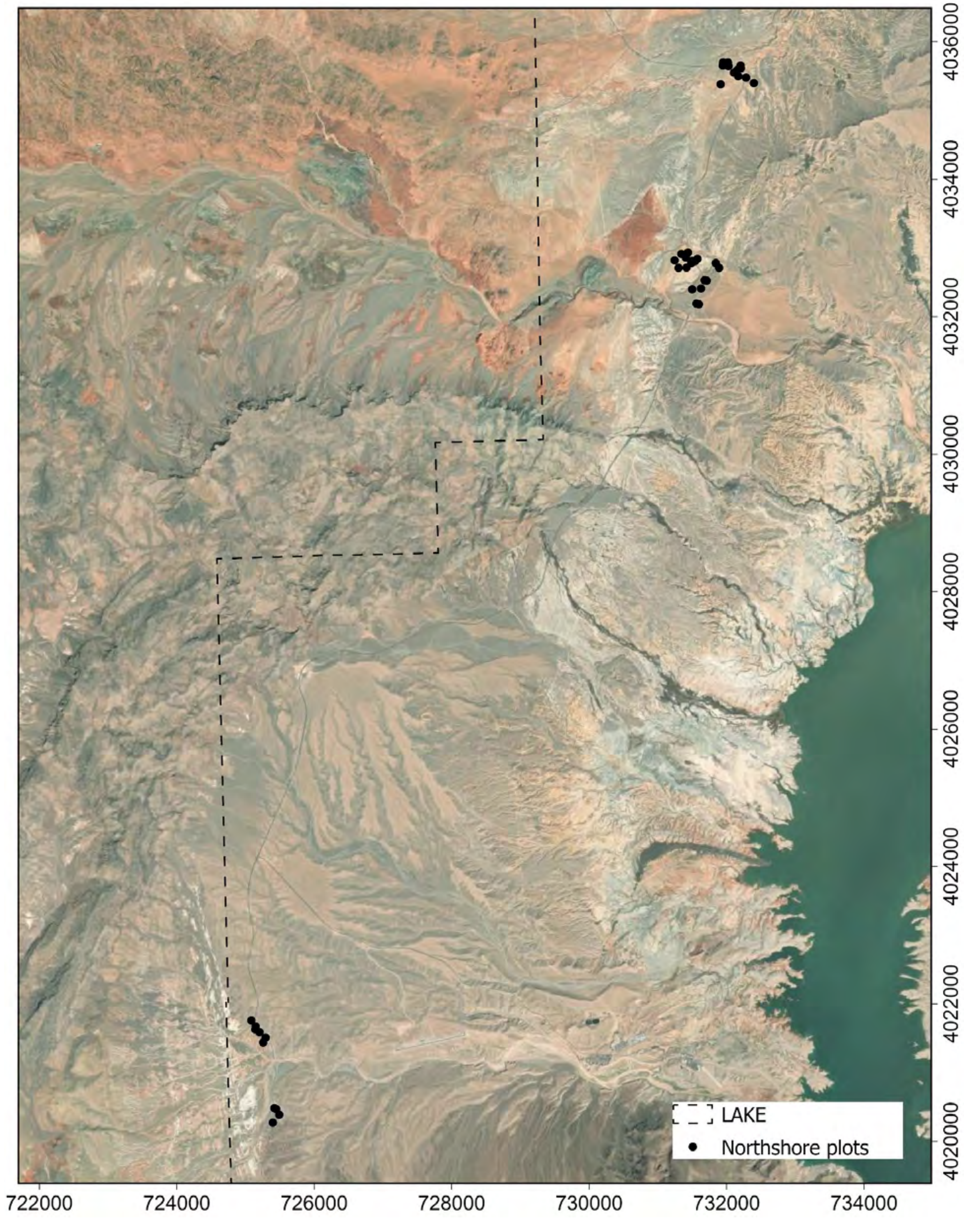
All four of the outplanted species occurred on the restoration plot: *Ambrosia dumosa* had 2.5%, *Bebbia juncea* 1.5%, *Encelia farinosa* 0.5%, and *Hymenoclea salsola* 0.5% cover on the restoration plot. However, as also occurred at the Callville Bay Landfill restoration, the native shrub with the highest cover (3.5%) on the restoration plot was *Larrea tridentata*, which was not outplanted and apparently colonized naturally. Interestingly, although native perennial cover was higher on the restoration plot, native annual cover was only 0.2% on the restoration plot compared with 2.5% on the reference plot. As also mentioned for the Lake Mead Lodge restoration site, determining whether an annual plant community develops in tandem with maturation of the young shrubs on the restoration site could be useful for assessing further community recovery and whether forage plants used by desert tortoises recover. Overall, we surmise that revegetation may have been particularly successful on this disturbance because of its small size and topographic setting of being in a concave location where water collected.

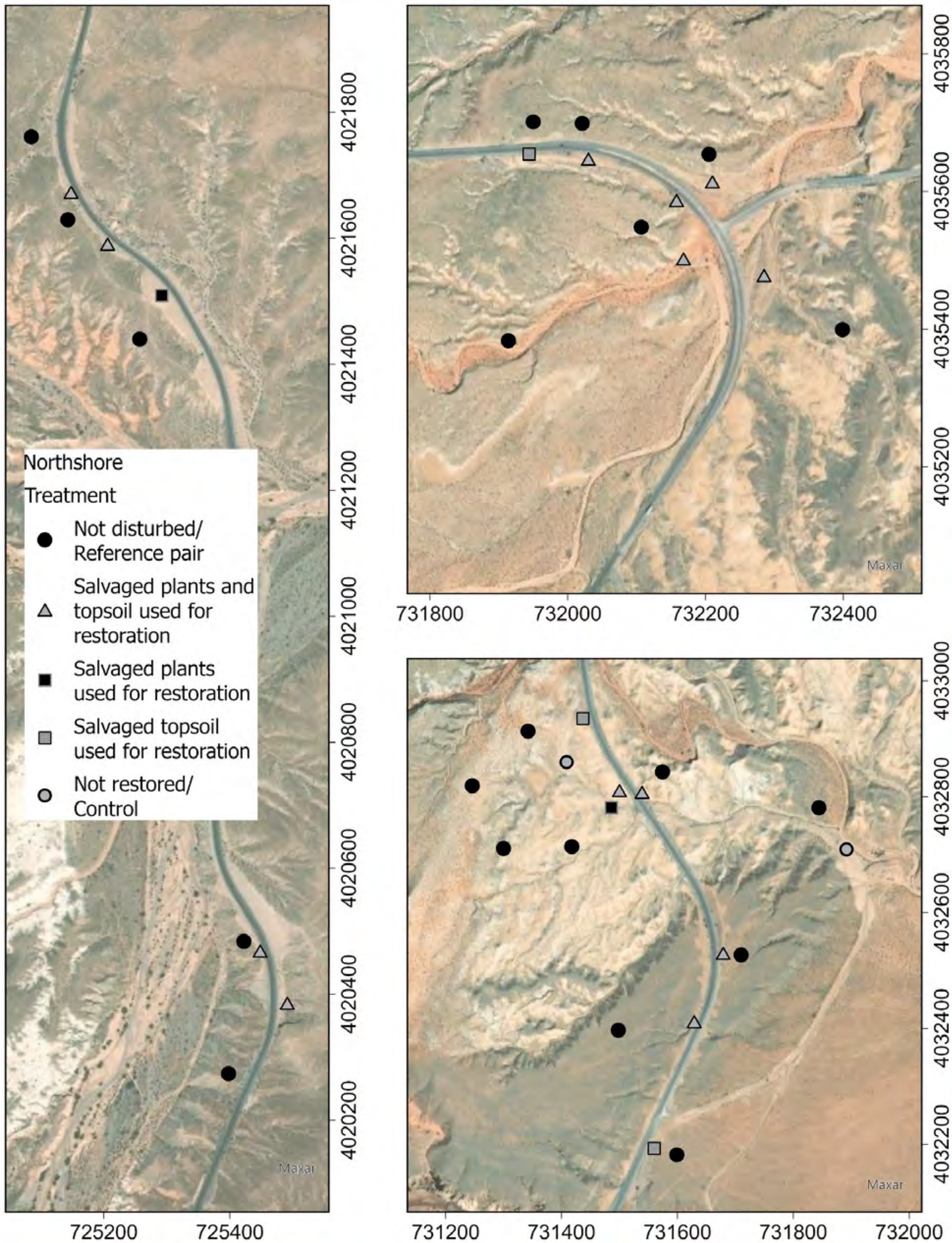
Northshore Road Planting and Topsoil Addition

This topsoil salvage and outplanting study occurred within the 563,513-ha Lake Mead National Recreation Area, managed by the National Park Service, in the eastern Mojave Desert. The Mojave is a hot desert receiving most of its precipitation in winter, with the remainder mainly monsoonal summer storms in July-August. A weather station near our experimental sites reported 1973-2012 averages of 16 cm/yr of precipitation (64% falling from November through April), 14°C January daily high, and 41°C July daily high (Valley of Fire State Park Weather Station, 610 m in elevation, Western Regional Climate Center, Reno, Nevada). Vegetation physiognomy is desert shrubland, with dominance by *Larrea tridentata*, *Ambrosia dumosa*, and *Atriplex hymenelytra* (Chiquoine et al. 2016). Livestock grazing is not authorized, but some trespass cattle and *Equus asinus* (feral domestic burro) inhabit the area.

Restoration occurred in three regions along Northshore Road containing soils classified as Gypsids and Calcids, suborders of Aridisols in U.S. soil taxonomy (Lato et al. 2006). Containing abundant gypsum or calcium carbonate, these soils have either gypsic or calcic horizons. As part of roadway maintenance, construction activities re-aligned Northshore Road to straighten and widen the road corridor. At each site beginning in 2008 using large machinery, the existing road pavement was torn up, topography re-contoured, and disturbed soil either smoothed or covered with topsoil salvaged from nearby areas to be destroyed by the new road.

Plant nursery for the salvaged Northshore Road plants in 2009 (salvage nursery care, left) and planting operations during January 2010 (planting, right). All transplants were caged using ¼ in. hardware cloth, anchored by rebar and landscape staples.


LAKE
Northshore plots
722000
724000
726000
728000
730000
732000
734000
4020000
4022000
4024000
4026000
4028000
4030000
4032000
4034000
4036000


Northshore
Treatment
Not disturbed/ Reference pair
Salvaged plants and topsoil used for restoration
Salvaged plants used for restoration
Salvaged topsoil used for restoration
Not restored/ Control
Maxar
725200
725400
4020200
4020400
4020600
4020800
4021000
4021200
4021400
4021600
4021800
731800
732000
732200
732400
4035200
4035400
4035600
4035800
731200
731400
731600
731800
732000
4032200
4032400
4032600
4032800
4033000

The table below shows the timeline of project activities and restoration treatments. Before construction activities, we salvaged perennial plants from the future new corridor of destruction using hand shovels to excavate as much of the root system as possible. We used bare-root salvage (i.e. minimal soil retained), which has an advantage of each plant being lighter in weight and easier to transport. We targeted plants to salvage that were larger than seedlings, but less than the 50th percentile of size, so that salvage operations and nursery processing could be done by hand. Twenty-three species of native perennials, including 2,105 total individuals in rough proportion to species abundance within the area to be destroyed, were salvaged in fall 2008. Each plant was numbered uniquely to track it throughout the early part of the study.

Events during the topsoil and transplanting restoration study along Northshore Road in Lake Mead National Recreation Area, Mojave Desert.

Events	Timing	Description
Salvage + treatments	Oct 2008	Plants salvaged, treated with IBA, slurry, or water
Construction	Nov 2008-Dec 2009	Old road removed, site re-contoured
Salvage nursery care	Oct 2008-Jan 2010	Plants reside in pots with drip irrigation
Final salvage assessment	Nov 2009	Plant survival assessed after 12 mo of nursery care
Topsoil application	Dec 2009-Jan 2010	Stockpiled topsoil applied to old roadbed
Planting + treatments	Jan 2010	Salvaged plants installed in field; irrigation started
Field planting assessment	Mar 2010, 2011, 2012	Plant survival after 3, 15, and 27 mo in field

On site immediately after salvage, plants within species were randomly treated with either: 1) root-stimulating hormone (1H-Indole-3-butanoic acid; $C_{12}H_{13}NO_2$ [IBA]), at a concentration of 100 ppm in water, by dipping roots into the solution for a few seconds (Hortus USA Corp., New York, New York); 2) a 4 g/L slurry of Watersorb water crystals, a gel polymer designed to absorb and slowly release water to roots (Watersorb Corp., Fayetteville, Arkansas,); 3) IBA + slurry; 4) simply dipping roots in water; or 5) soaking roots in water overnight for 12-14 h before planting in pots the next morning. After treatment, we potted plants in 4-L (smaller plants) or 19-L (larger plants) plastic nursery pots filled with 1:3 organic mulch:sand. Plants were stored in a temporary field nursery (fenced, open at top) at Overton Beach and were given 3 cm of water each day through drip irrigation. The drip system operated 3× daily for 8 min each time.

Topsoil was salvaged by heavy machinery scraping the upper 5-20 cm of soil. The salvaged soil was stored in piles (1.5-3 m high) on site. Salvaged topsoil was available in sufficient quantity to place in a layer up to 5 cm thick on about three-quarters of area within restoration sites in December 2009. In January 2010, we transplanted salvage survivors using hand shovels by digging holes appropriately sized, to accommodate either the 4-L or 19-L volume of pots in which plants had been kept. We filled the holes with water and then transferred in the potting soil and plant. We gave each plant 1 L of water and enclosed them in circular cages (1 m tall and open at the top), made of 0.5-cm mesh hardware cloth, with the bottom of cages buried 3 cm deep and affixed to the ground using rebar.

Plants were randomly assigned one of three irrigation treatments: DRiWATER (a slow-release irrigation gel), hand watering, or no watering beyond that given at the time of planting. We followed manufacturer recommendations for applying DRiWATER, by placing an 8-cm diameter plastic tube into the ground, angled toward plant roots and the top (covered with a plastic cap) near the soil surface. We then inserted a cylindrical DRiWATER gel into the buried tube (DRiWATER Inc, Santa Rosa, California). We replaced gel packs monthly in summer (May through September) and every three months in cooler months (October through April). The amount of water delivered by DRiWATER is variable, dependent on how much water that roots extract (DRiWATER Inc, Santa Rosa, California). The hand watering treatment delivered 0.5 L of water to each plant once a month. This delivered 10 cm of water for the year, representing a 63% augmentation of the long-term average rainfall of 16 cm/year.

The original data collection recorded plant survival at 3, 15, and 27 months after transplanting, with the final assessment in March 2012. During the final assessment, we also counted live perennial plants in areas that had received or not received topsoil, but that had received no active revegetation (i.e. no salvaged perennial plants were planted, representing natural recruits).

The early results, reported in Abella et al. (2015a) and focusing on survival of transplants as a function of the topsoil treatment and treatments such as DRiWATER irrigation applied to outplants, were encouraging for transplant survival at 27 months. Survival of salvaged plants and seedlings transplanted back to field restoration sites varied statistically among species, lifeform, and irrigation treatment. While topsoil salvage exhibited $p = 0.11$, its effects were ecologically noteworthy. Overall, transplants on salvaged topsoil exhibited 56% survival, compared to 25% without topsoil. Salvaging topsoil increased survival by at least a third across lifeform and irrigation treatments and in some cases by up to 5×. Moreover, topsoil salvage alone (with no irrigation of plants) resulted in survival nearly equivalent to irrigating plants.

Example restoration plot that received salvaged topsoil and outplanting (left) and a disturbed, unrestored plot (right), Northshore Road Planting and Topsoil Addition project. Photos taken in March 2024 by UNLV staff (plots DIST04, 10).

Based on the 2024 inventory, plots with the combination of reapplication of salvaged topsoil and planting of salvaged perennials had the highest native plant community metrics. Total native plant cover averaged only 2% in disturbed/unrestored plots, was twice as high (4%) in plots receiving either only topsoil or only planting, and was highest at 5% in plots receiving both topsoil and salvaged plants. This was approaching native plant cover in undisturbed reference habitat, which averaged 7%. Also noteworthy is that *Ambrosia dumosa* was a dominant in undisturbed reference habitat with an average of 1.5% cover there. This species also had 1.5% cover in disturbed/restored plots, suggesting full recovery to the reference level.

Considering native annual food plants of desert tortoises, the combination of topsoil application + planting also performed well. Annual food plants of tortoises were absent in disturbed/unrestored plots and in plots with only planting, averaged 0.2% cover in plots receiving only topsoil, and averaged 0.5% cover in soil + planting plots. This cover in the soil + planting plots was the same as occurred on average in undisturbed reference habitat.

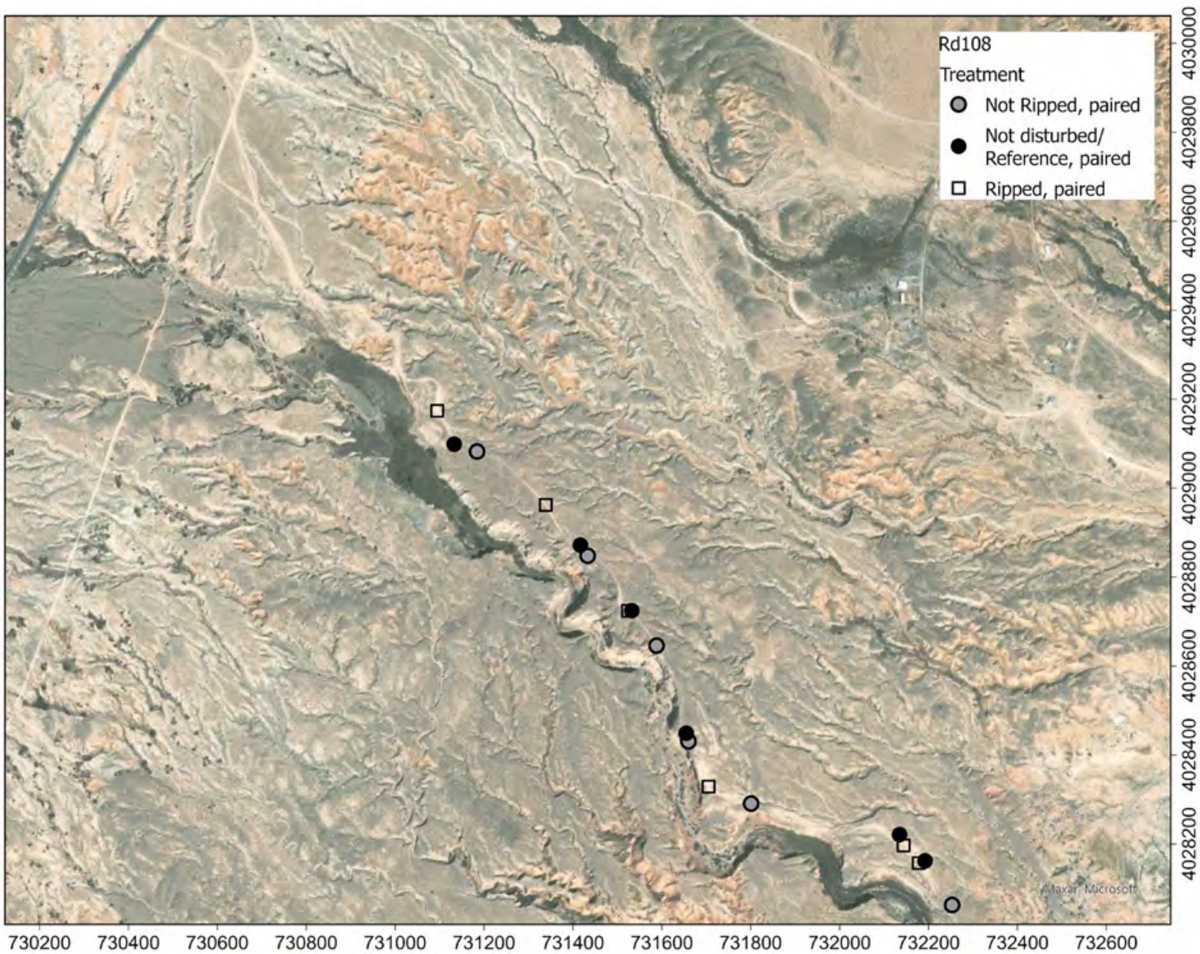
Plots receiving topsoil and transplants were the only ones on the disturbance to contain cacti. Cacti had a density of 8 individuals/100 m^2 on topsoil/planted plots, still lower than the 20 cacti individuals/100 m^2 in reference habitat but at least present. Cacti, such as *Opuntia basilaris*, had been among the best survivors of the transplanting. In addition to adding to biodiversity and providing other ecological functions, cacti can be important perennial foods of desert tortoises.

Overall, we consider the Northshore project to be successful based on several key metrics, such as foundational shrub cover, recovered to levels found in reference habitat or sharply higher in restoration plots compared with disturbed but unrestored plots. Although there are few other studies of the effectiveness of topsoil salvage and reapplication in southwestern deserts, topsoil application appears to have high potential to stimulate ecological recovery. Of course, one of the major challenges is identifying suitable donor sites. We suggest that evaluating whether some type of synthesized version of topsoil could offer at least some of the benefits of topsoil from natural desert habitats may enable this treatment to be more readily available.

Road 108 Soil Ripping

The road decommissioning and surface ripping project occurred within Lake Mead National Recreation Area (36°22'47.82"N, 114°25'25.57"W, 73 km northeast of the City of Las Vegas), managed by the National Park Service in the eastern Mojave Desert. Former approved road 108, a dirt road generally with no side berms, was closed to reduce the number of roads and associated disturbances in the area, which provided habitat on unique gypsum-carbonate soils for several endemic plant species. Soils in this area are marine-derived materials consisting of Typic Petrogypsids and Leptic Haplogypsids (Lato et al. 2006). Physical crusts often form on fine desert gypsiferous (calcium sulfate dihydrate, $CaSO_4 \cdot 2H_2O$) soils. Because of the unique properties of gypsum, unique surface biotic and plant communities occur, including rare and endemic species (Escudero et al. 2015), such as the critically endangered Las Vegas bearpoppy (*Arctomecon californica*), the rare southwest ringstem (*Anulocaulis leiosolenus* var. *leiosolenus*), and rare lichen and moss species, including Richards' peltula lichen (*Peltula richardsii*) and *Didymodon nevadensis*. Unlike most lower elevation Mojave Desert shrub communities in which creosote bush (*Larrea tridentata*) and white bursage (*Ambrosia dumosa*) predominate, undisturbed reference habitat adjacent to the restoration site includes the shrub Fremont's dalea (*Psorothamnus fremontii*) and the subshrubs shadescale saltbush (*Atriplex confertifolia*) and Parry's sandpaper plant (*Petalonyx parryi*), all of which are associated with gypsiferous or calcareous soils. According to land managers planning the restoration, the roadbed was heavily degraded and "the damage [was] so severe that all the topsoil [was] eroded away leaving exposed bedrock," which resulted in visitors "choosing to drive off road to avoid the undesirable designated path" (disturbance assessment, Lake Mead National Recreation Area, 2009). Intending to decompact and roughen the surface and enable native plant recruitment, a subsoiler was used to rip surface material (30 cm) on the roadbed in alternating 150 m intervals in November 2002.

In spring 2009, seven years after soil ripping, we identified three sets of ripped and unripped segment pairs and installed in each segment a 33.3 m × 3 m (100 m^2) plot. Adjacent and parallel to each plot pair and 20 m away from the old roadbed, we established undisturbed reference plots using the same dimensions and methods. In each plot, we categorized the aerial cover of each plant species rooted in plots using cover classes. We counted all seedling and adult perennial plants within plots. We repeated vegetation surveys in 2016 (14 years post-treatment), 2019 (17 years), and 2024 (22 years).



The entrance to Approved Road 108 from Northshore Road in 2002 prior to decommissioning and ripping treatments (left) and four days later after decommissioning, ripping, and fencing was installed (right). Photograph provided by Lake Mead National Recreation Area archives.

Example restoration plot that received ripping (left) and a plot on the same decommissioned road but that was unripped (right), Road 108 Soil Ripping project. There is a narrow trail shown in the center of both photos, as the road was closed to vehicles and converted pedestrian access only via the trail. Photos taken in March 2024 by UNLV staff (plots RIP06, NRIP02).

Although Road 108 occurs on difficult, shallow soils, the ripping treatment has been highly effective at facilitating natural plant establishment based on the 2024 inventory. Adult shrub density was over 7× higher (750 compared with 100 shrubs/100m^2) on ripped compared with non-ripped sections of the road. Shrub density is fairly low in nearby undisturbed reference habitat (983 shrubs/100 m^2), and the ripped section now approaches reference levels. Indigo bush is a dominant shrub in undisturbed reference habitat with 1.8% cover. The ripped road sections had recovered nearly all of that cover, with an average of 1.4% cover, compared with only 0.1% cover in non-ripped sections.

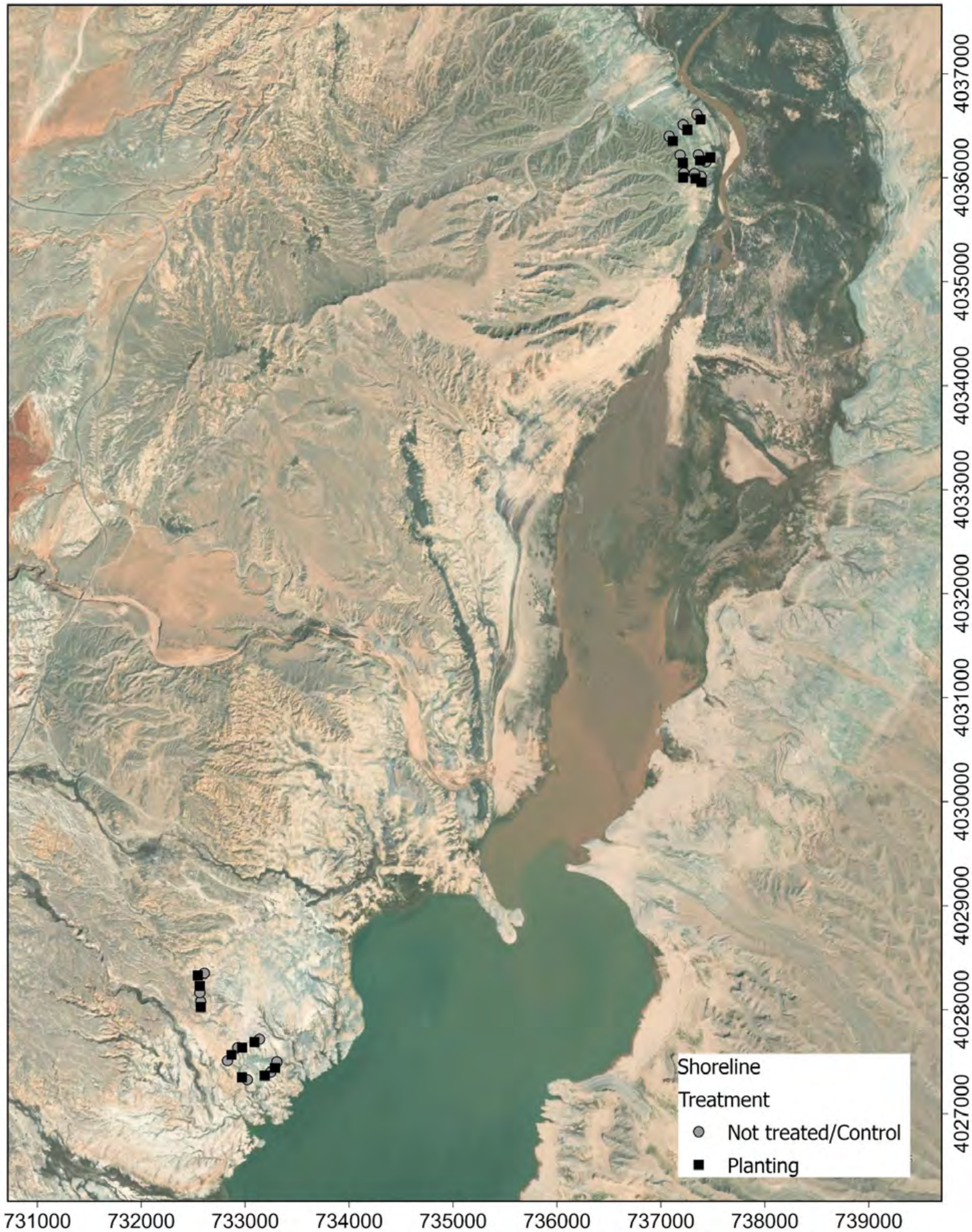
Although seedling populations of desert perennials can be quite transient with high mortality, seedling density was also much higher on ripped sections (167 seedlings/100 m^2 in total for all shrub species) than on non-ripped (17 seedlings/100 m^2). Reference habitat contained 100 seedlings/100 m^2. This observation suggests the possibility of continued shrub recruitment on ripped sections in years when seedlings survive. Annual plants of native and non-native species were sparse on the road regardless of treatment, but in 2024 they were also sparse in reference habitat. At least in 2024, there was no evidence that the ripping treatment promoted non-native plants.

As shown in the preceding example photos, the ripped sections including with their plant cover have blended the road disturbance into the surroundings, whereas the non-ripped portions still appear as a distinctive, compacted road even 22 years after closure. Interestingly, the soil compaction average rating in 2024 was identical (3.5) between ripped and non-ripped sections of the road. Soil compaction averaged 2.4 in the reference, indicating less soil compaction. The ripping may have functioned to create heterogeneity in the soil surface to facilitate plant recruitment, even if the underlying soil remained compacted.

As a relatively old restoration site (22 years), Road 108 may offer perspective for newer restoration projects on decommissioned roads that have not yet shown much success. Using the pre-existing long-term data available for Road 108 dating back to 2009, we plan on producing a value-added case study of this project for DCP to enable determining at what point plant colonization may have occurred after ripping. For example, it is possible that a series of wet years can provide a window of opportunity for plant recruitment, and a project first needs to experience that to be successful if relying on natural recruitment. Alternatively, perhaps new plants have accumulated gradually over the decades. Determining this could help inform what patterns may unfold on newer restoration projects on decommissioned roads, including those discussed in this report (e.g., SNWA Endcaps project discussed after the Shoreline project).

Shoreline Planting along Lake Mead

This outplanting project occurred along the Lake Mead shoreline that became exposed during the ongoing drawdown of the lake level. The project was initiated by the National Park Service in 2010 through a cooperative agreement with the University of Nevada Las Vegas (UNLV). The goal of the project was to determine whether active revegetation could accelerate the recovery of desired species along the newly exposed shoreline and shift habitat away from primarily either persistent bare ground or dominance by non-natives such as salt cedar.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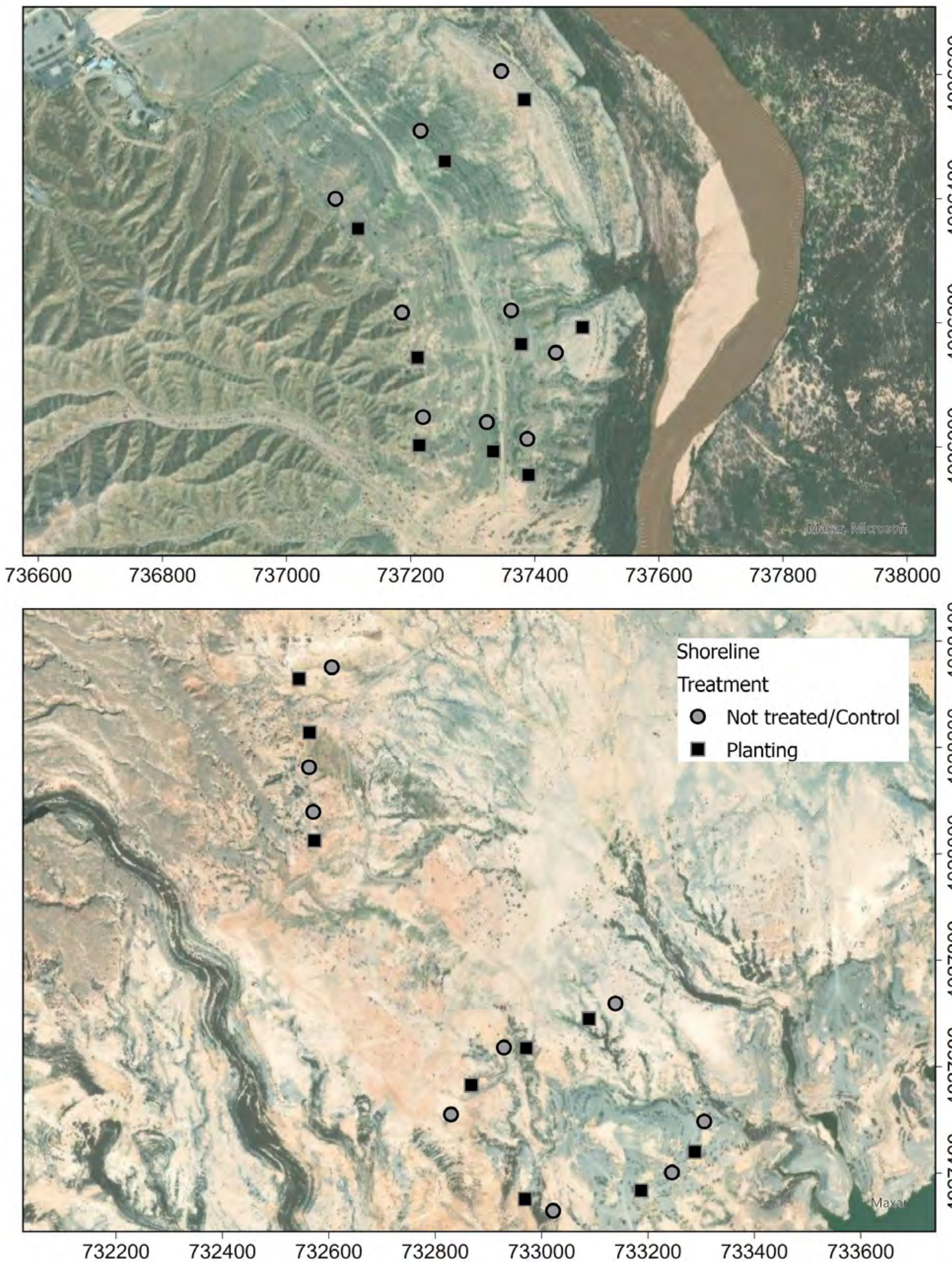
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Individuals of four species of native perennial species (*Ambrosia dumosa*, *Encelia farinosa*, *Hymenoclea salsola*, and *Larrea tridentata)* were propagated by the Lake Mead NRA nursery staff and outplanted (along with Driwater) by UNLV staff in an experimental design addressing survival across different post-submersion environments. UNLV staff aided LAKE nursery staff to sow seeds for all species intended for outplanting in January, 2011. Seedlings were outplanted in December 2011. We chose species that were common in never-submerged plots across all sites, with the idea that those species would have been present pre-submersion and therefore they would be the most appropriate candidates for active restoration. The goal of the outplanting experiment is to evaluate whether the changed soil and environmental conditions post-submersion would be hospitable for establishment of these species if outplanted.

Lake levels rose 12 m between November 2010 (project inception) and November 2011, flooding the lowest elevation (2008) plots. Additionally, with this rise in water levels, the visitor use areas of Boulder Beach were augmented to create more useable beach area, resulting in bulldozing some of the second lowest elevation (2005) plots, and leaving others very close to active recreation sites. After evaluation of the soil analysis data, we found that soil texture and composition was similar between Boulder Beach and Overton Beach sites. Therefore, due to high visibility to park visitors at the Boulder Beach plots, much disturbance, and flooding of the lowest elevation plots we decided to restrict outplanting experiments to the Stewarts Point and Overton Beach sites.

For the outplanting, we chose three transects at the Stewarts Point and Overton Beach sites, and outplanted plants at elevations equivalent to lake levels in 2005, 2002, and 1998. The number of transects planted were guided by the availability of suitable plant material. We instituted a minimum size of plant material for planting, individuals needed to be at least 10 cm in height. Our goal was six replicates per plot, but we agreed that four would be sufficient for the *L. tridentata*, which was the species with the most limited availability of plants of sufficient size. Reducing the number of replicates of *L. tridentata* per plot allowed us to plant plots along three transects per site. In each plot, we outplanted six individuals each of *A. dumosa*, *E. farinosa*, *H. salsola*, and four individuals of *L. tridentdata*. The number of outplanted individuals were largely determined by the availability of plant material. Each outplanted individual was outfitted with DRiWATER, and was monitored until late November, 2012 for survival.

We chose transects based on random assignment coupled with deliberately choosing transects whose plots in the monitoring portion of the study had low *Tamarix ramosissima* density. We believed this would better represent regions that would more realistically be candidates for restoration treatments due to an assumed higher survival probability from less active competition from saltcedar, and would better mimic active management practices. Sites were located adjacent to the community composition plots where soils should be equivalent to sampled and analyzed soils. Because we were planting along transects, not all plots along the transect would necessarily have sparse *T. ramosissima*. Rather, we selected transects as a whole with relatively low *T. ramosissima* abundance that also fit prerequisites of low disturbance and easy to moderate accessibility (also mimicking active management practices).

Seedlings were planted Dec 20, 2011 – Jan 4, 2012. Plots were arranged in a 4 × 6 grid such that each seedling was 2 m away from the nearest neighbor. Plants were randomly assigned grid positions for each plot. Individuals were planted traditionally using trowels and given supplemental aboveground water application (500 mL per plant) sufficient to saturate the soil around the plant in a 15-cm-diameter area. Each plant also had a DRiWATER installed at the time of planting. Each plant was outfitted with a hardware cloth cage to prevent herbivory made of 1-cm hardware cloth mesh, 50 cm tall, and 40 cm diameter (image below).

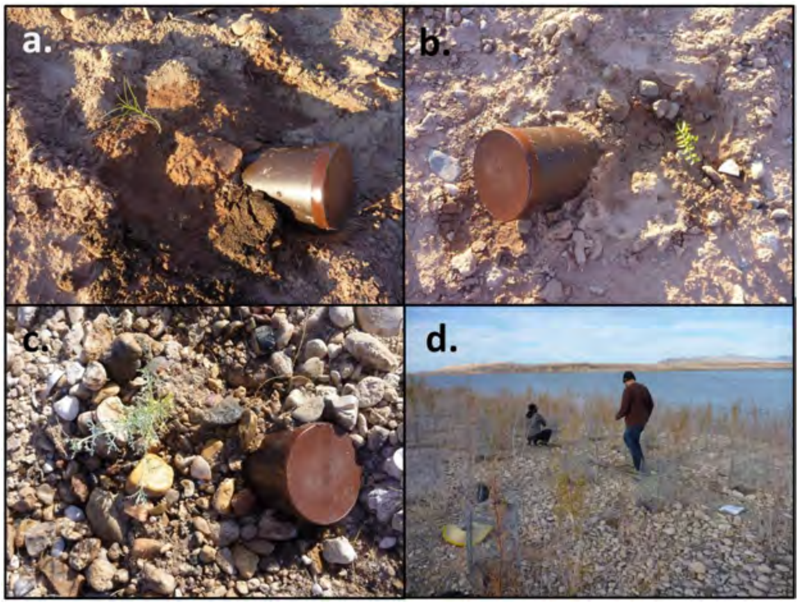
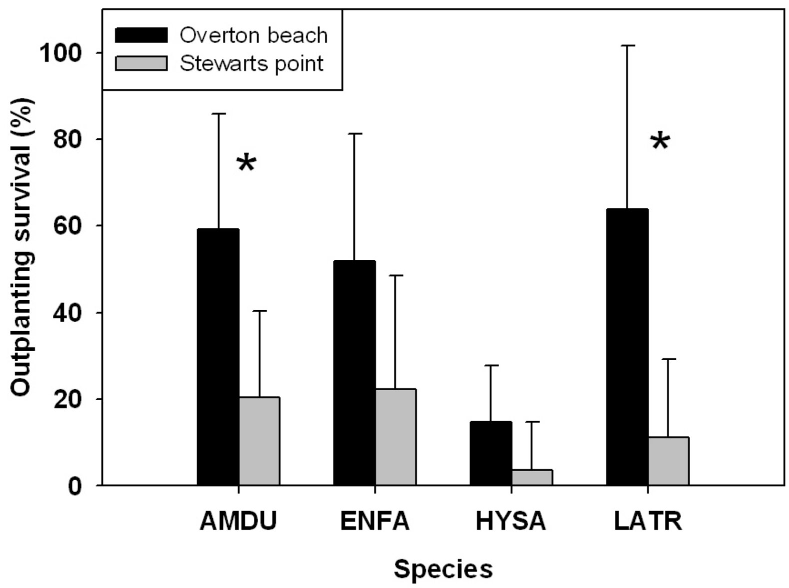


Image of the original outplanting along the Lake Mead shoreline, December 2011 – January 2012. *Hymenocla salsola* (a), *Larrea tridentata* (b), and *Ambrosia dumosa* (c) individuals shown on the day of outplanting, and (d) an example of a nearly finished plot with plants and protective cages installed at the 2005 lake elevation along the Overton Beach shoreline.

Plants received DRiWATER when it was depleted (approximately every 8-10 weeks in winter, every 4-6 weeks in summer). We assessed survival rates and performed upkeep on cages where necessary approximately every two months (generally coinciding with DRiWATER application) throughout the year after planting. The original outplanting survival recording was the last week of November, 2012, representing about one year after planting. Plants were labeled “live” if any green leaves were visible, or – when no leaves were present – if stem tissue was pliable. Plants were classified as dead if they were missing, lacking leaves, or fine stems were brittle.

The early results, one-year survival of outplants in 2012, were reported to the National Park Service but otherwise not publicly presented. Thus, these early results are summarized below, followed by the 2024 DCP re-inventory of the sites.

As of one year after planting, 120 (30%) out of the 396 individuals planted were alive. Survival rates were similar among *A. dumosa*, *E. farinosa*, and *L. tridentata*, all ranging from 37-40% survival. *H. salsola* performed poorly across the planting, with survival just under 10%. Time since submersion had no overall or interactive effect on outplanting survival ($p > 0.05$). Rather, planting site (Overton Beach or Stewarts Point) dictated survival rates ($p < 0.0001$, shown below). Survival rates also differed among species where *H. salsola* had lower survival rates than all other species ($p = 0.001$). Planting occurred within a two-week timeframe and all plants were randomly assigned plots from a common source so that variation in size and age were equivalent among sites. Thus, differences in soil properties are likely the driver for the differences in survival rates among the two sites (discussed in Engel et al. 2014).



Survival (mean ± 1 S.D.) among each of four species planted. Species codes are as follows: AMDU = *Ambrosia dumosa*, ENFA = *Encelia farinosa*, HYSA = *Hymenoclea salsola*, and LATR = *Larrea tridentata*. (*) represents differences between sites, within a species (p < 0.05).

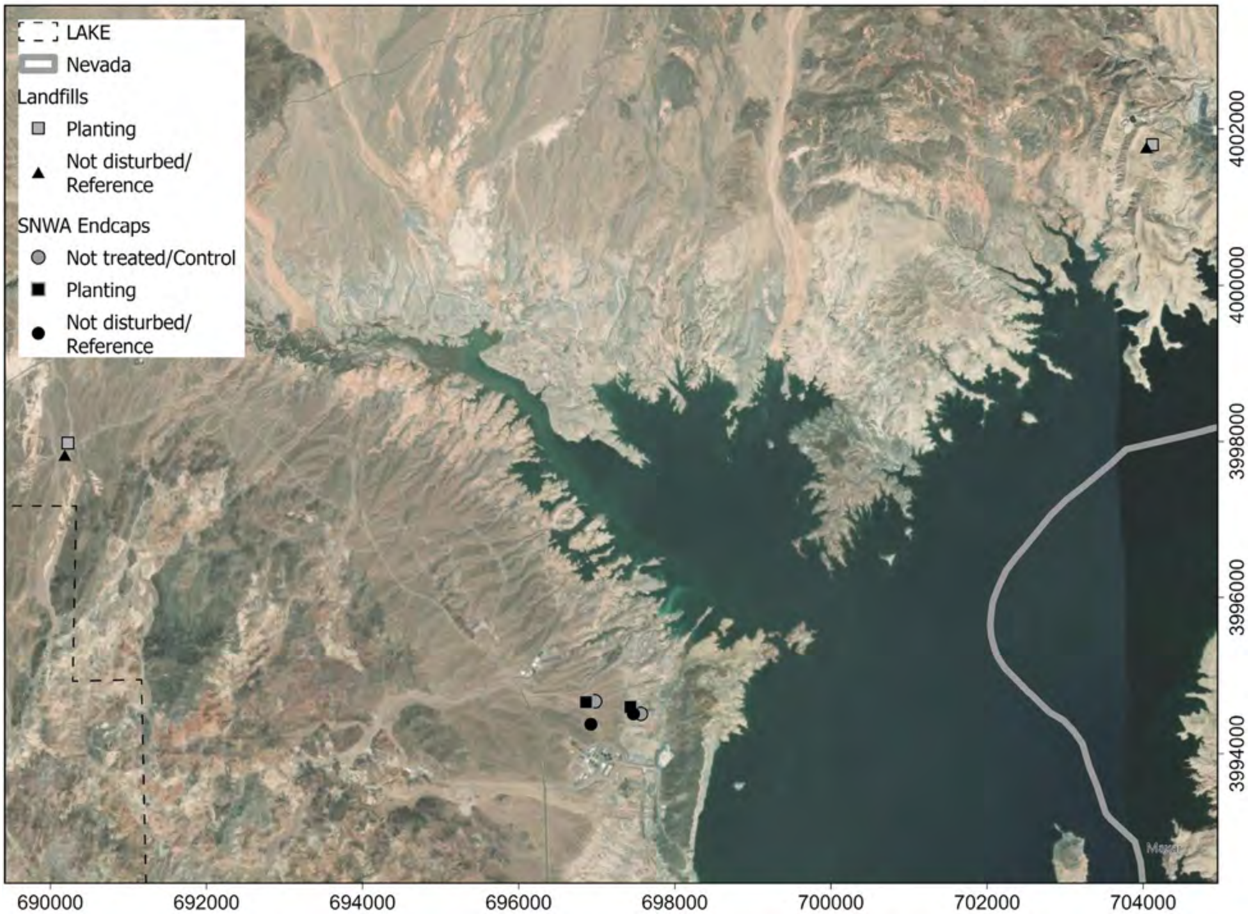
As of the 2012 survival analysis, the data suggested that site rather than time since submersion determined differences in survival rate within a species. Even in species that did not significantly differ across sites (*E. farinosa* and *H. salsola*) trends indicated that survival rates were greater in the Overton Beach plots than at Stewarts Point. The Huevi association soils like those at Overton Beach are more prevalent throughout LMNRA. Huevi soils are very deep, well drained soils that formed in mixed gravelly alluvium, and is the most dominant soil group along the shoreline of Lake Mead. The huevi soils appear to be better general planting habitat than the gypsum-rich soils found at Stewarts point. The soils at Stewarts point overall were more loamy and silty, which should be better for water retention, but also had greater electrical conductivity values, indicating greater salinity – likely consisting of various mineral salts in addition to deposition from tamarisk. The Stewarts Point sites were the only site where time since submersion – and correlated tamarisk abundance – did not predict soil salinity because of the naturally occurring compounds. These soils are better for establishing specialist watch species like *Arctomecon californica* observed occurring naturally (and re-establishing below the lake full pool).

By 2024, most of the outplanting plots did not contain any live individuals of the outplanted species. There were a few plots that had high cover of *Larrea tridentata*, such as one outplanted plot that had 20% Larrea cover. Overall, species composition of the plots along the shoreline was highly variable, driven primarily by non-planted species. For example, some plots had high cover (8-10%) of the non-planted *Bebbia juncea*, which interestingly struggled in outplanting in many of the other projects. The native shrub *Psorothamnus fremontii* had 2-3% cover on some plots. The non-native annual forb *Brassica tournefortii* predominated on some plots, with 20-30% cover, while the non-native *Schismus* spp. predominated on others, with 10% cover. The non-native tree *Tamarix ramosissima* was also sporadic, absent from about half of plots while attaining high cover of 15-20% on others. We surmise that site factors such as soils and natural seed rain may be primarily driving communities on the restoration plots rather than the outplanting 12 years ago.

Example plot receiving outplanting (left photo) and a non-planted plot (right) both in the drawdown zone of Lake Mead, Shoreline Planting project. Photos taken in March 2024 by UNLV staff (plots OB5P, OB5C).

Southern Nevada Water Authority Endcaps Planting, Soil Amendment

This project occurred along a decommissioned, dirt road (~ 5 wide, generally without side berms) and was intended to blend the disturbance into surrounding, mature creosote bush-white bursage (*Larrea tridentata-Ambrosia dumosa*) shrubland. The restoration site was within Lake Mead National Recreation Area, administered by the National Park Service, in the eastern Mojave Desert, southern Nevada. Restoration activities on the road included ripping the surface and moving boulders back on to the road surface, followed by outplanting in two phases in March and December 2020. Using seed collected from within the recreation area and including species inhabiting undisturbed reference areas adjacent to the restoration site, seedlings were propagated in bands and in 4-L containers at the park's nursery (Song Dog Native Plant Nursery, Boulder City, NV). In March 2020, 100 creosote bush and 25 white bursage seedlings were outplanted in a random pattern along the ripped road. In December 2020, another 147 creosote bush and 174 white bursage were also outplanted.



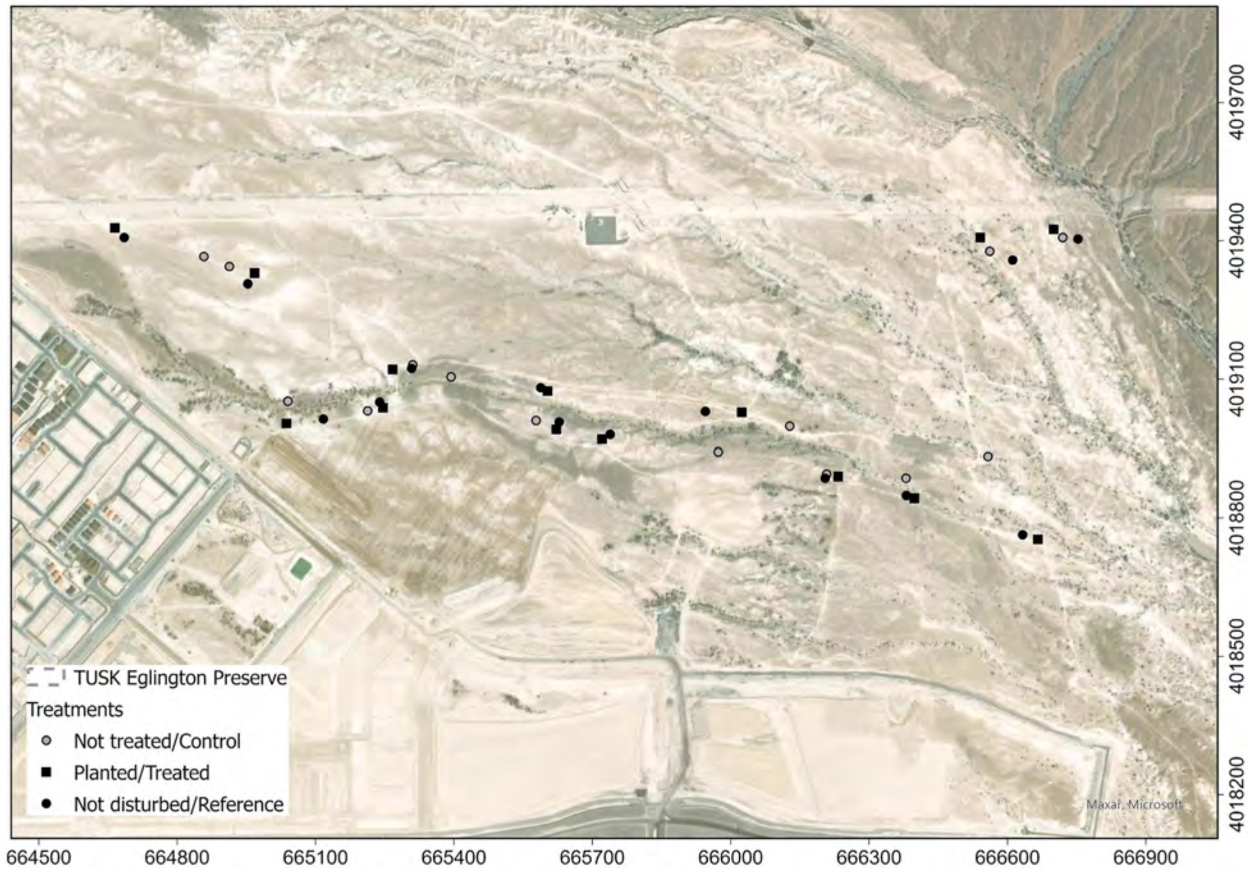
The SNWA Endcaps restoration site, after soil ripping and placement of boulders, during outplanting in March 2020 (National Park Service photo).

Example restoration plot receiving soil ripping and outplanting (left photo) and a control plot (right) both on the same decommissioned road, SNWA Endcaps Planting project. Photos taken in March 2024 by UNLV staff (plots SNWA1P, 2C).

Based on the 2024 inventory of the SNWA Endcaps project, four years after the soil manipulation and outplanting treatment, there was little indication that the treatments promoted the outplanted species or other native species. The two outplanted species, white bursage and creosote bush, occurred on only one of the two restoration plots and at < 0.02% cover. There was little difference in other vegetation metrics, such as native annual cover, between disturbed/restored and disturbed/unrestored plots. As shown in the preceding photos, the decommissioned road remained visually distinct from the surrounding, creosote bush-dominated reference habitat. It is possible that additional protection or irrigation of outplants may have been required to facilitate greater establishment of the outplants, as severe drought afflicted southern Nevada after the outplanting.

Tule Eglington Planting

This restoration project included a variety of soil amendments and mulches, but in our DCP project, we focused on areas where outplanting occurred as we could verify their locations by the presence of cages still present in 2024 encircling outplants.



Within Tule Springs National Monument administered by the National Park Service, this project sought to restore desert habitat on 4 ha of decommissioned dirt roads using outplanting. Seed for propagating the plants was locally collected within and around the monument. Seedlings were mostly propagated in 1-gallon containers and were produced by four nurseries: RECON Native Plants (Sand Diego, CA), Signature Botanica (Wickenburg, AZ), Nighthawk Natives Nursery (Tucson, AZ), and the Nevada Division of Forestry (Las Vegas, NV). Species chosen for outplanting (listed below) occurred in undisturbed reference habitat in the monument and included a variety of growth forms. Approximately evenly spaced within 26 outplanting locations, 3896 individuals of 15 species were outplanted at a density of ~ 1,000/ha. Outplanting occurred between November 2010 and November 2012. Although there was a lag of 4+ months from the start of outplanting in November 2010 until protective cages were installed, outplants were enclosed in wire mesh cages (2.5 cm openings) that were 60 cm tall and affixed to the ground using two rebar stakes. Soil in planting holes was watered to field capacity at the time of outplanting and to field capacity monthly in 2011 and 2012 after outplanting. In December 2012 when the maintenance period by the contractor ended for the project, outplant survival was reported to be 48% (1857 of 3896 outplants).

Survival in December 2012 for 15 native perennial species outplanted between November 2010 and 2012 to revegetate decommissioned roads in Tule Springs National Monument. These data were provided by the contractor who performed the outplanting and maintenance (RECON Environmental, Inc., Tucson, AZ).

Species	Growth form	No. planted	No. alive	Survival (%)
Ambrosia dumosa	shrub	1995	615	31
Atriplex canescens	shrub	350	158	45
Atriplex polycarpa	shrub	18	10	56
Chilopsis linearis	tree	30	8	27
Ephedra nevadensis	shrub	48	23	48
Eriogonum corymbosum	shrub	203	48	24
Gutierrezia sarothrae	forb	56	23	41
Larrea tridentata	shrub	630	535	85
Prosopis glandulosa	tree	14	12	86
Salazaria mexicana	shrub	86	76	88
Senegalia greggii	tree	50	45	90
Sphaeralcea ambigua	forb	83	26	31
Stanleya pinnata	forb	235	188	80
Xylorhiza tortifolia	forb	44	40	91
Yucca schidigera	shrub	54	50	93

Example restoration plot receiving outplanting (left photo, including showing cages still present) and a disturbed, control plot (right) both on disturbances on decommissioned roads, Tule Eglington Planting project. Photos taken in March 2024 by UNLV staff (plots Tule9P, 9C).

Although a diverse mixture of 15 native perennials were outplanted, the 2024 inventory of the Tule project revealed low native plant cover and minimal difference in native cover between restoration and disturbed, control plots. Total native cover averaged 1.7% in restoration and 1.6% in control plots, compared with 6.8% in undisturbed reference habitat. Native annuals, including desert tortoise forage plants, were nearly absent on disturbed plots, regardless of restoration (0.06% on restoration and 0.05% on control plots). Non-native annuals were also sparse, averaging only 0.12% on both restoration and control plots, compared with 1.0% on undisturbed reference plots. The overall conclusion is that vegetative cover is nearly absent on disturbed plots, regardless of restoration.

This restoration outcome seems particularly disappointing at this site given that the early results of outplant survival reported in the table previously mentioned were encouraging. This project highlighted that the early results did not necessarily predict later results, underscoring the utility of a long-term perspective to restoration success.

There could be several reasons for the minimal establishment of outplanted species and minimal evidence of facilitating further recruitment. Although cages to deter herbivory were still present on the restoration sites around outplants, it was noted at the time of outplanting that there was a delay between planting and when the cages were installed. Plants were noted to have been damaged or even removed in the intervening time, such that the early survival may not have fully represented the long-term prognosis. For example, a seedling could have been eaten down yet still be recorded as alive, but weakened to the point that death would ensue. A second potential reason for the difficulty could be that soils in this project are classified as Petrocalcids, which are gravelly, shallow soils on which natural plant establishment can be difficult. Third, soils on the decommissioned roads remained compacted and with little surface structure to limit water runoff and retain moisture or nutrients. Biocrust cover averaged 0.23% in reference habitat but was nearly non-existent (< 0.02% cover) on the road surfaces regardless of outplanting.

In meetings with Tule Springs park staff during the project, there remains much interest in developing reliable restoration practices for the unique soils and conditions of the monument. This is especially true given the recent establishment of the monument and the long preceding history of roads and off-road vehicle use before park establishment. We suggest that a next set of treatments that could be tried on these difficult soils include mixing in abiotic treatments (e.g., soil surface manipulations, vertical mulch) and potentially an intensive type of outplanting using potential early colonizers on these soils, such as *Atriplex*. Intensive outplanting could include using shelters (which also provide microclimate amelioration in addition to the herbivory deterrence offered by the wire cages) and irrigation. Although fertilization is not often recommended for desert outplantings due to it being unnecessary and potentially promoting non-native plants, it is possible that this is a case where fertilization could be assessed for its ability to aid outplant survival.

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ADDITIONAL MATERIALS

Also generated by this project and included with this report are the following:

- Digital photos of each of the 363 plots, including multiple images of each plot
- Geospatial data for plots for each project (shapefiles, layer files, kml files)
- Excel database containing all project meta-data, plot descriptive information (e.g., locations, elevation, Clark County soil survey taxonomy), and soil condition and vegetation plot data
- As mentioned in this report as value-added for this DCP project, we have communicated plans to the DCP Technical Representative to develop separate analyses and scientific manuscripts for a subset of projects that had existing, long-term plot community data and that were remeasured in 2024 as part of this DCP project. These five projects include: Bonnie Springs Fire Seeding, Fish Hatchery Planting and Soil Amendment, Jean Large-Scale Translocation Site (LSTS) Seeding, Northshore Road Planting and Topsoil Addition, and Road 108 Soil Ripping.

Included in the following appendix is an already completed example of above, for the Bonnie Springs Fire Seeding project. This case study example is in review with the practitioner-oriented journal Ecological Restoration. The manuscript is under consideration for a themed section of the journal featuring restoration projects associated with wildfires nationwide and is the only one representing a desert ecosystem. This completed writeup shows an example of what we are planning for the other four projects, which generally also achieved higher restoration success.

APPENDIX: BONNIE SPRINGS FIRE SEEDING MANUSCRIPT

The following manuscript submitted to the practitioner-oriented journal Ecological Restoration is value-added to this DCP project and presents long-term results including the 2024 DCP project remeasurement.